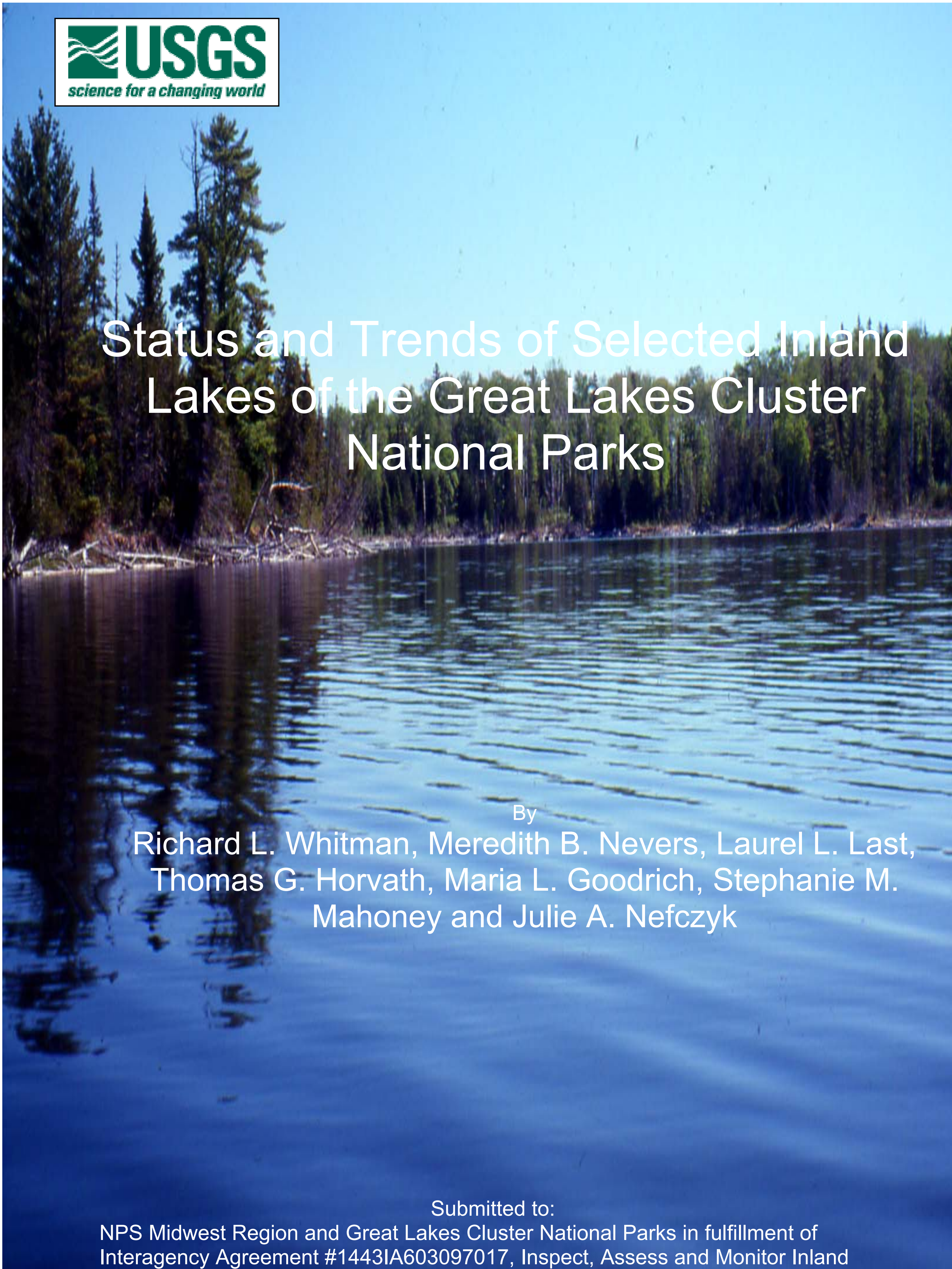

Status and Trends of Selected Inland Lakes of the Great Lakes Cluster National Parks

By

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Chapter 1 Executive Summary

In 1994, the need to inventory and monitor inland lakes was listed as the top priority for the Midwest region of the National Park Service. This issue has been among the high priorities for the National Parks for the last decade. In addition, it has been listed as a high priority in the majority of individual park's resource management plans. In response to these needs, Lake Michigan Ecological Research Station, Great Lakes Science Center, U.S. Geological Survey submitted a proposal to the National Park Service to study inland lakes of the Great Lakes parks and to develop a monitoring plan that would track lake status and trends. The funds were awarded in 1996, a formal study plan was accepted by the National Park Service Washington DC office in May 1997, and sampling began in May of 1997.

The environmental status and biological composition of the inland lakes of the Great Lakes Cluster Parks (Indiana Dunes National Lakeshore—INDU, Sleeping Bear Dunes National Lakeshore—SLBE, Pictured Rocks National Lakeshore—PIRO, Isle Royale National Park—ISRO, and Voyageurs National Park—VOYA) are poorly known. These lakes differ significantly as to atmospheric influence, exposure to cultural pressures, and trophic status. Very few have been inventoried, and fewer are biologically monitored. External threats to these lakes differ considerably in source, causation, and relative impact. Organic and metal non-point contaminant deposition and exotic species invasions are of a concern in the otherwise pristine waters of northern Great Lakes parks (PIRO, ISRO, and VOYA). At mid-latitude, SLBE is experiencing accelerated eutrophication due to point and non-point phosphate input. At the southern extreme, INDU lakes face acute degradation due to atmospheric, groundwater, and surface water contamination. Water quality monitoring programs exist at a few of these parks (INDU, APIS), but almost none have biological inventories or operational biomonitoring programs for their lakes. Exotic species such as purple loosestrife and zebra mussels may disrupt biological communities and ecological processes.

The objective of this study was to inventory reference locations at selected lakes in the Great Lakes Cluster Parks and to install uniform, scientifically defendable biomonitoring programs that could detect long-term ecological trends. The lakes selected were Long Lake—INDU; North Bar Lake, Loon Lake, and Round Lake—SLBE; Grand Sable Lake and Beaver Lake—PIRO; Sargent Lake and Siskiwit Lake—ISRO; and Mukooda and Locator Lake—VOYA. Inland lakes' reference stations were characterized as to their biological and limnological conditions. Environmental problems were identified and lake fitness established.

A baseline lake inventory was established over the course of the study. The lakes were sampled through two full summer seasons (May-September 1997 and 1998) and an additional month (May 1999) for physical, biological, and chemical variables. Physical variables included lake profiles for dissolved oxygen, temperature, pH, and specific conductance. Biological communities sampled once a month were the phytoplankton, zooplankton, and benthic macroinvertebrates. Water and sediment chemistry and chlorophyll *a* were all measured. Water chemistry was sampled monthly in the epilimnion, hypolimnion, and littoral zones for chlorides, total hardness, total alkalinity, ammonia, nitrates, total phosphorus, and sulfate. Sediment samples were collected monthly in 1997 for nutrients and once for pesticides, volatiles, and metals. Phytoplankton and zooplankton were collected in the littoral zone in 1997, and benthic macroinvertebrates were collected from the littoral zone throughout the study. Each of the study lakes at INDU, SLBE, PIRO, and ISRO were also surveyed for macrophyte communities.

Discriminant analysis showed that with few exceptions, the Great Lakes Cluster Parks lakes could be separated based on alkalinity, chloride, or silica. Siskiwit and Mukooda, and North Bar and Loon Lake had substantial overlap for alkalinity and chlorides. Round Lake clearly separated out by alkalinity alone. Water quality was clearly different at the Sleeping Bear Dunes and Indiana Dunes lakes than other lakes studied. Differences in general water chemistry were likely due to differences in basin geology. Only Long Lake showed clear signs of eutrophication. No substantial water quality degradation was identified at northern lakes.

Organic pesticides and PCBs (measured as Aroclors) were generally not detected. Where detected, they were below the consensus-based sediment quality guidelines or other benchmarks. Metals and one PAH exceeded the benchmark values in the limnetic zone at three of the lakes—Beaver, Sargent, and Siskiwit—and in the littoral zone at Siskiwit Lake.

Diatom populations dominated the phytoplankton of most lakes except for Long Lake and Beaver Lake, which had high euglenoids and yellow-green algae. Generally, lakes displayed pulses of algal growth in late spring and early fall. Organism richness ranged from 30 taxa at Beaver Lake to 49 taxa at Siskiwit Lake. Organism evenness was highest at Beaver Lake. High density for phytoplankton was generally around 10^6 cells or colonies per liter. Chlorophyll *a* did not track phytoplankton density counts well.

Except for Siskiwit Lake, rotifers dominated the zooplankton. Common species included *Keratella* spp., *Kellikotia longispina,* and *Conochilus unicornis*. Species diversity indices ranged from 1.7 for Beaver Lake to 2.5 for the Isle Royale Lakes. Lakes in the Lake Michigan parks had typical temperate microcrustacean species while lakes in the Lake Superior parks and Voyageurs had species representative of northern, oligotrophic diluted lakes (*Epischura lacustris*, *Skistodiaptomus oregonensis*). Siskiwit Lake clearly had the lowest zooplankton density, and Long Lake had the highest. Typically, lakes recorded 30-38 taxa of zooplankton in each sampling year.

Benthic macroinvertebrates in the littoral zones of lakes were generally more diverse but not necessarily higher in density than in the limnetic zones. Non-biting midges were the dominant species in the limnetic zone and most common in the littoral zone. The macroinvertebrate community in Siskiwit Lake differed in that the littoral zone had lower diversity than the limnetic zone, and the dominant deep water invertebrate was *Diporeia*. The Beaver Lake littoral zone had the highest mean macroinvertebrate densities and the highest Margalef diversity index. The littoral zone for Locator Lake had the lowest Shannon-Wiener diversity index. Limnetic zones in Mukooda and Locator were among the lowest in benthic density. Grand Sable Lake had the lowest Shannon-Wiener diversity, but otherwise most lakes had similar diversities. Species evenness in the limnetic zone was similar for all lakes studied although Loon and Mukooda tended to be higher.

The number of recorded plant taxa ranged from eight in North Bar and Siskiwit lakes to around twenty in Grand Sable and Sargent lakes. Lake size or geographic location does not appear to control either of these variables. Combinations of other factors, such as water clarity and substrate type, likely have a strong influence on the macrophyte communities. Nuisance species such as cattails, *Phragmites,* and purple loosestrife at Long Lake, *Phragmites* at Round Lake, and *Potamogeton crispus* from Loon Lake were noted. Plants were patchy in most of the lakes surveyed, with plant cover ranging from open water to extremely dense.

Options for a routine monitoring program are presented for individual parks or cooperative clusters of parks. The monitoring program protocols are described in an accompanying field handbook and in the methods section of the report. We developed interpretive materials including the field manual that are intended for use by resource managers, park managers, federal land trustees, and other scientists. Study results and data have been made available through reports, national databases, the Internet, and National Biological Information Infrastructure.

Chapter 2 Biological Monitoring

Changes in a natural lake system, whether natural or anthropogenic, can be detected with an effective monitoring program. Current conditions, ecological history, and the surrounding watershed all help in determining a lake's response to change. Short-term changes in a lake system can result from nutrient inputs, introduction of non-native species, overuse, or agricultural discharge, but each lake will be affected differently. The only way to observe these changes effectively is through a lake monitoring program. Traditionally, monitoring programs have been short-term and narrowly focused (Bricker and Ruggiero 1997). This type of monitoring provides little information regarding human impacts to the system because many impacts go unnoticed for many years. Magnuson (1990) refers to a single year of monitoring as "the Invisible Present." Extensive data may be collected, but without information from previous years, comparisons are not possible. Developing a long-term program can provide context for determining impacts and ecological processes (Herrmann and Stottlemeyer 1991). Furthermore, expanding long-term monitoring to watersheds or regional ecosystems can improve monitoring effectiveness even more by increasing timing and location accuracy (Skalski 1990, Herrmann and Stottlemeyer 1991). The scope and potential threats must be considered before beginning a monitoring program.

In order to create an effective monitoring program that will last well into the future, it must be both technically and statistically reliable (Dixon and Chiswell 1996). Just as Magnuson (1990) recognizes a year of useless data, it is possible to collect many years of unusable data if a monitoring program is not based on sound science. Ward et al. (1986) state that a monitoring program should be established as a complete system, recognizing all of the steps necessary to achieve the goals. These steps, however, may vary depending on the type of lake being monitored, and some flexibility must be built into the design (Stow et al. 1998). Without initial program design, collected data will be neither usable nor suitable for manipulation into useable data (Ward et al. 1986).

There have been numerous approaches to monitoring lake status including the lake watershed (Herrmann and Stottlemeyer 1991), trophic status (Carlson 1977), and biological community structure (Herricks and Cairns 1982). Modes of sampling for monitoring programs are also highly varied, and preferences range from fixed-point stations (Murray 1987) to station pairing (Dixon and Chiswell 1996) to reference sites (Hughes et al. 1994). With the range of aquatic ecosystems that are monitored, it is unlikely that one monitoring program will suffice for all conditions, so variations are sought. The added variables of time, resources, available personnel, and expertise complicate monitoring decisions further. In order to determine the most fitting monitoring techniques for a particular system, various approaches should be reviewed and considered before committing to a long-term plan.

Before constructing a monitoring program, background information on the lakes being considered is needed. A familiarity with the resource will help in assessing the threats to the ecosystem and potential impacts. Background information or baseline data is often gleaned from past research, state testing, and historical records. It may also be helpful to collect observational information from local residents and resource managers. Many lakes are sampled occasionally during the course of different research projects, but collective, continuous data are often unobtainable. Research that is directed toward establishing a monitoring program often involves the collection of baseline data, which typically includes varied and numerous data. Long-term data sets exist for some ecosystems that are often composed of a single variable collected over time, such as the ice history data available for Lake Mendota that covers 132 years (Magnuson 1990). Longer-term limnological history can also be valuable for assessing the rate of change in a system, which is often due to anthropogenic sources. With a sediment core, paleolimnological analysis can be used to delineate phases of the lake and obtain a continuous history that may be thousands of years long. Lake attributes can be obtained from a sediment core, but shorter-term historical information can include more detailed and numerous variables. In addition to information related to the lake, other data may be useful in assessing the lake ecosytem: local watershed land-use, airshed quality, sewage or septic systems, river inputs, and exotic and nuisance species.

When a sufficient data set has been assembled, goals specific to the type of ecosystem can be established (Palmer et al. 1985). Emphasis is often, and appropriately, placed on the importance of establishing monitoring goals, which will increase efficiency of funds and resources (Skalski and McKenzie 1982, Palmer et al. 1985, Perry et al. 1987, Dixon and Chiswell 1996). A monitoring program will be ineffective if it is not developed based on the decided goals or if its implementation varies from set guidelines. Using the collected background information, probable and potential anthropogenic impacts can be determined, and mode of monitoring can be developed accordingly.

Considering the funding resources available can be a real factor constraining a monitoring program. Ideally, availability of resources would not be the factor directing program design (Perry et al. 1987), but realistically, it should be incorporated. For this reason, a varied monitoring program that is based on the present availability of funds may encourage lake monitoring in organizations that might be lacking resources. During a low-funding period, fewer variables could be analyzed. Depending on the potential threats to the ecosystem, variables can be selected that would best reveal the continuing status of the lake. Either TSI (Carlson 1977) or phytoplankton communities (Sullivan and Carpenter 1982) can be used to determine trophic status. Water chemistry can be assessed using composite variables (Landwehr and Deininger 1976, Dinius 1987). Organism groups can be used to indicate the biological quality of the water (Patrick 1949, Sprules 1977, Sullivan and Carpenter 1982, Pace 1986, Metcalfe 1989, Reynoldson et al. 1989, Stemberger and Lazorchak 1994). A combination of variables would provide the most insight for the ecosystem, but selected variables can provide evidence when there are significant changes in lake quality. Assessments of organism community interactions can also be a useful monitoring tool (Karr 1981). Integrated monitoring programs can provide insight by incorporating a number of the biological and chemical factors of a lake ecosystem (Karr 1987).

With the available chemical and biological data for the lakes under consideration, decisions can be made for sampling locations and frequencies based on previous research and site characteristics. One possibility is sampling a single site repeatedly. The benefits of a fixed sampling location include its response to changes on a larger scale (Jassby 1998) and the known consistency among data collections (Ward et al. 1986). Another option is the use of a reference site, where the study lake is similarly sampled, but its biological and chemical data are compared to a least-impacted lake located in the same region (Heiskary and Wilson 1989). A third design, control treatment pairing, is typically used when it is suspected that some factor is impacting the lake ecosystem (Skalski and McKenzie 1982). Additionally, multiple sites can be sampled in an effort to characterize as much of the lake as possible (Skalski 1990). Design should be carefully considered and based on goals determined by resources managers.

The variables to be analyzed in a specific lake will be based on the issues of concern. When nutrient input and eutrophication are a concern, nutrients and chlorophyll *a* can be analyzed. If the lake under consideration is an important component for fisheries, fish community indices can be used. Biological indices, which are based on a variety of parameters, can range from a simple TSI to a complex community or multivariate index. If the ecosystem threat is unknown, biological analysis, using indicator species or community biotic integrity, can be used as an integrative tool.

National Park Monitoring

In response to laws and management regulations in the National Park Service regarding resources inventory and monitoring, several parks have adopted aquatic monitoring programs. The National Parks Omnibus Management Act of 1998 encouraged the development of monitoring programs in order to protect natural resources:

> The Secretary [of the Department of the Interior] shall undertake a program of inventory and monitoring of National Park System resources to establish baseline information and to provide information on the long-term trends in the condition of National Park System resources... (1998).

Furthermore, the tenets in the National Park Service's plans for managing resources presents a need for monitoring and inventorying these natural systems. With the data provided in a baseline collection, important management decisions can be made:

> The National Park Service will assemble baseline inventory data describing the natural resources under its stewardship and will monitor those resources at regular intervals to detect or predict changes. The resulting information will be analyzed to detect changes that may require intervention and to provide reference points for comparison with other, more altered environments (National Park Service 1988).

Some parks face very specific and direct threats, and others may experience water quality degradation very slowly. Monitoring programs have been established at some locations for several years (including INDU, SLBE, and VOYA), but consistency within and among parks is lacking. With a consistent monitoring protocol throughout a particular region, information and assistance could be shared among resource managers at numerous parks. This project was begun in an effort to determine the extent and impact of these threats on the Great Lakes Cluster National Parks and to establish consistent monitoring protocols. The accomplishment of these goals will provide a means of protecting these valuable natural resources from destruction related to anthropogenic impact.

Around the Great Lakes, the threats to inland lakes at National Park units are wide-ranging. Located at the southern end of Lake Michigan and within the Chicago metropolitan area, Indiana Dunes faces multiple impacts where fragments of natural land are interspersed with heavy industry and residential development. At Sleeping Bear Dunes, the direct impacts also include development and the additional threats of agricultural drainage and highway runoff. On Lake Superior, Pictured Rocks is further removed from highly populated areas, yet water quality is threatened by residential development and building within the lakes' watersheds. Voyageurs

National Park, though not directly adjacent to the Great Lakes, is primarily a park of lakes and rivers. Water quality problems in Voyageurs are complex and include artificial lake level control, dikes, dams, levees, boat traffic, and development within lake watersheds. Finally, Isle Royale lies in Lake Superior and is a designated wilderness area; despite this designation, there is a threat of aerial deposition of contaminants from industrial complexes across the lake. All of these parks face the additional impacts of non-native species invasion, increased visitor use, overfishing, increased boat activity, and increased aerial deposition. Because of these numerous threats and the potential for them to worsen, water quality monitoring would play an essential role in highlighting and gauging the rate of change in these lake ecosystems.

The background information available for these lakes is scattered and limited, so an extensive database would be a useful addition to that information. The goal of this project was to describe the current state of the water resources at these parks and to collect continuous baseline information. Then, using this information, the best plan for monitoring these lakes would be determined.

Chapter 3 Park Literature Reviews

Indiana Dunes National Lakeshore

Figure 3-1. Map of Indiana Dunes National Lakeshore

In 1966, the Indiana Dunes National Lakeshore was established as part of the National Park system. For years, industrial and residential development in the area encroached on the rare dune ecosystem and threatened complete obliteration. Even today, development around the protected dunes impacts the various ecosystems of the park. The Indiana Dunes National Lakeshore extends from Michigan City to Gary, Indiana, but it is far from a continuous swath of land. Heavy industry, residential communities, and municipalities are scattered along the east-west length of the park, forcing the park's boundaries to bend around them. With the close proximity of developed areas, water resources in the park experience contamination from aerial inputs, runoff, and sewage systems. Many of the wetlands have been drained or ditched in order to protect residences from flooding, thereby sacrificing their natural filtration qualities. Lakes in the park, including Long Lake, have been subject to disturbance by roads and runoff. Despite these ecosystem threats, limited research has been conducted on water resources in the park.

Long Lake has been subject to a recent history of severe degradation. At the beginning of the 1900's, railroads were built on the north and south sides of the lake. This increased the threat of contamination and added to the impacts the lake was experiencing from the highway south of the railroads. In the mid-1920's a major road was constructed, and it cut right through Long Lake, effectively dividing it into two parts. Eventually, the railroad on the north side was abandoned and was submerged by rising lake levels. Despite the presumed degradation of Long Lake, the amount of information available is very limited. There are brief mentions of the lake in historical documents and maps, but research is limited to the period after the area was incorporated into a national park.

When the Wisconsin Glacier retreated 14,000 years ago, Lake Michigan went through a series of lake fluctuations. The receding water exposed land areas marked by sand dunes, relict beaches, sandbars, and spits (Chrzastowski et al. 1994). Lake fluctuations combined with driving winds resulted in a series of dunes and swales that are still present. Ponds in the Miller Woods section of the park delineate separate dune ridges, and numerous interdunal ponds are present at West Beach, including Long Lake.

There are numerous and varied water resources located in the park. Among the aquatic ecosystems are wetlands, bogs, pannes, ponds, rivers and streams; all of which have been the subject of some research (Gilles and Lapham 1980, Wilcox 1982, Hardy 1984, Cole et al. 1990, Whitman et al. 1990, Jokinen 1994). Specifically, Cowles Bog and the Miller Woods ponds have been studied in some depth. Studies of historical aerial deposition have been conducted at Cowles Bog (Cole et al. 1990), and macrophytes within this wetland have been examined in relation to disturbance (Wilcox et al. 1985). An extensive survey at Miller Woods ponds of water chemistry, macrophytes, phytoplankton, and macroinvertebrates formed an important baseline for these numerous ponds (Whitman et al. 1990). Other research examined the macrophyte community as related to age of the ponds (Wilcox and Simonin 1987, Jackson et al. 1988), which was similar to the successional research conducted on interdunal ponds early in the century (Shelford 1911). Long Lake, the largest interdunal pond in the park, exhibits succession stages at a decreased rate in a larger system.

One of the earliest records of Long Lake ecology includes a list of fish species found in Long Lake (Brenan 1923). At that time, the lake was well used by recreational fishermen, and fish species included yellow perch, bass, and chain pickerel. A survey of the fish community was again conducted in 1988, and similar species were identified. Additional species present included green sunfish, bluegill, carp, black bullhead, yellow bullhead, golden shiner, and fathead minnow (Spacie 1988). The population has changed dramatically in recent years. Yellow perch have not been identified in Long Lake since a massive fish kill in 1990 that was likely the result of winter die-off.

In a study of water resources around the park, Jokinen (1994) also examined the mollusk community in Long Lake. Communities around the park were clearly distributed based on the age of the pond they inhabited, as was also found by Shelford (1913). According to the study, however, Long Lake's mollusk population was scarce in density and diversity.

In a study conducted to determine the extent of contamination at a dump site, Long Lake was used as a control site because it was presumed to have healthy biota (Steffeck 1990). Interestingly, when carp and snapping turtles were analyzed in tissue analysis, high levels of mercury were found. Elevated lead concentrations were also found in carp samples. The contaminant tissue analysis has not been repeated since these results were found.

A study of groundwater movement in the area indicated that Long Lake is a recharge lake with water leaving to the south and likely to the north (Isiorho et al. 1996). Other studies of groundwater movement have indicated that ditching on the south side of U.S. 12 may be draining water away from the lake (Dolak 1985).

The information about Long Lake is limited to specific communities and hydrology, so the need for a comprehensive review of the lake is clear. Baseline data on biological communities and water chemistry would be helpful in determining management activities and future goals. Evidence of contamination and low diversity among biological groups indicates the lake is in need of protection.

Sleeping Bear Dunes National Lakeshore

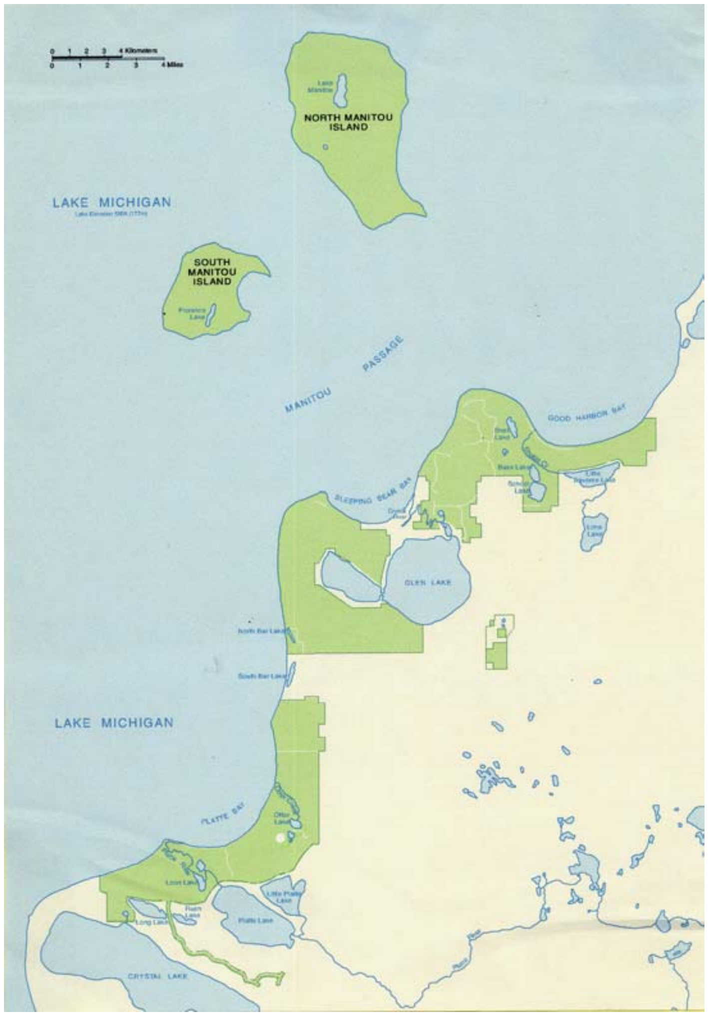


Figure 3-2. Map of Sleeping Bear Dunes National Lakeshore

Sleeping Bear Dunes was established as part of the national park system in 1970. Prior to its establishment, the area was heavily used for logging. By 1910, much of the land had been cleared, but it has since reforested. Since the establishment of the park, the area has been used for recreation, with the many inland lakes offering water-based activities. Because many of the lakes lie only partly within the Lakeshore, they are subject to anthropogenic effects that might not otherwise affect National Park waters. The Platte River flows directly through Loon Lake, and more than 90% of the watershed lies outside park boundaries. Glen Lake and Little Glen Lake have only limited borders with the park, so the area subject to outside influence is great. The anthropogenic influences are numerous and threatening because the park is not isolated, like Isle Royale, nor bordered by a buffer zone, like Pictured Rocks. The many lakes have invited much aquatic research since before the park's inception, so there is some information available. The majority of research in this section of Michigan has focused on the Platte Lake system and Crystal Lake, due to their size and extreme popularity among recreationists and residents. Research on the lakes in Sleeping Bear Dunes is scattered and discontinuous, however, so interpreting and condensing the information for management purposes is a sizeable, but necessary, task.

A geological study of the area describes the lakes' formation as a function of glacial scouring followed by falling lake levels and post-glacial uplift (Calver 1942). According to Handy and Stark (1984), after the last major glacial retreat, two minor advances and retreats covered the northwest portion of Michigan where Sleeping Bear Dunes is located. Glacial processes eroded the lowlands, which the larger lakes now occupy. Areas that were inundated with water were landlocked during the last glacial retreat, and these form many of the inland lakes in the park (Calver 1942). Paleolimnological studies have not been conducted on the Sleeping Bear Dunes lakes.

One of the earliest studies conducted on the lakes that eventually became part of the Sleeping Bear Dunes was an intensive survey of the Platte Lake system, including Loon Lake, in 1940 (Brown and Funk 1940). At this time, Loon Lake was referred to as Round Lake, and it was described as a river lake (Brown and Funk 1940). Maximum depth was defined as 21.6 m, and surface area was 38 hectares. The surrounding area was heavily wooded with a low, swampy shore. The lake showed distinct thermal and chemical stratification. Measurements taken in July of 1940 include a dissolved oxygen reading of 9.1 mg/L at the surface, with a temperature of 23^0C. The temperature fell to 10 ^{0}C at 21 m deep. Conditions were deemed appropriate for fish to survive below the thermocline (Brown and Funk 1940). Biological communities were also surveyed, and extensive fish and plant lists are provided. Fish collected included perch, northern pike, rock bass, long-eared sunfish, smallmouth bass, pumpkinseed sunfish, rainbow trout, and cisco.

Another widespread survey in 1950 was conducted on Little Glen and Big Glen Lakes (Rodeheffer and Day 1950). This was the earliest extensive survey on those lakes, but it did not expand to other lakes in the present-day Sleeping Bear Dunes. Aside from these State of Michigan surveys, not much research was performed on the Sleeping Bear Dunes lakes, until the land was incorporated into a National Park.

In an attempt to inform management about the condition and importance of the aquatic resources in Sleeping Bear Dunes and Pictured Rocks, Stockwell and Gannon collected baseline data on the lakes in 1975. Again, the work focused on the Platte River system, including Loon Lake, but they also studied Florence Lake, located on South Manitou Island. A maximum depth of 20.1 m was recorded, and surface area was measured as 38.4 hectares (Stockwell and Gannon 1975). The extensive chemical analysis included nutrients, dissolved oxygen, pH, alkalinity, conductivity, and chlorophyll *a*. Dissolved oxygen was near saturation through the water column, although there was some oxygen depletion in the hypolimnion. Stockwell and Gannon (1975) determined that Loon Lake probably functions as a nutrient sink in the system, and later, Boyle and Hoefs reached the same conclusion (1993). Moderately low nutrient concentrations indicated good water quality, and the rapid flushing rate of 12 days may contribute to this classification (Stockwell and Gannon 1975).

Around the same time, fish surveys on Loon Lake found only shorthead redhorse, pumpkinseed sunfish, and rock bass (Kelly and Price 1979). Collections at other lakes resulted in far longer lists: North Bar- alewife, sand shiner, spottail shiner, northern pike, smallmouth bass, yellow perch, and johnny darter; Round Lake- bluntnose minnow, golden shiner, blacknose shiner, sand shiner, white sucker, banded killifish, largemouth bass, yellow perch, and johnny darter (Kelly and Price 1979). The low species diversity for these lakes, particularly Loon Lake, is reason for some skepticism about results accuracy.

An intense study of the aquatic macrophytes of Sleeping Bear Dunes and surrounding lakes in Benzie and Leelanau counties resulted in lists of floating, submerged, and emergent plants for each lake (Hazlett 1989). This valuable survey provides lists for each lake, including Loon, North Bar, and Round Lakes. Among these three, North Bar had the most floating and submerged species, and Round Lake had the most emergent and shoreline species. Loon Lake had far fewer of both types (Hazlett 1989).

Aquatic research conducted in 1984 highlights the geological history of the region, including lake origin (Handy and Stark 1984). North Bar is unique in that it is connected to Lake Michigan at times (Handy and Stark 1984). Lake level fluctuations in North Bar are common because of its close proximity to the large lake. Interestingly, South Bar Lake is situated identically, but the sand bar between it and Lake Michigan does not wash out. This provides an interesting pair of lakes for study. Measurements of specific conductance were taken in Loon and Round Lakes. Loon Lake had a range of 289-330 μmhos, and the one measurement in Round Lake was 300 μmhos at 25 ^{0}C. Inland lakes at Sleeping Bear Dunes are typically ice-covered from January to early April.

A far more extensive aquatic inventory was conducted in 1993 on 20 lakes in the park (Boyle and Hoefs 1993a). A similar study was conducted the year before, but only four lakes were included (Albert 1992). Water chemistry in the lakes was sampled once in summer for three years. Unlike the previous study by Handy and Stark (1984), the goal of the study was water quality. Loon Lake had a Secchi depth range of 2.6-5.0 m, and chlorophyll *a* was 0.55-1.85 μg/L. North Bar Lake appeared much more productive with a Secchi range of 2.0-2.8 m and chlorophyll *a* range of 1.4-3.26 μg/L. Round Lake had a Secchi range of 3.5-5.3 m and chlorophyll *a* range (for only one year) 1.69-3.96 μg/L. All three lakes showed distinct temperature stratification and clinograde curves. Nutrient data was also collected during this sampling period. In addition to the report, this study resulted in a manual for monitoring these aquatic resources (Boyle and Hoefs 1993b). The manual provides instruction for performing the sampling conducted in the inventory.

In 1994 and 1995, another baseline study on the Sleeping Bear Dunes lakes was conducted (Last et al. 1995). Twelve of the lakes were examined for water quality characteristics, including nutrients and ambient conditions. Again, there was oxygen depletion in the hypolimnion in Loon Lake (Last et al. 1995), as previously indicated (Stockwell and Gannon 1975). North Bar had the lowest transparency, the lowest surface pH, and the highest specific conductance among the lakes over the two years studied. North Bar also had the highest ammonia and nitrate concentrations. Loon Lake had among the lowest nitrate values. With these results and data collected from streams in the park (Whitman et al. 1995), a critique was made of the constancy of the lake monitoring at Sleeping Bear Dunes (Last and Whitman 1996); the need for a documented monitoring program that is maintained for data consistency was stressed.

Pictured Rocks National Lakeshore

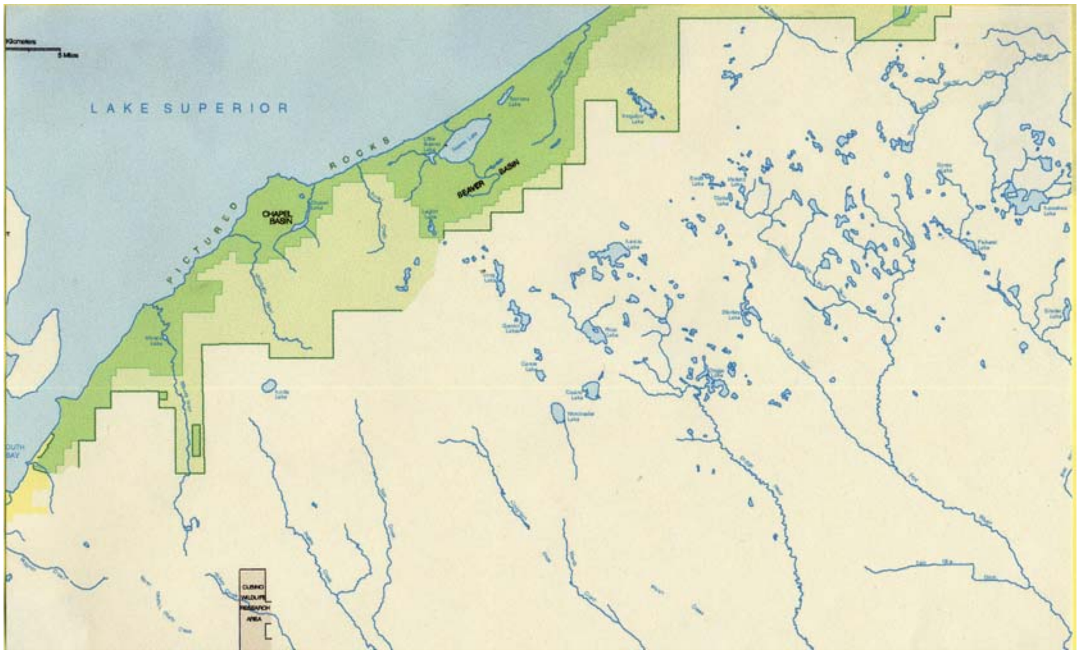


Figure 3-3. Map of Pictured Rocks National Lakeshore

Pictured Rocks National Lakeshore was established as a national park in 1966, and its isolation from highly populated areas and its proximity to major universities make it a common site for research. There are eight inland lakes within the park boundaries and numerous rivers and creeks; additional lakes are located in the inland buffer zone, which protects the waters that flow through the park to Lake Superior. Graduate students, the state of Michigan, and the National Park Service have all conducted limnological studies at Pictured Rocks.

A geological study of Beaver Lake conducted in 1996 was designed to discover the history of Beaver Lake sedimentation. Sediment cores were collected that dated to the time Beaver Lake was an embayment of Lake Superior (Fisher and Whitman 1998). The sediment history revealed that since it became a lake, the sediment has been composed primarily of sand. Coarse substrate is limited to one section on the southeast corner of the lake, and this appears to be the only location where this substrate ever existed (Fisher and Whitman 1998).

In 1972, personnel from Alger County along with the National Park Service studied the physical and biological characteristics of several lakes in the county, including Beaver and Grand Sable Lakes (Doepke 1972). Basin morphometry information was collected, and in August some chemical data were also collected. In Beaver Lake alkalinity was 70 mg/l as $CaCO_3$ and hardness was 75 mg/l as $CaCO_3$. In samples from Grand Sable Lake, alkalinity was 47 mg/l as $CaCO_3$ and hardness was 46 mg/l as $CaCO_3$. Physical parameters were collected in depth profiles, with measurements of temperature, dissolved oxygen, specific conductance, and pH. In 1970, nutrient analysis on all of the lakes in the park showed similar, low concentrations in Beaver and Grand Sable lakes of ammonia (<0.03, <0.03), nitrate (0.02, 0.02), and total phosphorus (0.016, 0.010) (Limnetics 1970). Another water investigation conducted between 1979 and 1981 again showed low concentrations of macronutrients in both lakes (Handy and Twenter 1985).

An extensive survey of four Pictured Rocks lakes in 1987 provided nutrient concentrations for Beaver and Grand Sable Lakes. Total Kjeldahl nitrogen was 0.29 mg/l at Beaver and 0.39 mg/l at Grand Sable, and total phosphorus was 0.012 mg/l at Beaver and 0.011 mg/l at Grand Sable (Kamke 1987). Beaver Lake had low concentrations of most nutrients, but its zooplankton community had the highest number of species among the lakes studied. According to Kamke (1987), this may be the result of the fertile watershed that the lake drains. Dissolved oxygen, however, was high through the water column in both lakes. Beaver Lake is a polymictic lake, so mixing throughout the year oxygenates the water column. Grand Sable is a dimictic lake, but the water column is well-oxygenated with clear water.

Doepke (1972) called Beaver Lake a moderately fertile lake and stated that Grand Sable was nearing the peak of its natural productivity. Kamke (1987) described Beaver Lake as early mesotrophic and Grand Sable as late oligotrophic. Limnetics (1970) stated that several of the lakes in the park were in states of advanced eutrophication. These varied conclusions regarding the lakes' conditions deserve some attention and correlation.

Fish species collected from Beaver Lake in 1995 include rock bass, white sucker, johnny darter, pumpkinseed sunfish, golden shiner, spottail shiner, yellow perch, log perch, and bluntnose minnow (Gerovac and Whitman 1995). This study was not inclusive of all species present in the lake, but it does provide information on relative abundance. A survey of Grand Sable during the same period resulted in a species list including white sucker, mottled sculpin, Iowa darter, smallmouth bass, spottail shiner, yellow perch, log perch, and bluntnose minnow (Gerovac and Whitman 1995).

The Michigan Department of Natural Resources and Department of Environmental Quality have studied sediment contamination in the inland lakes of Pictured Rocks extensively. Due to its location and plant community composition, Pictured Rocks has been identified as a medium-low risk for air pollution (Bennett and Bannerjee 1995). In 1991, the Michigan DNR examined contamination in inland lakes across the state, including Grand Sable Lake (Evans et al. 1991). Sediment cores were collected, sectioned, and analyzed for 19 contaminants at each depth. Additionally, benthic macroinvertebrates and fish and bird tissues were examined. Between 1981 and 1995, lake trout and northern pike from Grand Sable were examined several times for tissue contamination (MIDEQ 1981, 1991, 1993, 1995). Contaminants including mercury, PCBs, toxaphene and several others were measured in ~12 fish each sampling period.

Among the lakes in Pictured Rocks, Grand Sable Lake has by far the most diverse population of mussels (Nichols 2000). Within this community, however, is the lowest abundance of individuals (1.7/100 m^2) (Nichols 2000). It has been hypothesized that the introduction of lake trout to Grand Sable Lake led to the decline in numbers of mussels present (Nichols 2000). Beaver Lake has a stable, reproducing population (Nichols 2000).

One of the primary recreational activities at Pictured Rocks is fishing, so maintaining water quality should be one of the park's top priorities. Some work has been conducted by the State of Michigan, but continuous data are lacking. Future research on these lakes should involve some trend analysis.

Isle Royale National Park

Figure 3-4. Map of Isle Royale National Park

Isle Royale National Park has been a research destination for geologists, biologists, and natural resource managers. Its remote location and lack of anthropogenic impacts makes it a unique, unimpacted ecosystem for natural interaction studies and a reference site for comparative studies. The island became a national park in 1931, and in 1976, 98% of the island was designated as federal wilderness. Since the early 20th century, research has been conducted on the island, but little is known about the ecology of the many inland lakes scattered about the island.

Glacial scouring formed the basins of most of the inland lakes. When the glaciers receded and Lake Superior levels dropped, the inland lakes surfaced. Paleolimnological studies in Lily Lake, located on the western end of the island, have revealed periodic advance and retreat of the glaciers over the island (Winkler and Sanford 1997), and a continuous history from ~9500 yr. before present has been determined.

Paleoecology has revealed significant differences between Isle Royale and the mainland in the timing and magnitude of changes in vegetation and aquatic environments (Flakne 1997). The Isle Royale aquatic environment went through waves of changes as lakes shallowed and deepened again over thousands of years (Winkler and Sanford 1997), but it was altered dramatically in the late 19th century over the course of a few decades when Europeans began settling on the island (Flakne 1997). A paleoecologic study conducted on Lily Lake, Lake Ojibway, and Wallace Lake revealed an increase in pollens not naturally found on Isle Royale (Flakne 1997). With an increase in visitors, pollen and plants from the mainland had an easy means of transport to this isolated island.

With the arrival of Europeans, research on this pristine island not subjected to the destruction associated with industry and settlement became common. Studies on mosses (Cooper 1912), lichens (Hedrick and Lowe 1936), and fungi (Povah 1935), and catalogs of vegetation (Wheeler 1909, Cooper 1914, McAtee 1921) brought many researchers to the island. Geological studies of the island's history were common. Some of the first studies conducted on the inland lakes were general ecological studies by Adams (1909), which included descriptions of Siskiwit Lake, Sumner Lake, and Rock Harbor. Descriptions of a naturalist, the document provides a picture of the natural landscape of Isle Royale in the early 20th century. Adams mentions tamarack swamps surrounding most of the smaller lakes and an extended water lily zone that prevents wave action on the lakes. The lake bottoms are described jointly as Acovered with peaty mud or with slime ((Adams 1909) p. 63). Bottom fauna listed are limited to mussels and snails and a few amphipods. Adult dragonflies were among the aerial fauna in addition to the butterfly *Argynnis atlantis* Edw. Siskiwit Lake was briefly described in association with Lake Superior because the two, according to Adams, share essentially the same conditions. In 1929, Koelz described 38 of the inland lakes in minute detail, including Sargent and Siskiwit.

Surveys of the phytoplankton in the early 1930's were the first in-depth aquatic research performed on the inland lakes (Taylor 1935, Prescott 1936, 1937, 1939, 1940). Taylor collected samples at sites in Wallace and Sargent Lakes, in addition to sites in Lake Superior and McCargoe Cove. Two samples were collected from Sargent Lake, and Taylor described the lake bottom as "thin mud over loose rock" (Taylor 1935). There was no floating vegetation mat present, and the water was clear and brown. An extensive species list, including diagrams, is provided, with lake and site locations. Prescott's work (1936, 1937, 1939, 1940) examined the desmids of Isle Royale in an extensive, four-paper series compiling the data from a collecting trip by the University of Michigan in 1930. Little background information is provided about the lakes except that Isle Royale's topography creates lakes with habitat suitable for a rich algal flora (Prescott 1936). The second paper in the series is devoted to the genus *Cosmarium* (Prescott 1937), and the third paper examines the genera *Staurastrum*, *Micrasterias*, *Xanthidium*, and *Euastrum* (Prescott 1939). Neither collection sites nor techniques are provided in the papers, but from the list of species and collection locations, it can be determined that inland lakes sampled include Moose Lake, Hidden Lake, Lake Richie, Wallace Lake, Sargent Lake, Forbes Lake, and Siskiwit Lake (Prescott 1936, 1937, 1939, 1940). Additional sites were also sampled around the island: Tobin Harbor, Passage Island, Gull Island, Bat Island, Scoville Point, and a "small pool in sphagnum swamp" (Prescott 1936).

In 1929, Koelz conducted one of the first fishery studies at Isle Royale. He surveyed 39 lakes and provided descriptions of the lakes in addition to discussions of the fishes (Koelz 1932). In 1959 originally, Lagler and Goldman wrote a fishing guide that listed 32 lakes suitable for fishing; in 1982, they updated the book. They describe 42 species from 29 genera common to Isle Royale (Lagler and Goldman 1982), three of which are not members of the primitive fish fauna: rainbow trout, smelt, and sea lamprey (Sharp 1960). Lagler and Goldman (1982) divide the lakes into two categories: those that contain members of the whitefish group and those that do not. Lakes among the former are large, deep lakes (Desor, Siskiwit), and shallower, boggier lakes are among the latter (Wallace, Sumner, Mud, Stickleback, Lily, Ahmeek, and Sholtz). Because Isle Royale is used for recreation, fishing pressure has affected some of the inland lakes (Sharp 1960). Commercial fishing has not been permitted since 1962, but approximately 1/3 of park visitors fish in the accessible waters of the park (Wallis 1966).

In 1960, Sharp and Nord conducted a survey of water chemistry and fish communities. In addition to the overall report and recommendations, appendices describing specific inland lakes are presented at the end of the report. In Siskiwit Lake, fish collected in gill nets included lake trout, whitefish, yellow perch, burbot, and common sucker, and creel survey results included lake trout, pike, brook trout, whitefish, yellow perch, and rainbow trout (Sharp 1960). Koelz (1929) had previously reported chub, herring, trout perch, nine-spined stickleback, stripenose, spottailed shiner, lake shiner, and two sculpin species. Sargent Lake fish collections included northern pike, white sucker, yellow perch, herring, and spottail shiner. The poor condition of

Sargent Lake fishes, small and slim with a "black spot" parasite, was most likely a result of overcrowding, since lake access is difficult and fishing pressure is minimal (Sharp 1960).

Sharp and Nord's research was the first examination of water chemistry in these lakes. A lengthy description of Siskiwit Lake provides depth profiles of temperature and oxygen. The study was conducted in August of 1960, when maximum surface temperature was 68 ^{0}F (20 ^{0}C), and temperature at 125' (41 m) was 42 ^{0}F (5.5 ^{0}C). Dissolved oxygen at 125' was 13.1 mg/L, and no surface reading was taken. Depth profiles were also taken in Sargent Lake to 39' (12.8 m). Surface temperature was 74 ^{0}F (23.3 ^{0}C) and bottom temperature was 57 ^{0}F (13.9 ^{0}C). Dissolved oxygen was "adequate" at the surface, 6 ppm at 21' (6.9 m), and 1 ppm at 39' (12.8 m). Total alkalinity was 34.2 ppm. They described the water in Sargent Lake as slightly discolored, and they recorded a Secchi depth of 4' (1.3 m).

In 1978, Swain, in response to reports of elevated pesticide concentrations in western Lake Superior, measured chlorinated organic residues in water, fish, and precipitation at Isle Royale (Swain 1978). Siskiwit Lake was originally selected as a control site for testing PCB concentrations in Lake Superior, and the results showed significantly higher concentrations of several organic residues in Siskiwit fish than in the Lake Superior fish (Swain 1978). The results were astounding, and numerous studies followed this discovery. In 1984, heavy metals were discovered in the sediments of Siskiwit Lake (Czuczwa et al. 1984). The dioxins and furans could only have reached the island by atmospheric transport since Isle Royale is so isolated; and because Siskiwit Lake is 17 m above the Lake Superior water level, the contaminants could not have been transported by water (Czuczwa et al. 1984). Atmospheric transport has been cited as the reason for high concentrations of atrazine, which has been deposited in the Isle Royale lakes in precipitation (Cromwell 2000). Fish toxicity tests revealed toxaphene concentrations higher in Siskiwit Lake lake trout than in lake sediments (Gooch and Matsumura 1987). Further research revealed that contaminants in the fish were significantly more toxic than the compound originally used as an agricultural pesticide (Gooch et al. 1990). Toxicity tests were conducted on snow samples from Isle Royale in order to determine aerial input to the island, and concentrations of PCBs were five times higher in Isle Royale precipitation than samples from the Duluth/Superior metropolitan area (Swain 1978). Precipitation is also a potential source of high concentrations of ammonia that have been found in Isle Royale streams (Stottlemeyer and Toczydlowski 1997). A study of national parks' vulnerability to air pollution, however, ranked Isle Royale at low risk (Bennett and Bannerjee 1995).

Elevated concentrations of contaminants have also been found in soil located within the watersheds of some Isle Royale lakes. Mercury has been found in the Sargent Lake watershed in concentrations as high as 10-370 ppb (Woodruff 2000). Although the fish in Sargent Lake also have elevated concentrations of mercury, only two cases have been identified when the two fluctuate similarly (Woodruff 2000), meaning there is no basis for cause and effect.

The most extensive study for baseline biological and chemical information on the lakes of Isle Royale was conducted in 1980 (Toczydlowski et al. 1980). Samples were collected at 13 of the inland lakes and along 8 streams. Siskiwit Lake was among the lakes studied and described. June profiles in Siskiwit were taken at the east and west ends where depths were 40 m and 20 m. In addition to temperature and DO profiles, alkalinity, pH, PO_4, Si, and NO_3-N were taken at 6 depths. Secchi depth was 7.6 m. Among the lakes, the phytoplankton communities were distinct. Small flagellates were most numerous in all lakes, and in Siskiwit, they composed 67% of the phytoplankton community. The dominant species in Siskiwit were among those described by Hutchinson as indicative of cold, oligotrophic waters (Hutchinson 1967); of all the lakes sampled, Siskiwit had the lowest nutrient concentrations.

A great deal of research has been conducted on Isle Royale, but the dataset is periodic and incomplete. A collective dataset on these lakes would help to monitor changes that might result from recreation activites or from aerial contamination. This island has been a valuable study site because of its isolation, and if anthropogenic input is affecting these lakes, some effort at prevention should be made.

Voyageurs National Park

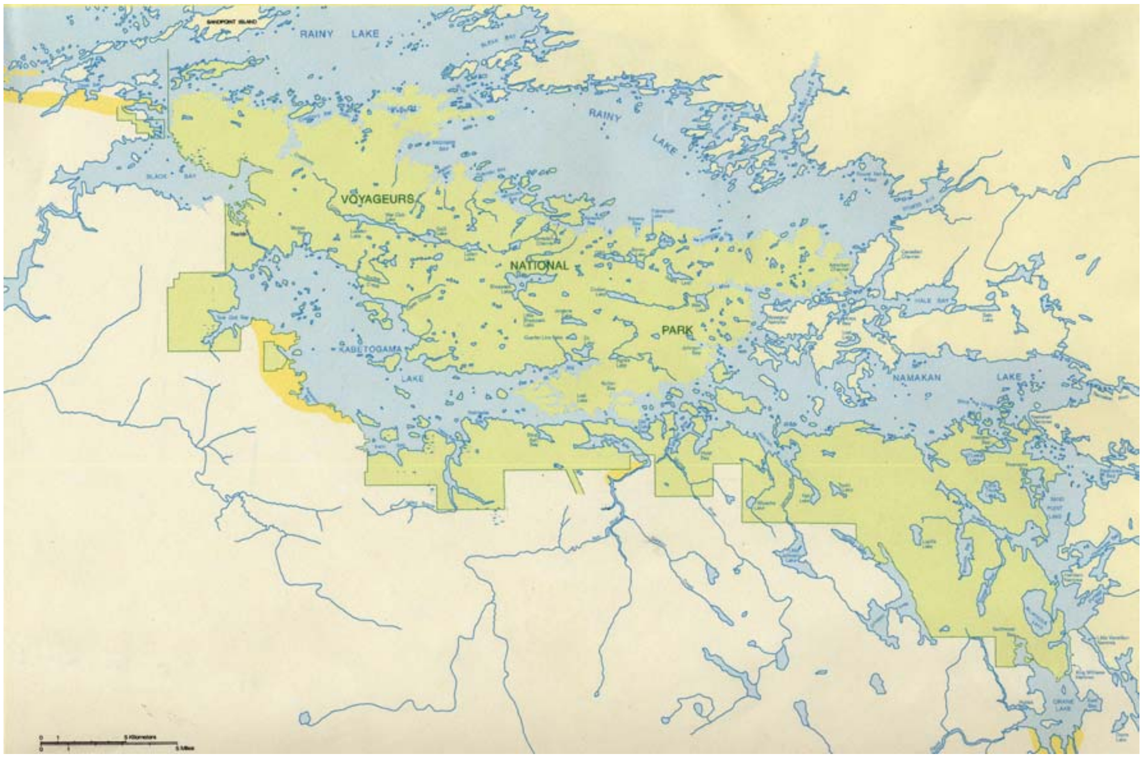


Figure 3-5. Map of Voyageurs National Park

Voyageurs National Park was established in 1975 to preserve part of the water trail traveled by voyageurs in the 19th century; most of the park is composed of lakes and rivers. Three of the largest lakes in the park lie both in Canada and the United States, and because they are used as a reservoir, water level control is a complicated issue between governments. Other complications with such an arrangement include governing fisheries and water quality. Inland lakes are numerous in Voyageurs National Park, and their maintenance is greatly important because they are highly used fishing and recreation areas.

The majority of research conducted on the Voyageurs lakes has been related to the fisheries resource. Some paleolimnological research conducted by Winkler and Sanford (1997) provided some important information about the geologic history of the park and its lakes. Sediment cores from Cayou Lake (located ~1 mile south of Quill Lake) were dated to 10,000 years before present. The Voyageurs lakes were formed during deglaciation after the Canadian shield bedrock that underlies the area was modified by the Laurentide ice sheet (Winkler and Sanford 1997). Cayou Lake has a maximum depth of approximately 12 meters, and it now lies in a chain of paternoster lakes in the forest. Sediment cores reveal the diatomaceous and pollen history of the lake. The earliest records show an environment similar to an alkaline arctic climate, around 10,000 years before present. During the period 8000-4000 years before present, the climate was considerably warmer, and from 4000 years before present to just before European settlement, the climate was wetter and cooler. Drastic changes occurred in the lake's diatom composition with the arrival of European settlers; natural fire frequency decreased, and individual peaks in charcoal can be attributed to known fires. In the lakes, blue-green and green algae increased with an obvious increase in nutrient input. Further, soot particles from industrial emissions were far more abundant (Winkler and Sanford 1997). Large-scale, slow changes were associated with glaciation and deglaciation, but the Voyageurs lakes have changed at a more rapid rate in the past 110 years than in the rest of the history of Cayou Lake.

In studying sediment cores from 80 lakes in northern Minnesota, Sorenson et al. (1990) discovered that mercury concentration in sediment had doubled since 1880. Mercury concentrations in precipitation, water, sediment, and plankton were examined in these Minnesota lakes, including the large reservoir lakes and Mukooda Lake in Voyageurs National Park. Direct correlations were found between mercury concentration in water, zooplankton, and fish (Sorensen et al. 1990). It was also determined that the source of mercury was atmospheric, and mercury from geologic and point sources was originally atmospheric (Sorensen et al. 1990). Because of its remote location, it has been classified as low risk for air pollution (Bennett and Bannerjee 1995). With the knowledge of mercury contamination in this otherwise pristine northern park, the extent of other atmospheric contamination is an important consideration.

Current water quality of Voyageurs lakes has been the subject of some research. The large lakes that make up the reservoir system, Rainy, Namakan, Sand Point, and Kabetogama Lakes, have been heavily studied because of the many anthropogenic impacts associated with regulated water levels, boat traffic, and fishing pressure. Less is known about the smaller, inland lakes, which are subject to many similar anthropogenic impacts. The large lakes range from oligotrophic to mesotrophic with low dissolved solids and alkalinity, and the interior lakes typically have lower dissolved solids and alkalinity (Webster et al. 1993). Among the four large lakes, Kabetogama was statistically different from the others based on trophic state indicators (Kepner and Stottlemeyer 1988). Kabetogama could be described as eutrophic due to the input from an area of calcareous glacial drift (Kallemeyn et al. unpublished). Certain interior lakes were more similar to the large lakes, e.g. Mukooda Lake, according to Payne (1991).

Interior lakes, including Locator and Mukooda, have been surveyed for water quality, but they have not been the subject of much biological research. Locator Lake lies within the chain of lakes on the Kabetogama peninsula that also includes War Club, Quill, and Loiten Lakes. It has been classified as extremely sensitive to acidic precipitation because of its low alkalinity (<5 mg/L as $CaCO_3$) (Payne 1991). Webster et al. (1993) also determined that Locator Lake has a low acid neutralizing capacity. Mukooda Lake has a much higher alkalinity (>20 mg/L as $CaCO_3$), so it is classified as nonsensitive (Payne 1991). Dissolved oxygen in Mukooda Lake over the summer ranged from 10.5 mg/L in May to 3.0 mg/L in August; in Locator Lake, dissolved oxygen concentration was 3.5 mg/L in May and 0.8 mg/L in August (Payne 1991). Between 1983 and 1989, Locator Lake had a mean pH of 6.69 and a mean conductivity of 26.8 mg/L (Webster et al. 1993). Other measurements collected and averaged over this period include SO_4=69 mg/L, Cl=12 mg/L, NO_3=2 mg/L, Ca=126 mg/L, Mg=87 mg/L, K=16 mg/L, Na=43 mg/L, and DOC=10 mg/L (Webster et al. 1993). Biological research conducted on Mukooda Lake by Payne (1991) revealed that primary productivity may be limited by low phosphorus availability (Payne 1991). Seasonal declines in productivity could be the result of limited trace nutrient availability or grazing (Payne 1991), but the responsible factors are unknown due to the small amount of biological research conducted on these lakes.

Because the majority of Voyageurs National Park is water, most recreation and visitation is dependent on water activities. Much of this significant resource is shared between the United States and Canada, so laws governing its use are complex. Water quality has recently become an issue because there is so much anthropogenic input and activity associated with the reservoir lakes. Nutrient availability in the system is directly affected by the amount of drawdown during winter (Kepner and Stottlemeyer 1988). Subsequently, algal and macrophytic standing crop varied with nutrient availability. Macrophyte community composition was thus dually affected by nutrients and actual water level (Wilcox and Meeker 1992). Benthic communities in Namakan Lake were also negatively affected by winter drawdown; community diversity was highest in the year with the least drawdown (Kraft 1988), perhaps due to the limited habitat in the macrophytes (Wilcox and Meeker 1992). Water level regulations also affected interactions in the fish community (Cohen and Radomski 1993). The water level necessary for emergent vegetation—the preferred spawning habitat of walleye and northern pike—was attained only incidentally during the period studied by Kallemeyn and Cole (1990). When this water level was not actualized due to regulating actions, fish populations for the year decreased (Kallemeyn and Cole 1990). The ecological interactions in these lakes are such that water level regulations impact the whole system.

Assessing and monitoring the quality of all the water bodies in Voyageurs is a huge undertaking. The heavy boat traffic and lake use warrant some consideration, however. This popular park is used for much water recreation, so the lakes should be a top priority.

Chapter 4 Lake Descriptions

Long Lake, Indiana Dunes National Lakeshore

Long Lake is one of the only inland lakes located in the Indiana Dunes National Lakeshore. The lake developed as a result of wind effects on the dune ecosystem. Winds that create the dunes also move sand out of the swale areas, and when these blowouts go below the groundwater surface, interdunal lakes and ponds are formed.

The watershed of Long Lake is limited in size, but the entire national park lies in a heavy residential and industrial area. The park extends along the Lake Michigan shore of Indiana for approximately 21 miles. Steel mills are spaced along the entire Indiana shoreline, and residences and municipalities are abundant due to the region's proximity to Chicago. Lake Michigan lies less than 0.5 mile north of Long Lake, across the dunes. Heavily used roads are located on the south and west side of the lake, and surface drainage from these roads flows into the lake. The road along the west side of Long Lake was cut through the lake when it was built, so the lake that is now present is only a fraction of its original size; the other half has quickly transformed into wetland. The natural habitat is a dune and swale ecosystem. To the south and east of the region there is agricultural land, and to the west there is prairie land.

Long Lake is encircled by successional vegetation including sand reeds and cottonwoods. Aquatic vegetation in the marshy areas immediately surrounding the lake is primarily cattails. Shoreline length equals 4622 meters (Simon et al. 1997). The shoreline is relatively even, but changes in water level and infilling have led to the formation of embayments. The lake also has three small islands, and the shoreline development factor equals 2.5 (Simon et al. 1997).

Total surface area equals approximately 34 hectares. At its greatest width, Long Lake measures 378 meters (Simon et al. 1997), and during high winds, the lake's shallow waters are often completely mixed. According to Simon et al. (1997), maximum depth of the lake is 1.8 meters, and the mean depth is 1.1 meters. Deep areas can be found at different locations around the lake, but a deeper basin is located in the west end, near to the largest island.

The water in Long Lake is not very clear, and it often has a heavy brown appearance. Early in the spring, the Secchi disk can be seen to the bottom of the lake, but by autumn, the water is very brown. Mean Secchi disk depth for the months of May through September in 1997 was 1 meter, and the range was 0.77-1.4 meters. In 1998, the mean was 0.85 meters, and the shallowest Secchi depth recorded was only 0.5 meters, in August. The lake does not stratify, due to its shallow depth, and the surface temperature for April-September 1997 ranged from 12.6 to 27.4 ^{0}C, with a mean of 21.9 ^{0}C. Mean dissolved oxygen for the same period was 5.87 mg/L, with a range of 2.74-10.12 mg/L.

Sediment in Long Lake is composed primarily of organic material. There is sand under the layers of organics, but the surface sediments are almost solely degrading plants and other organic material (Simon et al. 1997). Submerged vegetation grows abundantly on the sediment, and massive root systems of the floating vegetation also grow on the sediment surface.

Aquatic vegetation includes floating vegetation, *Nymphea* and *Nuphar*, *Potamogeton*, and pickerel weed (Simon et al. 1997). Submerged vegetation is primarily *Chara* sp. By late summer, the lake surface is almost completely covered by *Nymphea* sp. and *Nuphar* sp., and the water column is full of *Chara* sp. By the end of summer, the plants begin to deteriorate, and dead plants are abundant on the lake.

Among the fish species present in Long Lake are *Lepomis cyanellus* (green sunfish), *Lepomis macrochirus* (bluegill), *Ictalurus melas* (black bullhead), *Notemigonus crysoleucas* (golden shiner), *Esox americanus vermiculatus* (grass pickerel), and *Cyprinus carpio* (carp) (Spacie 1988).

Figure 4-1. Photograph of Long Lake.

Loon Lake, Sleeping Bear Dunes National Lakeshore

Loon Lake is located in Sleeping Bear Dunes National Lakeshore along a string of lakes connected by the Platte River. The Platte River begins as an outflow of Little Platte Lake and flows through Platte Lake and Loon Lake before emptying into Lake Michigan. Also located along this path is Mud Lake, which feeds into the river.

The land around the lake is relatively flat although the region in which Loon Lake lies is quite hilly. There are many standing dead trees along the south shore, and lower foliage is prevalent near the Platte River entrance. Around much of the lake, wetlands lie outside of or are interspersed with the wooded areas. There are no islands or significant outcroppings around the lake, and the shoreline development factor equals 1.32 (Stockwell and Gannon 1975).

Loon Lake is primarily used for recreation, and residences and agricultural areas lie within the watershed. Further upstream from Loon Lake on the Platte River, there is a fish hatchery that adds significant amounts of phosphorus to the river and subsequently to Loon Lake. Only a few residences are located directly on the lake, but a boat ramp allows for heavy boating and fishing. Residential development around Platte Lake, upstream, may have a negative impact on the Loon Lake system. Neither Little Platte Lake nor Platte Lake is located within the boundary of Sleeping Bear Dunes National Lakeshore.

The Loon Lake basin has a volume of $8.5 \times 10^7\ m^3$, and the mean depth is 9 m (Stockwell and Gannon 1975). The deepest part of the lake is located in the southern section near the boat ramp and parking lot where the depth is 21 meters. There is a steep drop-off from the litoral zone to the limnetic zone along the south and west sides of the lake. Surface area of the lake equals 38.4 m^2, and the fetch is 494 m (Stockwell and Gannon 1975). The path of the Platte River through Loon Lake does not freeze in winter with the rest of the lake. The river enters Loon Lake on the east side and exits to the northwest.

Lake water is quite clear, and this may be a result of a rapid flushing time. Whereas most lakes have a flushing time measured in terms of years, the retention time for Loon Lake was recorded in 1974 as 12 days. Mean Secchi depth for the lake is 4 m. The lake is dimictic, and surface water temperature from May to October ranges from 11.4 to 23 ^{0}C, with a mean of 18.3 ^{0}C (Last et al. 1995). The surface dissolved oxygen for the same period had a range of 7.3-12.8 mg/L (Last et al. 1995). Sediment in the littoral zone is composed of very fine silt and sand, and the limnetic zone is primarily fine sediment.

Figure 4-2. Photograph of Loon Lake.

North Bar Lake, Sleeping Bear Dunes National Lakeshore

North Bar Lake is located near the town of Empire, Michigan, and it lies within the Sleeping Bear Dunes National Lakeshore. Formerly an embayment of Lake Michigan, North Bar Lake is now separated from Lake Michigan by a narrow sandbar. Occasionally the sand bar is opened by wave action or humans, and this allows the mixing of the two lakes' waters.

The lake and the surrounding area are heavily used for recreation, in part due to its proximity to Lake Michigan. The closeness of the town of Empire and the direct road access encourage visiting by swimmers, hikers, fishermen, and boaters. Some houses are located close to the lake and within the area, but the primary human impact is recreation. Sand dunes border the lake on the west and north sides, and forested areas are located on the east and south sides of the lake. The only connecting body of water is Lake Michigan, and that is a periodic occurrence.

The shoreline of North Bar Lake is relatively smooth with few embayments and no islands. There is an incline from the lake surface up the dunes, but there is little rise in the land along the rest of the lake's border.

North Bar Lake is used for many water recreation activities. The lake basin is relatively deep with a maximum depth of 9.8 meters and a mean depth of 3.1 meters. During high winds, the sand bar is often washed out, and Lake Michigan waters mix with North Bar waters.

Relatively clear water in the lake results in a mean summer Secchi depth of 3.4 m. Surface water temperature during the summer months ranges between 10.94 and 23.84 ^{0}C, with a mean of 18.12 ^{0}C (Last et al. 1995). Surface dissolved oxygen during the summer ranges between 8.49 and 12.57 mg/L (Last et al. 1995). Sediment in North Bar Lake is primarily sand, but the limnetic zone has quite a bit of marl.

Aquatic vegetation is abundant in the shallow areas, with large areas of *Potomogeton* sp. Abundant fish species include yellow perch and alewives (Kelly and Price 1979).

Figure 4-3. Photograph of North Bar Lake.

Round Lake, Sleeping Bear Dunes National Lakeshore

Round Lake is one of the smaller inland lakes at Sleeping Bear Dunes National Lakeshore. This lake is the furthest south in the park near the very large Crystal Lake.

Round Lake is a tributary to Crystal Lake and lies within the Betsie River system. Although the area around Round Lake is not developed, nearby Crystal Lake is surrounded by residences and municipal development. A small creek connects the lake to Crystal Lake, through a heavy marsh area.

There are no islands or significant embayments on the aptly named Round Lake. The shoreline is mostly even, and there is no incline of the land surrounding the lake. Vegetation around the south side of the lake is marsh habitat. Around the rest of the lake, submergent vegetation along the shoreline grades into wooded area.

Because of its small size, recreationists do not use Round Lake often. The lake is accessed from a small drive leading off the road, but there is not a boat ramp or dock on the lake. The lake has a maximum depth of 8 m, and this site is located near the center of the lake. The basin gradually deepens from shore to center, and there are no steep drop-offs from the littoral to the limnetic zone. Surface area of the lake is only 6 hectares. Because of its small size, strong winds probably cause some mixing. During heavy winds from the south, however, it is possible that waters from Crystal Lake are forced upstream into Round Lake.

The water is very clear, and the Secchi depth between April and September 1998 had a mean of 3.2 m with a range of 2.6-3.8 m. The lake stratifies, and it can be described as dimictic. During the winter months, the lake freezes. Surface temperature for April through September 1998 ranged from 9.19 to 25.9 ^{0}C. Surface dissolved oxygen for the same period had a range of 8.24-10.67 mg/L.

Fish in the lake include bluntnose minnow, golden shiner, blacknose shiner, sand shiner, white sucker, banded killifish, largemouth bass, yellow perch, and johnny darter (Kelly and Price 1979).

Figure 4-4. Photograph of Round Lake.

Beaver Lake, Pictured Rocks National Lakeshore

Beaver Lake is located in the central section of Pictured Rocks National Lakeshore. Extending 40 miles along the southern shore of Lake Superior, Pictured Rocks was established as part of the National Park Service in 1966.

Beaver Lake was once an embayment of Lake Superior, but during glacial retreat and subsequent isostatic rebound, it became separated from the large water body (Flint 1957). The soil surrounding the lake is primarily outwash and beach terrace. Beaver Creek still connects the two water bodies, flowing approximately 0.5 mile from Beaver Lake to Lake Superior. Lowery Creek flows into Beaver Lake from the southeast, and Little Beaver Lake is separated from Beaver Lake by a very short channel to the southwest. To the northeast, a wetland area separates Beaver Lake from the next lake to the north, Trappers Lake. The entire watershed measures 2694 hectares (6656 acres) (Doepke 1972), and it is heavily forested with northern hardwood and mixed conifer species typical of the area. Wildlife that use the surrounding area and the lake include deer, beaver, otter, mink, muskrat, eagles, loons, gulls, waxwings, mallards, mergansers, herons, and sandpipers (Doepke 1972).

Shoreline around Beaver Lake is relatively smooth and oval with no deep embayments or islands. Shoreline development equals 1.25 (Doepke 1972). Along the southeast side of the lake, there is a gradual incline from the basin, and a steeper incline rises from the southwest side. Forest extends to the sandy shore of the lake, and a steep sand terrace encircles the lake.

Use of the lake and surrounding area is probably primarily limited by its isolated locale, but recreation activities include fishing as well as hiking along the southwest and north sides of the lake. There is a campground and boat launch on Little Beaver Lake, and the boating distance to Beaver Lake is easily traversed.

With a maximum depth of 13 m (mean depth = 6 m) and a surface area equal to 310 hectares, the total volume of Beaver Lake equals 21,071,033 m^3 (Kamke 1987). Water retention time is 2.9 years (Doepke 1972). The long fetch, 3406 m, combined with prevailing winds contribute to frequent mixing of the water column, so it can be described as a polymictic lake (Kamke 1987). The water is fairly clear, and mean Secchi depth for the summer of 1997 was 4.3 m. Surface water temperature in the limnetic zone from May-October of 1997 ranged from 9.12-22.26 ^{0}C with a mean of 18.2 ^{0}C, and the dissolved oxygen range was 8.49-12.74 mg/L with a mean of 9.56 mg/L for the same period.

Along the littoral zone of the lake, sand dominates the sediment, and a shallow shelf extends far into the lake before a steep drop-off. A sand-shelf in the eulittoral zone is the result of an early 19th century dam in Beaver Creek (Loope 1993). Limnetic zone sediment is thicker muck with more organic material.

Aquatic vegetation in the littoral zone includes such species as *Potomogeton gramineus*, *Potomogeton praelongus*, *Chara* spp., and *Nitella* spp., according to an Institute for Fisheries survey in 1953. The lake is a popular fishing area, and resident populations include yellow perch, white sucker, northern pike, and rock bass (Doepke 1972). The lake is also stocked with walleye by the Michigan Department of Natural Resources (Heritage Research Ltd 1999). The exotic zooplankter, *Bythotrephes cederstroemi* has been found in Beaver Lake (Lora Loope, biotechnician Pictured Rocks National Lakeshore, *personal communication*, June 22, 1998).

According to Kamke (1987), Beaver Lake can be classified as mesotrophic, based on the watershed, biota, and nutrients.

Figure 4-5. Photograph of Beaver Lake.

Grand Sable Lake, Pictured Rocks National Lakeshore

Grand Sable Lake is located near the northern end of Pictured Rocks National Lakeshore, just south of the Grand Sable Dunes. The lake is located entirely within national lakeshore property, and a buffer zone extends three miles south of the southern end of the lake. It was previously understood to be a glacial kettle lake lying along a northeast-southwest axis (Kamke 1987), but new evidence suggests it is partially a result of dune-damming (Loope et al. 2000).

Sable Creek flows out of the north end of the lake and connects to Lake Superior, and Rhody Creek enters the lake on the southwest end of the lake. The north end of the lake borders the Grand Sable Dunes. The areas surrounding the lake are used for hiking and recreation, and no residences are located in the vicinity. Picnic areas, a hiking trail, and a boat launch attract visitors to the lake, and a road along the northwest side of the lake encourages visitation.

The watershed measures 3263 hectares (8064 acres) (Doepke 1972), and the soil is not fertile. Forested areas surround the lake except in areas where parking lots, trails, and facilities have been developed. Wildlife that inhabit the watershed and that use the lake include deer, beaver, otter, mink, and muskrat, loons, gulls, waxwings, mallards, mergansers, herons, and sandpipers (Doepke 1972).

The land rises from the shore of the lake, so the terrain is forested hills and dunes. A few embayments around the lake contribute to a shoreline development factor of 1.65, but there are no islands. Total shoreline length equals 4298 m (Doepke 1972).

Grand Sable Lake is a large cold lake with a surface area of 255 hectares (630 acres), and an exchange time of 3.7 years (Doepke 1972). Maximum depth is 22 m, and the mean depth is 10.5 m. During the summer months, the mean Secchi depth is 3.5 m. When wind is out of the southwest, it can create considerable wave action on the lake due to the long fetch of 1335 m. Surface temperature during the summer months has a range of 13-21 ^{0}C, with a mean of 18.4 ^{0}C. During the same period, surface dissolved oxygen ranged between 8.5 and 10.2 mg/L with a mean of 9.44 mg/L.

The sediment throughout the lake is primarily sand with some pebbles. Some organic material can be found in the embayments around the lake, but due to sand input and an infertile watershed, only a small fraction of sediment provides organics to the system. In part because of this, aquatic vegetation is sparse. With high waves and few organics, little vegetation can survive; however, some aquatic vegetation can be found in the protected embayments.

The lake is stocked with fish due to its popularity as a sport-fishing site. Stocked fish include rainbow trout, smelt, lake trout, and splake. Lake trout have been stocked almost yearly since 1960 (Heritage Research Ltd 1999). Dominant fish that inhabit the lake include yellow perch, rock bass, white sucker, and northern pike (Kamke 1987).

Figure 4-6. Photograph of Grand Sable Lake.

Sargent Lake, Isle Royale National Park

Sargent Lake is located on the northern side of the Greenstone Ridge at Isle Royale National Park where it lies along the same northeast/southwest axis as the island of Isle Royale. Like all of the inland lakes on Isle Royale, it surfaced when the water level of Lake Superior dropped after the glaciers receded. Since 1976, 99% of Isle Royale has been designated as a wilderness area, and in 1980, the island was designated as an international biosphere reserve. The inland lakes, therefore, are not subject to direct cultural degradation.

According to information obtained from a GIS map at Isle Royale National Park (Jack Oelfke, Resource Management Specialist Isle Royale National Park, *personal communication* October 1997), the watershed surrounding Sargent Lake has vegetation similar to the rest of the island. The watershed includes approximately 1145 hectares (2829 acres) of land. It is heavily forested with some rock outcroppings. The dominant vegetation is aspen and birch forest, 742 hectares, and aspen and fir compose 144 hectares of the watershed. Other vegetation types include lowland brush, swamp conifer, and upland brush, and bedrock outcroppings make up 17.9 hectares of the watershed. A small creek drains into Sargent Lake on the northwest end of the lake, and a larger creek flows out of the southwest end of the lake to McCargoe Cove along a declining slope to the northwest. There is a small wetland area near the northeast end of the lake and quite a large wetland along the larger creek. Along the south side of the lake is a steep incline up to the Greenstone Ridge, and the north side has a much lower grade incline. Soil analysis has revealed high concentrations of mercury in soils within the Sargent Lake watershed (Woodruff 2000).

The Sargent Lake shoreline is highly convoluted with many bays and land projections. Additionally, ten islands, including one fairly large island, dot the lake. Shoreline length is 15,343 m, and shoreline development equals 3.64. The addition of island shorelines, 1449 m, increases the shoreline development to 3.98. Aspen and birch forest, primarily, encircles the lake, and a large rock outcropping is located along the northeast shore. Driftwood and downed trees are common around the shore; some aquatic grasses and vegetation are rooted in the littoral zone. Moose frequent the shoreline to drink from the lake. The lake is used for canoeing, kayaking, and fishing, and the watershed includes some hiking trails; however no trails lead directly to the lake, so it is even more isolated than other inland lakes on the island.

Sargent Lake has a mean depth of 8.2 m, and the deepest site lies in the southwestern basin of the lake where maximum depth is between 14 and 15 m. Surface area measures 141.5 hectares (349.7 acres), and the irregular shape of the lake limits the fetch. The water is brownish in color and fairly clear. Sargent Lake can be classified as an oligotrophic northern lake, which is characterized by clear water, cool temperatures, and high oxygen content. Secchi depth throughout the summer averages between 3.5 and 4 m. The lake is dimictic, and surface temperature in May through September of 1997 averaged 18.6 ^{0}C. The high temperature of 22.1 ^{0}C was recorded in August, and the low temperature of 12.8 ^{0}C was recorded in May. The bottom had only a slow and slight temperature rise over these months, from under 8 ^{0}C in May to just over 9 ^{0}C in September. Surface dissolved oxygen for the same period had a mean of 10.2 mg/L.

In the eastern basin of the lake, the littoral zone is gradually sloping to a depth of 7.9 m. In the western basin, the littoral zone is steeper, particularly around the deepest portion of the lake. The littoral sediment is sand and gravel with many larger rocks. Sediment in the limnetic zone is thick silt with some pebbles.

Figure 4-7. Photograph of Sargent Lake.

Siskiwit Lake, Isle Royale National Park

Siskiwit Lake is located on the south side of Isle Royale toward the center of the island, in Portage Lake Volcanics with sandstone and conglomerate. Isle Royale's bedrock is a thick pile of lava flows and sedimentary rocks that have been tilted toward the southeast, then eroded by streams to form a ridge-and-valley topography (Huber 1983). The lake was formed as glaciation accentuated the existing topography and interrupted the preglacial stream channels with lake basins.

The watershed is forested and currently used only for recreation. The dominant vegetation is aspen and white birch forest, followed by lowland brush, aspen, and balsam fir forest, upland conifer forest, and swamp conifer forest. The watershed is 7284 ha total, with 1605 ha of that area Siskiwit itself. The lake has eight inlets and one outlet (sometimes called Siskiwit River) and is connected to Wood Lake by a narrows. Intermediate Lake, Mud Lake, and several small, unnamed lakes also lie within the watershed. The outlet flows less than 1 km, falling 17.7 m, to Malone Bay of Lake Superior.

The Siskiwit shoreline is forested and rocky, with many bays and peninsulas. The lake contains 13 islands, with Eagle Nest Island, Ryan Island, and Teakettle Island being the only ones named. The lake shoreline is 31,776 m in length, and the shoreline development factor equals 2.24. Shoreline of the islands equals 6641 m, and the additional length makes a shoreline development factor of 2.71.

Siskiwit Lake is used for recreation, mainly fishing and boating. It is the most heavily fished water on Isle Royale, partly due to its relative accessibility via a short portage from the Lake Superior shoreline. No motors are allowed on the lake.

Siskiwit Lake has a mean depth of 25 m and a maximum depth of 49 m. The surface area of the lake is 1605 ha. It is oriented west-southwest to east-southeast and has a long fetch, which can encourage large waves, particularly in a west wind.

The water in Siskiwit is very clear, with little color. The Secchi transparency averages 8 meters during summer. The lake is dimictic, mixing early in the spring and maintaining a strong stratification throughout the summer. Bottom temperatures during summer stratification range between 4^{o} and 7^{o} C, while surface temperatures can reach over 20^{o} C.

Siskiwit's bottom consists of bedrock, glacial till, and sand in some shallow bays. The limnetic site is very rocky, whereas the littoral site is somewhat sandy, with much rock and timber debris (Toczydlowski et al. 1980).

Siskiwit Lake supports many species of aquatic plants, mostly in the shallower east and west ends of the lake. They include floating leaved pondweed, Richardson's pondweed, long leafed pondweed, northern naiad, bladderwort, burreed, spike rush, yellow water lily, and cattail.

A popular lake for fishing, Siskiwit's sport fish include brook trout, lake trout, northern pike, and yellow perch. Whitefish and lake trout are the dominant fish species. Other species include burbot, common sucker, trout perch, nine-spined stickleback, stripenose, spottailed, and lake shiners, two species of Miller's thumb, and possibly the chub and lake herring (Lagler and Goldman 1982).

Small flagellates and *Cyclotella melosiroides* dominate Siskiwit's phytoplankton community, with subdominants of *Asterionella*, *Rhizosolenia*, *Oocystis*, and *Oscillatoria* (Toczydlowski et al. 1980).

Along Siskiwit's rocky shore near its outlet and in sheltered areas of the lake, occur the molluscs *Limnaea stagnalis*, the dominant species, along with *Planorbis bicarinatus royalensis*, *P. campanulatus*, *Lampsilis luteolus*, *Anodonta marginata*, and *A. grandis footiana*. *Planorbis trivolvis*, *Anodonta marginata*, and *A. grandis* occur in sheltered areas in sand or mud. Water striders (*Gerris remigis*) and whirligig beetles, (*Gyrinus minutus*) are common (Adams 1909).

Siskiwit is an oligotrophic lake, with low nutrient concentrations. The water is quite soft, so it is probably susceptible to acid precipitation damage. Polychlorinated dibenzo-*p*-dioxins and dibenzofurans, presumed to have arrived through atmospheric deposition, have been found in the lake's sediment (Czuczwa et al. 1984).

Figure 4-8. Photograph of Siskiwit Lake.

Locator Lake, Voyageurs National Park

Locator Lake is located within Voyageurs National Park near International Falls, Minnesota. It is part of a chain of lakes created by glacial retreat. There are two inlets and one outlet of this drainage lake. War Club Lake flows into Locator at the east end. There is also swamp drainage into the lake. The only outlet is Cranberry Creek, which then flows into Cranberry Bay. Locator Lake is only accessed by portage trails. The lake is mainly used for recreational fishing and boating in canoes or small, motorized boats.

The watershed is primarily second growth mixed forest, made up of conifers and hardwoods, on steep rocky hills. The primary use of the watershed is recreation, mainly hiking. A campground and boat launch are located on the south shore.

The shoreline length of Locator Lake is 6530 m, with a shoreline development of 2.45 (Larry Kallemeyn, biologist U.S. Geological Survey, *personal communication* October 9, 1997). There are no islands within the lake but there is a bay (Long Finger Bay) on the south side of the lake. The shoreline is rocky and slopes toward the lake. Vegetation around the shoreline is limited.

Locator Lake has a surface area of 56.7 hectares (140 acres) and a maximum depth of 15.9 m (mean=8.1 m). Water clarity is low due to humic acids. The water retention time is approximately 1.8 years (Webster et al. 1993). The lake is dimictic with stratification occurring in the spring. The littoral zone of Locator Lake makes up a total of 44 acres. There are large amounts of woody debris in the littoral zone. Limnetic zone sediment is thick and black.

Aquatic vegetation covers only 2 % of the lake surface. The east and west nearshore are the most heavily vegetated with emergents. Emergent vegetation includes: sweet gale, three-way sedge, spikerush, and narrow leaf sedge. The inlet, outlet, and shallows of the bays have submergent and floating macrophytes. These consist of watershield, white water lily, wild rice, little yellow water lily, yellow water lily, floating leaf burreed, water milfoil, and bladderwort (MNDNR 1983).

The fish community is made up of largemouth bass, johnny darter, northern cisco, northern pike, and rock bass. The most commonly caught fish in Locator is the northern pike. This is also the only species of fish that have a good spawning area within the lake. The spawning area is located in the marsh areas of the outlet and in Long Finger Bay. All other fish spawning areas were listed as poor to fair due to limited vegetation within the lake (MNDNR 1983).

Figure 4-9. Photograph of Locator Lake.

Mukooda Lake, Voyageurs National Park

Water dominates the landscape of Voyageurs National Park located on the northern border of Minnesota, about 15 miles from International Falls. Approximately 40% of the park's area is composed of four large lakes: Rainy, Kabetogama, Namakan, and Sand Point. Geologically, the park is located on the southern end of the Canadian Shield, which represents some of the oldest exposed rock formations in the world. Much of the land area is rolling hills interspersed between bogs, beaver ponds, swamps, and smaller lakes. Mukooda Lake is located near the southeast end of the park and is accessed by a short portage from the southern end of Sand Point Lake.

The watershed area around Mukooda Lake is primarily undeveloped second growth mixed forest on moderately steep rock banks. Dominant soil types are bedrock, sand, and rubble on sand. The watershed also contains several flat bogs that are dominated by spruce and alder. There are no inlets into the lake, but a beaver dam impairs an unnamed outlet on the eastern side of the lake (MNDNR 1997).

The shoreline of Mukooda Lake is 8.06 km, encompassing a surface area of 305 hectares (754 acres) (Kallemeyn 1997). The three islands have a combined shoreline length of 0.8 km (0.5 mi) and a total area of 1.46 ha (3.87 acres). Shoreline development without the islands is 1.30 and 1.43 including the islands. Nearby there is one campground that is quite busy during the summer.

The lake basin has a mean depth of 12.25 m, with a maximum depth of 23.77 meters. Maximum length and width of the lake are 2.86 km and 1.48 km, with a fetch of 1.09 km. This equates to a surface area of 304.98 ha (753.6 acres) with a volume of 37,351,090 m^3 and a watershed area of 754 ha (1863.1 acres) (Kallemeyn 1997).

Mukooda Lake is dimictic and has clear water, with an average Secchi depth of 4.25. The most common shoalwater substrates are ledge rock and sand.

There are several species of aquatic vegetation at Mukooda Lake, up to a depth of about 4.9 m (15 feet). The most abundant are *Lobelia dortmanna, N. flexis,* and *Potamogeton* sp. (variable pond weed) (MNDNR 1983). The Minnesota Department of Natural Resources (MDNR) stocks lake trout in the winter and largemouth bass and crappie in the summer. In 1997, the MDNR conducted seining and gillnet catches on Mukooda Lake. *Perca flavescens* (yellow perch) and *Pimephales notatus* (bluntnose minnow) were the most abundant species caught by seining, and *Coregonus artedi* (tullibee or cisco), *Pomoxis nigromaculatus* (black crappie), *Catostomus commersoni* (white sucker), and *Salvelinus namaycush* (lake trout) were the most abundant species caught by gillnetting.

Figure 4-10. Photograph of Mukooda Lake.

Chapter 5 Methods

Field sampling was organized and coordinated by personnel at Lake Michigan Ecological Research Station (LMERS) to have sampling take place simultaneously at all parks. At the start of the field season, sampling schedules were sent to the parks. Sampling days were selected so that samples collected at Isle Royale could be mailed out to the laboratory in a relatively timely fashion. Flexibility was built into the schedule by selecting Wednesdays as a sampling day and allowing parks to sample on the surrounding days.

All of the parks had identical field sampling equipment so that field instructions could be uniform. The first year of sampling, 1997, was conducted after LMERS personnel trained field technicians individually at the parks and then provided written field and processing instructions. Between the first and second seasons, a field manual was developed that provided specific, written instruction for use in 1998 and additional background information. Also in 1998, LMERS personnel traveled to the parks that had hired different technicians unfamiliar with the sampling protocol.

Unless otherwise noted, all types of sampling were performed from May-September in 1997 and 1998 and in May of 1999. Additionally, winter depth profiles were collected between January and March of 1998.

Field Methods

At each park, two lakes were selected for sampling (only one lake at Indiana Dunes National Lakeshore). Some of the park technicians were occasionally able to sample both lakes on the same day: Sleeping Bear Dunes and Pictured Rocks; at other parks, technicians needed two days to sample their lakes: Isle Royale, and Voyageurs. A fixed site was selected in both the littoral and limnetic zones of each lake from which all samples were collected (Loftis and Ward 1982, van Belle and Hughes 1983).

Every two weeks, technicians went out to the study lakes for sampling. Before going to the lakes, the YSI 6820 sonde was calibrated. The conductivity probe was calibrated at least every six weeks, and the dissolved oxygen and pH probes were calibrated each time before going out in the field, using absolute barometric pressure for calibrating the oxygen probe.

On every second visit to the lakes, the technicians collected depth profiles at the deepest point in each lake (as determined by available bathymetric maps). Using a YSI 6820 sonde attached to a 610-DM readout, measurements were taken at the surface and in one-meter increments to the bottom of the lake, as measured on the sonde depth gauge. Parameters measured included dissolved oxygen, pH, conductivity, temperature, and total dissolved solids. Lakes at Sleeping Bear Dunes were analyzed for the same parameters, but the park used a Hydrolab Datasonde 3 instead of the YSI model sonde. Additionally, weather conditions were recorded in the field including air temperature, wind speed and direction, and recent weather history.

On alternate visits, a full suite of samples was collected, including water and sediment for chemical analysis, phytoplankton, zooplankton, and benthic macroinvertebrates. Sampling at the limnetic site began with recording weather conditions and field notes and measuring Secchi depth. Depth profiles were taken using the 6820 sonde, and the same set of parameters was measured. All of the readings were collected on the 610-DM to be recalled and uploaded later on the computer. Sampling descriptions that follow were conducted at both the limnetic and littoral sites in each lake.

Using a Kemmerer sampler, water was collected for chemical analysis (Lind 1985, American Public Health Association 1995). During lake stratification, separate water samples were collected from the epilimnion and hypolimnion. The sampler was lowered to one meter below the surface to collect epilimnetic samples, and three bottles supplied by the chemistry laboratory were filled. The sampler was then lowered to one meter above the lake bottom to collect hypolimnetic samples, and three laboratory bottles were filled. In May of 1997, samples were collected differently: epilimnetic samples from the middle of the epilimnion and hypolimnetic samples from the middles of the hypolimnion. During lake mixing, samples were collected from near the bottom and near the surface and composited into three sample bottles supplied by the laboratory. Water samples were preserved according to the tests to be conducted; each of the bottles supplied by the laboratory contained sulfuric acid, nitric acid, or no preservative. All water samples were placed in a cooler immediately after collection.

Again using the Kemmerer sampler, water was collected from one meter below the surface for whole water phytoplankton samples (Welch 1948, American Public Health Association 1995). Three one-liter plastic bottles were filled, and Lugol's solution was added to each (approximately 2 milliliters) and mixed gently (American Public Health Association 1995). Samples were placed in a location out of direct sunlight to prevent degradation. Phytoplankton samples were not collected in May of 1997 because of a request from park managers for a change in protocol.

At one-meter below the surface, water was collected for chlorophyll *a* analysis using a Kemmerer sampler. Using a Nalgene filtering apparatus and hand pump, the water was filtered on a 0.45-um filter in the field (American Public Health Association 1995). The filter was wrapped in aluminum foil, labeled with date, lake, and volume filtered and was then placed in a cooler. Gelman glass-fiber filters were used in 1997, and Millipore membrane filters were used in 1998 and spring, 1999 when it was determined that membrane filters would be more accurate according to laboratory protocol.

Zooplankton samples were collected in the limnetic zone by vertical tow using a Wisconsin net with 80-um mesh. The net was lowered to one-meter above the lake bottom, without touching the sediment, and it was slowly pulled to the surface at a rate of 0.5-1 meter/second (American Public Health Association 1995). The outside of the net was rinsed to concentrate the zooplankton in the collecting bucket, and the contents were poured into a plastic sample jar. In 1998, a narcotizing procedure was added to the protocol whereby the netted collecting bucket, containing the sample, was placed in a plastic bowl containing water and Alka-Seltzer for one minute before being poured into the sample jar (Gannon and Gannon 1975). This additional procedure was to aid in the identification of soft-form organisms such as rotifers (Gannon and Gannon 1975). The bucket was completely rinsed into the jar using filtered water, and the sample was dyed and preserved with approximately 2 milliliters of Lugol's solution (American Public Health Association 1995). This was repeated two more times, for a total of three zooplankton vertical tows at each site.

Zooplankton samples were collected in the littoral zone by horizontal tow using a Wisconsin net with 80-um mesh. Using an underhanded toss, the net was thrown away from the boat and parallel to shore to a distance of 5 meters. The net was retrieved by slowly pulling it at a rate of 0.5-1 meter/second without allowing the opening to rise above the surface or sink below the top layers of water. Upon retrieval, zooplankton were rinsed down into the collecting bucket, and the net was tossed and retrieved again in the same manner. The net therefore had been towed through a total of 10 meters of water. After the second retrieval and rinse, the sample was processed in a manner identical to the technique used in the limnetic zone.

In May of 1997, a 63-um mesh Wisconsin net was used for collections at all parks except Voyageurs due to a delay in equipment delivery. In the same month, limnetic zooplankton tows at Sleeping Bear Dunes were two combined tows from the Secchi depth to the surface, collected before protocol refinement.

Benthic macroinvertebrate samples were collected last to avoid stirring up the water column during other collections. An Ekman dredge sampler was set and lowered to the lake bottom. After tripping the Ekman to take a grab sample, it was raised to the surface and the contents were emptied into a 30-um sieve bucket (Lind 1985, American Public Health Association 1995). The Ekman itself was rinsed of any remaining sediment, and the sample was sieved at the water surface until all of the small sediment was removed from the sample. The remaining sample was emptied into a one-liter plastic jar, and the sieve bucket was rinsed. Ethanol was added for preservation to make 70% of the total volume of the sample. This was repeated for a total of three samples at each site.

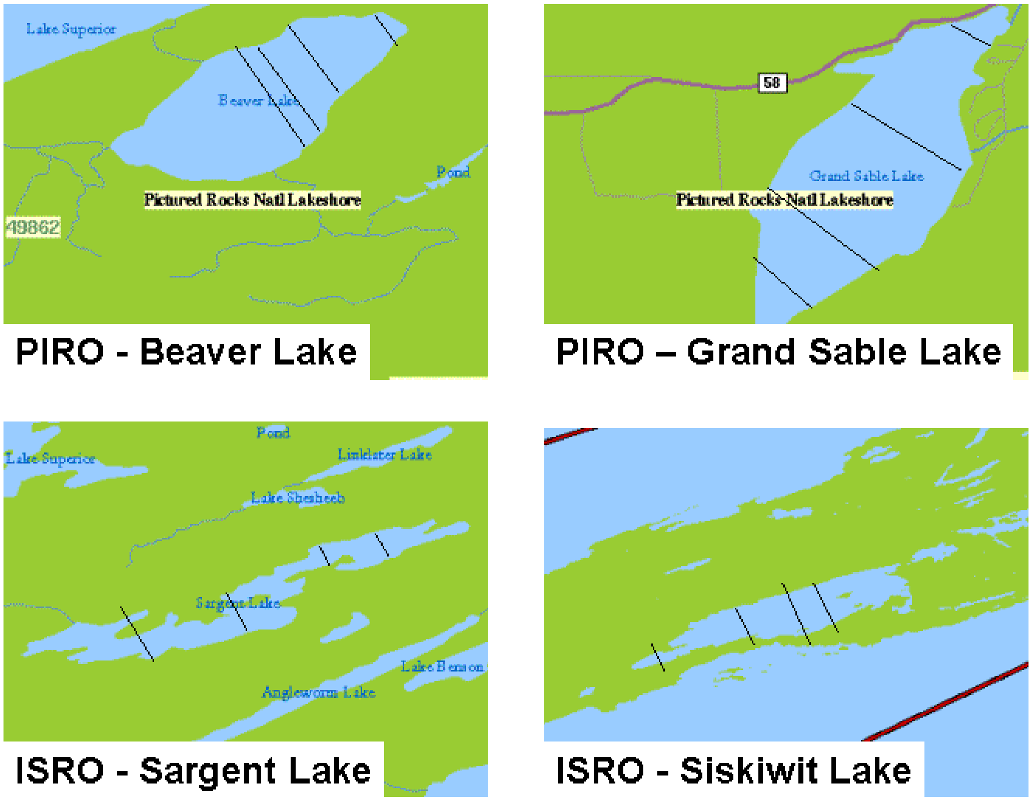
Sediment samples for chemical analysis were collected in 1997 using the Ekman dredge (Lind 1985, American Public Health Association 1995). Once a month, samples were collected for nutrient analysis, including phosphorus, total Kjeldahl nitrogen, and nitrate plus nitrite as N. After June, total organic carbon was added to the list of nutrients analyzed. In August of 1997, samples were collected to measure concentrations of heavy metals, volatiles, nonvolatiles, and pesticides. After setting and tripping the Ekman dredge, it was brought to the surface, and a portion of the sample was taken from the top of the device using a teflon spoon. Portions of three Ekman grabs were composited into each laboratory jar. Samples were placed in a cooler after collection. At Voyageurs, sediment samples for heavy metals were collected in June while sediment samples for volatiles, and nonvolatiles were collected in August.

In the littoral zone, the same procedures were followed with minor variations. The littoral site was selected based on depth and distance from shore. Depth profiles included readings at the surface and at one meter. Water samples for chemical analysis were collected one-meter below the surface. Neither zooplankton nor phytoplankton were collected in the littoral zone in 1998 or 1999 after a consensus between park managers and the project director regarding its usefulness for the purpose of this study.

Macrophyte surveys were conducted once during the course of the study at Isle Royale, Pictured Rocks, Sleeping Bear Dunes, and Indiana Dunes. Personnel at Voyageurs did not want macrophyte surveys conducted on Locator and Mukooda because previous studies had been conducted.

Isle Royale and Pictured Rocks: Each lake was divided into 50-70 sections. Sections were perpendicular to each lake’s longest axis. The sections were numbered, and a random number generator was used to select four sections from each lake (Figure 5-1). Within a section, the sampling area was chosen randomly. Both shores in a section were sampled, and when islands fell in a section both shores of the island were sampled. In each section, macrophyte cover was estimated by observations from a canoe or by snorkeling. In analysis, the whole-lake coverage was calculated as a mean (n=4) of the coverage estimates for each transect, including area of the lakes that were not specifically surveyed (i.e., the profundal zone where plants would not be expected to occur). Species composition was recorded, and voucher specimens were collected. In addition, the entire perimeter was qualitatively surveyed to estimate if the sections were representative of the entire lake (in all cases, sections were representative), and some plants

Figure 5-1. Maps of the 4 lakes surveyed at Pictured Rocks and Isle Royale. Black lines indicate the locations of the sections sampled. [1] Transects are numbered 1 through 4 from left to right.



were collected for species identification. Macrophytes were identified at Indiana Dunes using Fassett (Fassett 1957) and Voss (Voss 1972, 1985, 1996).

Sleeping Bear Dunes: Plant surveys were conducted between July 11 and 29, 1998. Plants were collected from four transects at North Bar and Round Lakes and from three transects at Round Lake. Randomly selected transects were started at the shoreline and extended 50 meters perpendicular to shore or until water depth was 1.5 to 2.0 meters. Plants were collected by hand or from a canoe/rowboat. At deeper sites, a bottom rake was used for submerged plants.

Indiana Dunes: Plant surveys of Long Lake were conducted on July 31, 1998. Two 500-meter zones were sampled along the northeast and northwest shores of the lake. Starting at the boat launch on the north shore, sampling was conducted moving to the east and then moving to the west. All aquatic macrophytes were identified or collected for later identification.

[1] Grand Sable Lake was divided as above, but the map was lost prior to sampling. In this case, the sections were selected randomly from evenly-divided lake quarters (also perpendicular to the longest axis).

Table 5-1. Sampling dates of the macrophytes survey conducted.

Park	Lake	Date
ISRO	Siskiwit	2 August 2000
ISRO	Sargent	4 August 2000
PIRO	Beaver	6 August 2000
PIRO	Grand Sable	7 August 2000

Pre-Processing

After returning from the field, technicians placed the chlorophyll *a* filters in the freezer until shipment to LMERS. Water and sediment chemistry samples were refrigerated until samples from both lakes had been collected. Both sets of samples were packed in coolers with chain of custody forms and shipped overnight to the laboratory for analysis. Samples from Isle Royale were shipped to the mainland and mailed the following day, so there was some delay in processing these samples. Phytoplankton and zooplankton samples were stored in a dark location to avoid degradation of the Lugol's and were then shipped to LMERS along with benthic macroinvertebrate samples for analysis. Profile data was uploaded from the 610-DM display into PC6000 or EcoWatch software (provided by YSI). All files were sent electronically or mailed on disk to LMERS.

Laboratory Methods

A contracted laboratory processed all of the water samples. The initial contracted laboratory, Edglo Laboratories in Fort Wayne, Indiana, was used for May and June of 1997, but for July 1998 and all subsequent months, another laboratory was used at the request of park personnel: Quanterra Analytical Laboratories, Arvada, Colorado (now STL, Inc.). Water samples at both laboratories were analyzed according to EPA approved methods (US EPA 1983). Parameters analyzed and methods used include ammonia (US EPA method 350.1), nitrate plus nitrite (353.2), total phosphorus (365.3), silica (6010), chloride (300.0), calcium and magnesium hardness (6010B), alkalinity (310.1), and sulfate (300.0). Silica was not included in the analysis until July of 1997.

Quanterra Laboratories was also responsible for processing and analyzing sediment samples for chemical analysis. Metals were analyzed by EPA methods, plasma emission spectroscopy (Al, Sb, As, Ba, Be, B, Cd, Ca, Cr, Co, Cu, Fe, Pb, Li, Mg, Mn, Mo, Ni, P, K, Se, Ag, Na, Sr, Tl, Sn, V, Zn (Method 6010), and Hg (7471)) (US EPA 1999). Organic pesticides were analyzed using method 8080, GC/MS (US EPA 1999). Volatile organics were measured using method 8260, GC/MS, and semivolatile organics were analyzed using method 8270, GC/MS (US EPA 1999). The GC/MS detections were made using library matches and examination of masses by the analyst. No external standards were used for GC/MS detections. Consequently, the exact compound identification is not assured, although the correct classes of compounds are more likely. Other variables measured included total Kjeldahl nitrogen (351.2) and total phosphorus (365.3) (US EPA 1983), silica (6010) and organic carbon (9060) (US EPA 1999).

Phytoplankton samples were composited using 100 ml from each of the triplicate samples. Following Standard Methods (American Public Health Association 1995), thirty drops of glutaraldehyde (1 drop/10 ml) were added to each sample as a preservative, and the samples were gently mixed and refrigerated for four hours. After refrigeration, samples were vacuum filtered in 25-ml and 50-ml sub-samples using Millipore 0.45-um filters (gridded). Before filtering completely, the membrane was rinsed with 20 ml 50% ethanol, followed by 20 ml of 95% ethanol. Filters were placed on a microscope slide containing clove oil and were then covered with additional clove oil. After placing a coverslip over each filter, all slides were placed in a closed, dark drawer for 10-14 days to clear the filter, after which slides were ready for examination. Slides were examined using a Nikon Labophot-2 and a Zeiss Laboval-4 at 400x and 1000x to clarify species. Starting with fields-of-view near the center of the slide, at least 200 cells were identified and counted in each sample to give the relative abundance of each taxon, using natural unit counts (Ingram and Palmer 1952). In the case of blooms of 100 cells or more, counts of the dominant taxon were discontinued. At that point the number of fields-of-view was noted and the counts of the remaining taxa were continued to at least 200 cells. Number of phytoplankton per liter was calculated using the number of field-of-views, sub-sample size, and number of cells counted in each taxon. Standard taxonomic texts were used for identification (Patrick and Reimer 1966, Whitford and Schumacher 1969, Patrick and Reimer 1975, Prescott 1982). Phytoplankton were identified by Julia Nefczyk and confirmed by Paul M. Stewart.

Vials containing chlorophyll *a* filters were frozen and transported to LMERS. Chlorophyll *a* determination followed the methods of Wetzel and Likens (Wetzel and Likens 1979) and Standard Methods (American Public Health Association 1995). Filters were removed from the freezer, unwrapped, and placed in 10 ml of 90% acetone; filters were extracted for 22-24 hours in a dark refrigerator. Because glass fiber filters were used in 1997, these had to be ground in a tissue grinder prior to analysis, which procedure followed Wetzel and Likens (1979). After calibrating the fluorometer (Sequoia-Turner model 450) to a spectrophotometer (Bausch and Lomb Spectronic 21) using a chlorophyll standard, as outlined in Wetzel and Likens (1979), extracts were poured into a cuvette and fluorescence was measured. Fluorometric readings were entered into a database, and chlorophyll *a* concentration was calculated.

One-milliliter was measured from a thoroughly mixed zooplankton sample using a Hensen-Stempel pipette and placed in a Sedgwick-Rafter counting cell. Zooplankton were identified on a compound microscope at 100X. If necessary, certain organisms were removed for identification at a higher magnification. Two microscopes were used for identifications: a Nikon Labophot-2 with phase contrast and an Olympus BH-2 with Nomarski differential interference contrast. If a sample was particularly sparse, more than 1 ml of sample was examined for counting. The procedure was repeated for each of the triplicate samples. Identifications were verified with a taxonomic expert. Standard taxonomic texts were used for identifications (Edmondson 1959, Stemberger 1979, Balcer et al. 1984, Pennak 1989, Thorp and Covich 1991). Zooplankton were identified by Laurel Last and Stephanie Mahoney and confirmed by Bruce Davis, Great Lakes Science Center.

Benthic macroinvertebrate samples were picked and sorted for identification in the laboratory. Organisms were identified on an Olympus SZ-60 dissecting scope. In 1997, chironomids were slide-mounted and identified on a compound scope. Oligochaetes were slide-mounted in CMCP and allowed to clear for 24 hours before being identified on an Olympus BH-2 compound microscope at a range of magnifications. A taxonomic specialist confirmed all species identifications. Standard taxonomic texts were used for all identifications (Wiggins 1977, Pennak 1989, Peckarsky et al. 1990, Merritt and Cummins 1996, Kathman and Brinkhurst 1998). Benthic macroinvertebrates were identified by Thomas Sobat in 1997 and Maria

Goodrich in 1998 and 1999 and were confirmed. Thomas Sobat was not available after 1997, and with local expertise, identifications of chironomids could be to family with confidence.

Laboratory results for water and sediment chemistry were received in database format, and all data were checked and organized into Microsoft Access. Biological results, including phytoplankton, zooplankton, and benthic macroinvertebrates, were also entered into the database program.

Profile information supplied in PC6000 or EcoWatch software format was exported to the information database. Profiles were imported into Microsoft Excel and arranged by lake and date. All weather data saved in the 610-DM was uploaded into a word processing program.

Biological samples were preserved for permanent storage at each park.

Statistical Methods

Benthos

Benthos data consisted of three samples collected from one location in each of the littoral and profundal (termed limnetic throughout). Means were generated from these three samples to calculate abundance and relative abundance for the taxa identified. Comparisons between the limnetic and littoral macroinvertebrate communities within each lake consisted of paired t-tests, with means being paired by date. Bonferonni-adjusted $P_{critical}$ was used ($P_{critical} = \alpha$/no. of tests run). Comparisons among lakes were made using ANOVA with time as a cofactor. In no cases did time have a significant effect in the model. Tukey's HSD test was used for multiple comparisons when original ANOVA's were significant. Tests were run using SYSTAT (version 7.0).

Primer (version 4.0) was used to calculate community indexes for the data. Primer calculates Shannon-Wiener diversity index (H′, called Shannon diversity from here) as $-\Sigma_i\, p_i(\log p_i)$, where p_i is the proportion of the total count arising from the ith taxon. Margalef's index (d) is calculated as d = (S-1)/log N, where S is the number of taxa and N is the total number of individuals. Margalef's index is an index of taxa richness or heterogeneity (Clarke and Warwick 1994). Data were polled from all samples to estimate these indices.

Some among-lake similarities were made using Bray-Curtis Similarity, and further analyzed with cluster analysis. Primer (version 4.0) was used for these analyses.

Water Chemistry

Water chemistry data consisted of three samples collected from one location in each of the littoral zone and the limnetic zone. Within the limnetic zone, samples were taken from epilimnetic and hypolimnetic waters. Means were generated from the three samples for each sample date. Comparisons between the different waters within each lake consisted of paired t-tests, with means being paired by date. Bonferonni-adjusted $P_{critical}$ was used ($P_{critical} = \alpha$/no. of tests run). Among-lake comparisons were made with Discriminant Analysis, using SYSTAT (version 7.0).

Zooplankton

Zooplankton samples consisted of three samples collected from one location each in the littoral zone and limnetic zone. Means were generated from the three samples for each sample date to calculate abundance and relative abundance for the taxa identified. Primer (version 4.0) was used to calculate community indexes for the data. Primer calculates Shannon-Wiener diversity index (H′, called Shannon diversity from here) as $-\Sigma_i\, p_i(\log p_i)$, where p_i is the proportion of the total count arising from the *i*th taxon. Margalef's index (d) is calculated as d = (S-1)/log N, where S is the number of taxa and N is the total number of individuals. Margalef's index is an index of taxa richness or heterogeneity (Clarke and Warwick 1994). Data were pooled from all samples in calculations. Because numerous data were collected, community indices were plotted as cumulative data (not pooled) and the value was estimated from the flattened portion of the generated curve.

Chapter 6 Water Chemistry Results

Loon Lake

Based on the clinograde curves (see handbook for definitions of limnological terms) of the temperature profiles (Figure 6-1), Loon Lake was stratified from about late-May to mid-October. The dissolved oxygen profiles exhibited an occasional positive heterograde curve in June of 1997 and also in May-June 1998 (Figure 6-2). The elevated oxygen concentration tended to occur very near, or within, the metalimnion. Production was not extremely high, as evident from the normal clinograde oxygen saturation curves (Figure 6-3).

The water chemistry is summarized by years in Tables 6-1 and 6-2. Parameters examined failed to display significant differences between years in the epilimnion (all t-test P-values > 0.10; P-critical = 0.007 after Bonferonni adjustment). In general, within-year variation was minimal as suggested by SEM's (Standard Error of the Means) of <20% of the means for all parameters.

The water chemistry in the hypolimnion did not show much between-year variation (Table 6-1). All parameters failed to show significant differences (all P-values > 0.09). Water samples from the littoral zone showed similar trends as was observed in the hypolimnion. Ammonia was consistently low, often below detection limit.

When parameters were compared between epilimnion and hypolimnion (paired tests), no parameter showed significant differences. There was also no significant difference between the epilimnion and littoral water samples for all parameters. However, alkalinity (P = 0.002) and silica (P < 0.001) were significantly higher in the hypolimnion compared to the littoral.

Limnology

Alkalinity and hardness suggest that Loon Lake is a hardwater lake. The difference between the hypolimnion and littoral, although significant, was low (<20 mg/L) and likely not of any biological significance. Chloride ion concentrations are higher than some of the other lakes sampled, but they are still close to the average for freshwater and thus do not illicit any concern. The high concentrations of cations were shown in the similarly high specific conductance measurements.

Nutrient concentrations were consistently low. Nitrogen and phosphorus were generally below detection levels. Ammonia was detected on a few sampling dates (see Appendix 1). In 1997, ammonia appeared to increase during the early sampling but dropped to below detection limits by the end of summer possibly because of uptake by primary producers. One sampling date in 1998 showed a high ammonia level, but levels dropped below detection limit by the following sampling date, which indicates that this number may not be accurate. In general, ammonia levels were low and indicate a non-polluted system.

Silica was lower in the littoral than in the hypolimnion, suggesting that diatom production was high in the shallower zone and was able to remove much of the dissolved silica. The diatom production could be associated with epiphytic growth in the littoral. Sulfur did not differ with depth despite stratification throughout much of the summer. This could suggest that the low dissolved oxygen in the hypolimnion was not low enough to drop the redox potential to the point that sulfates would be converted to hydrogen sulfide.

The general conclusions based solely on water chemistry are that Loon Lake is in good condition. The water is very hard, but this is of no concern. The lake should support a healthy community of typical plants and animals found in a temperate lake. The calcium levels are favorable for macrocrustaceans (e.g., crayfish) and mollusks.

Figure 6-1. Temperature profiles for Loon Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

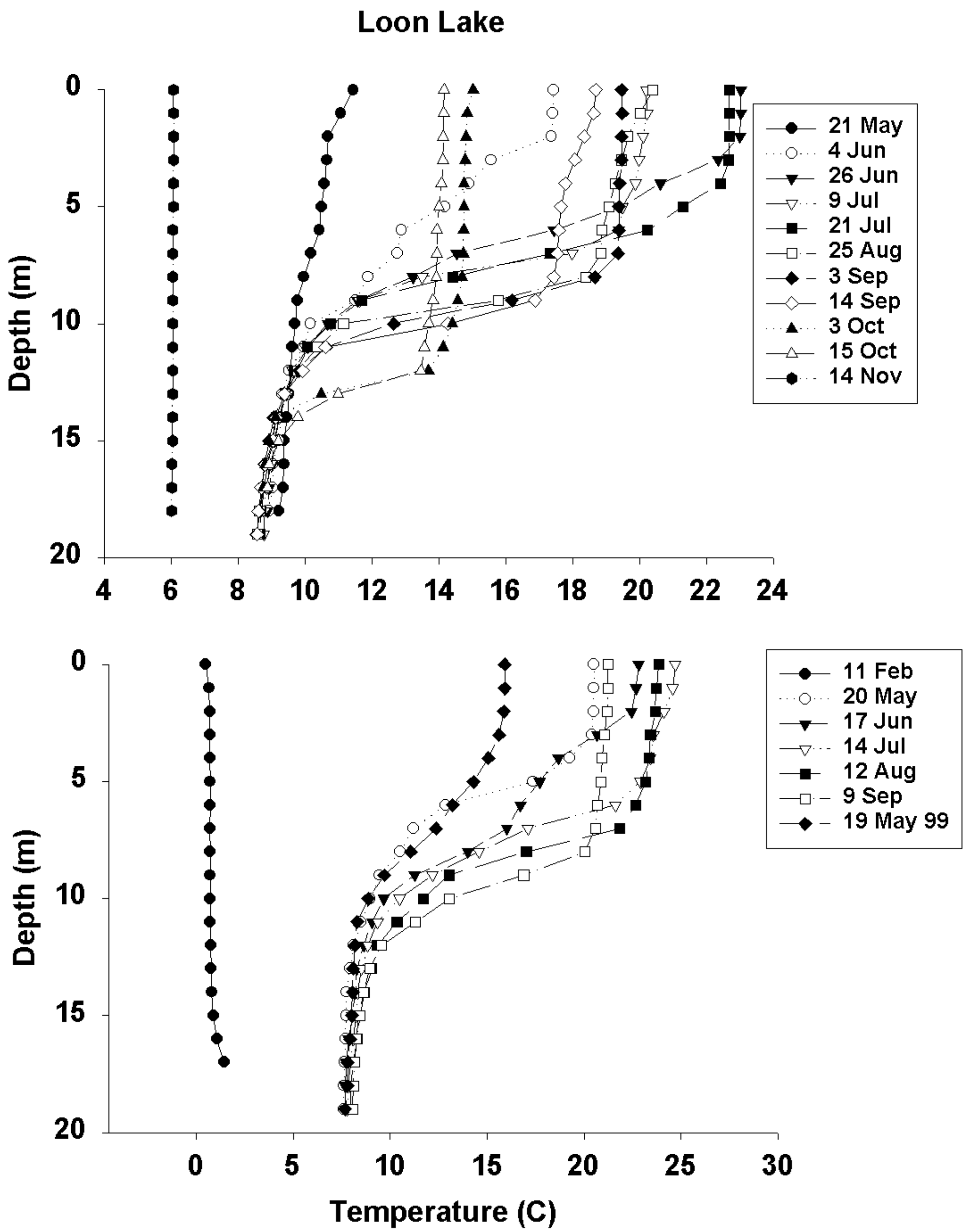


Figure 6-2. Dissolved oxygen profiles for Loon Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

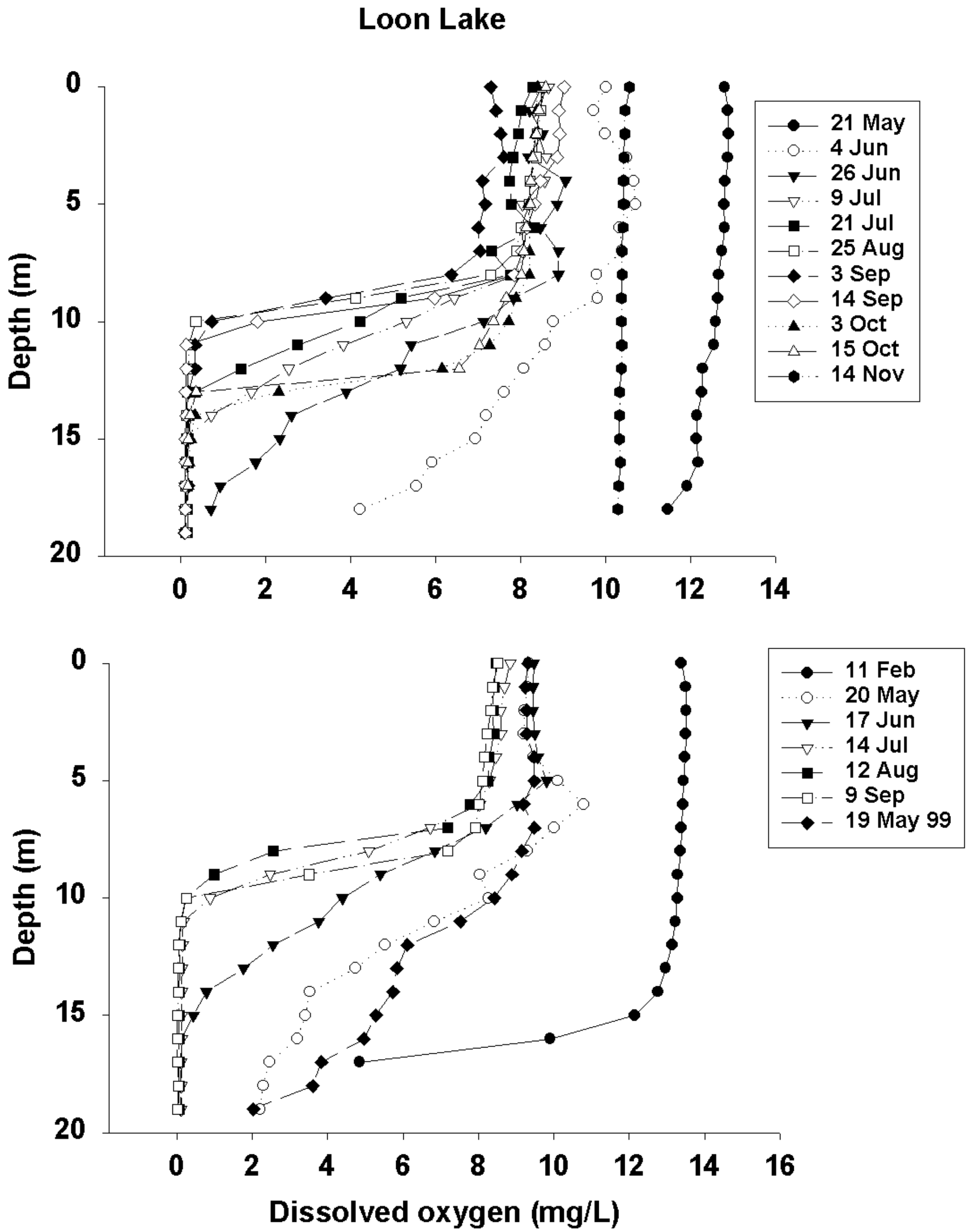


Figure 6-3. Oxygen saturation calculated from measured water temperature and dissolved oxygen in Loon Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

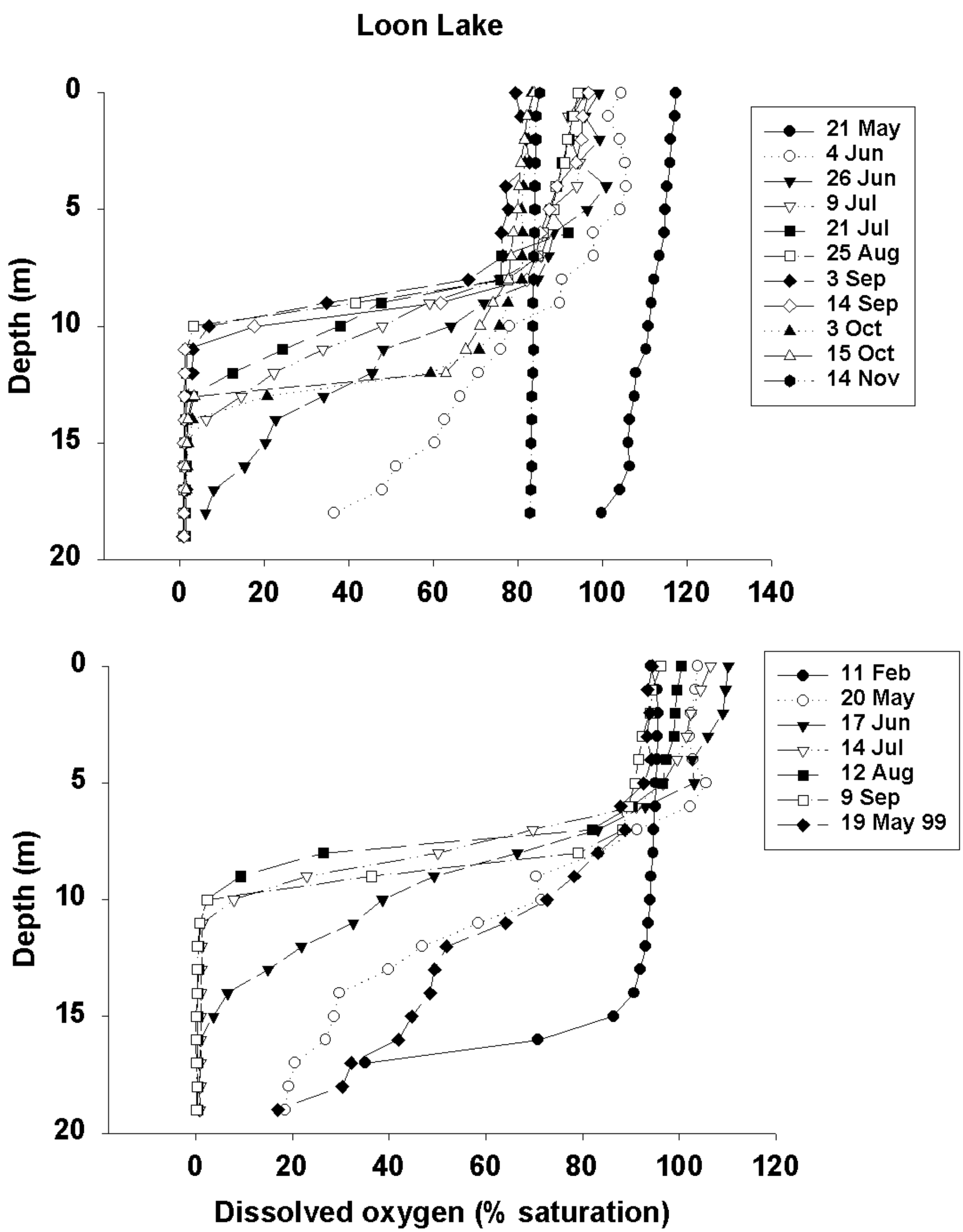


Figure 6-4. pH profiles for Loon Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

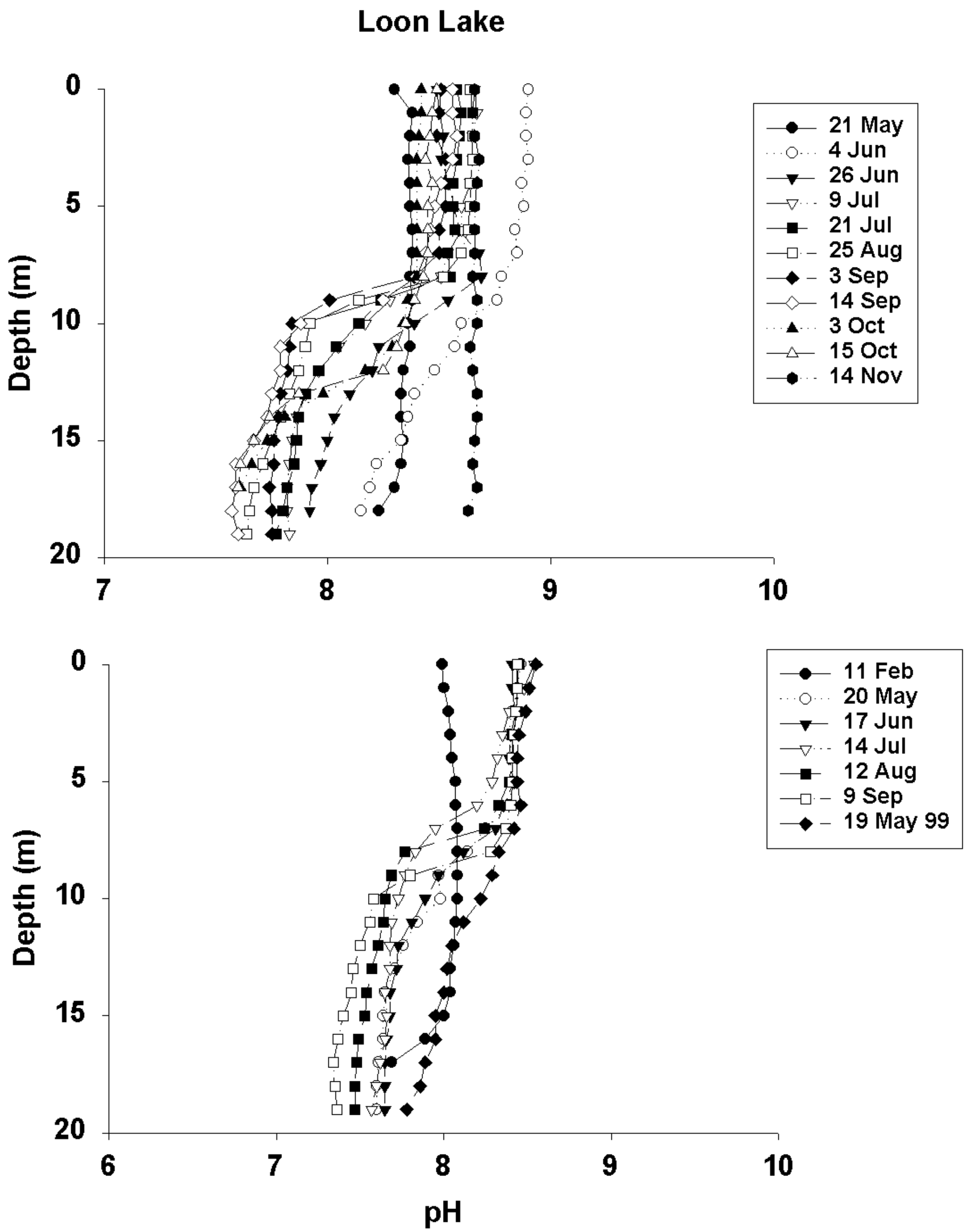
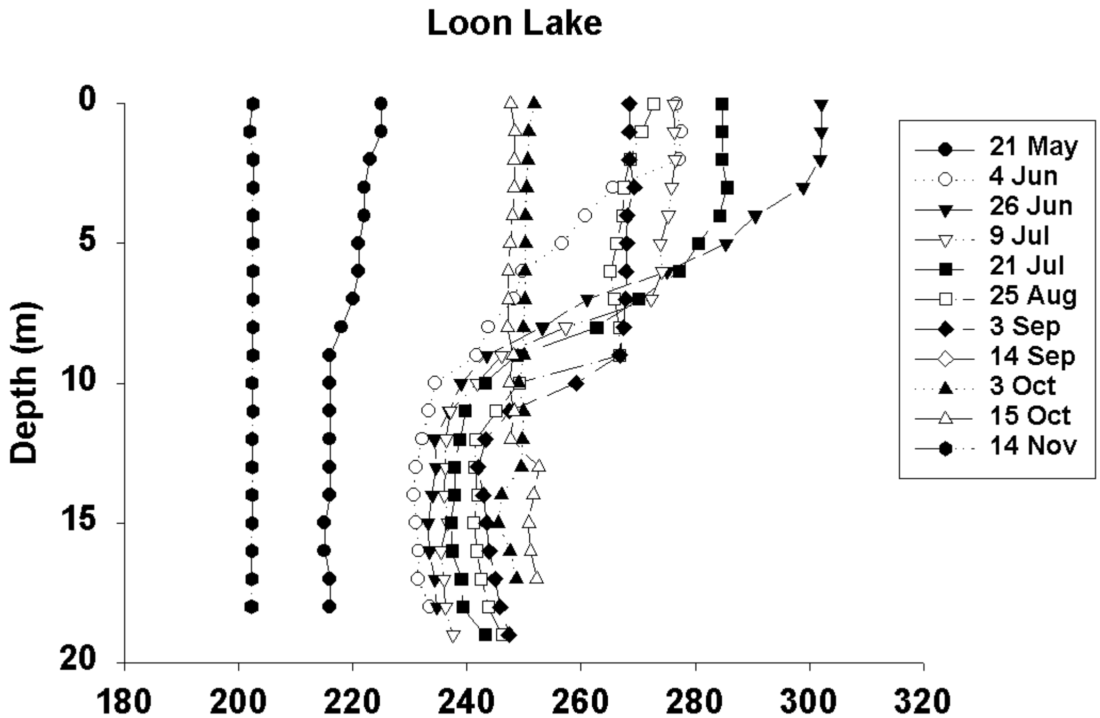


Figure 6-5. Conductivity profiles for Loon Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.



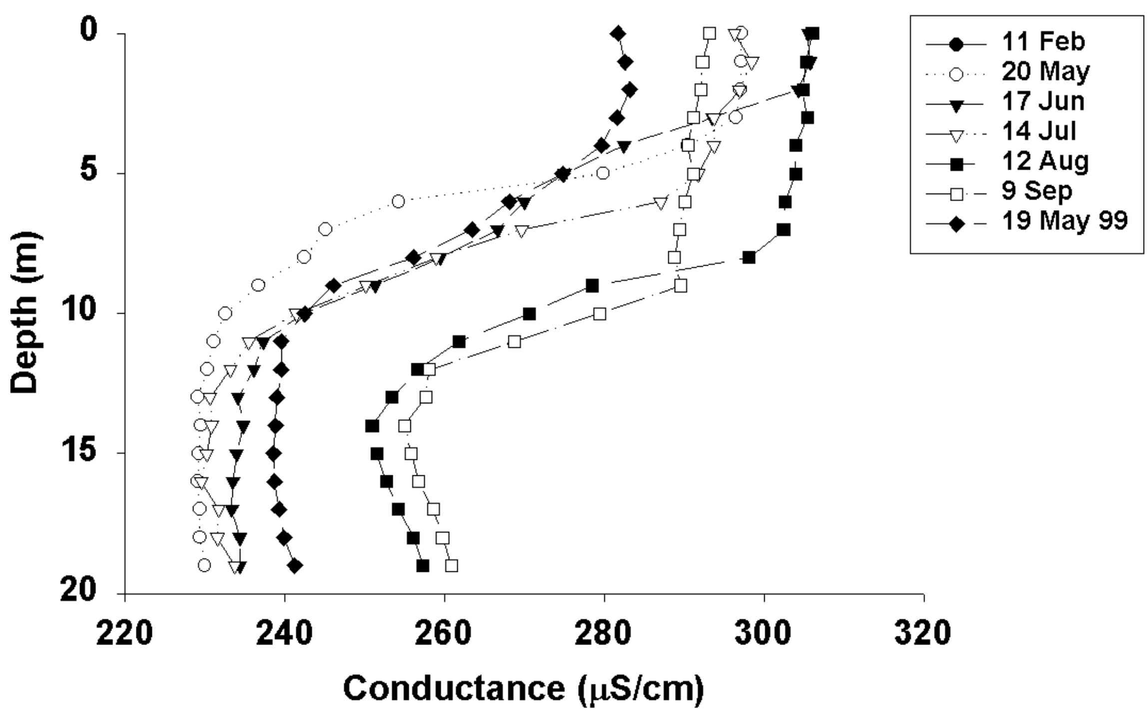


Figure 6-6. Specific conductivity profiles for Loon Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

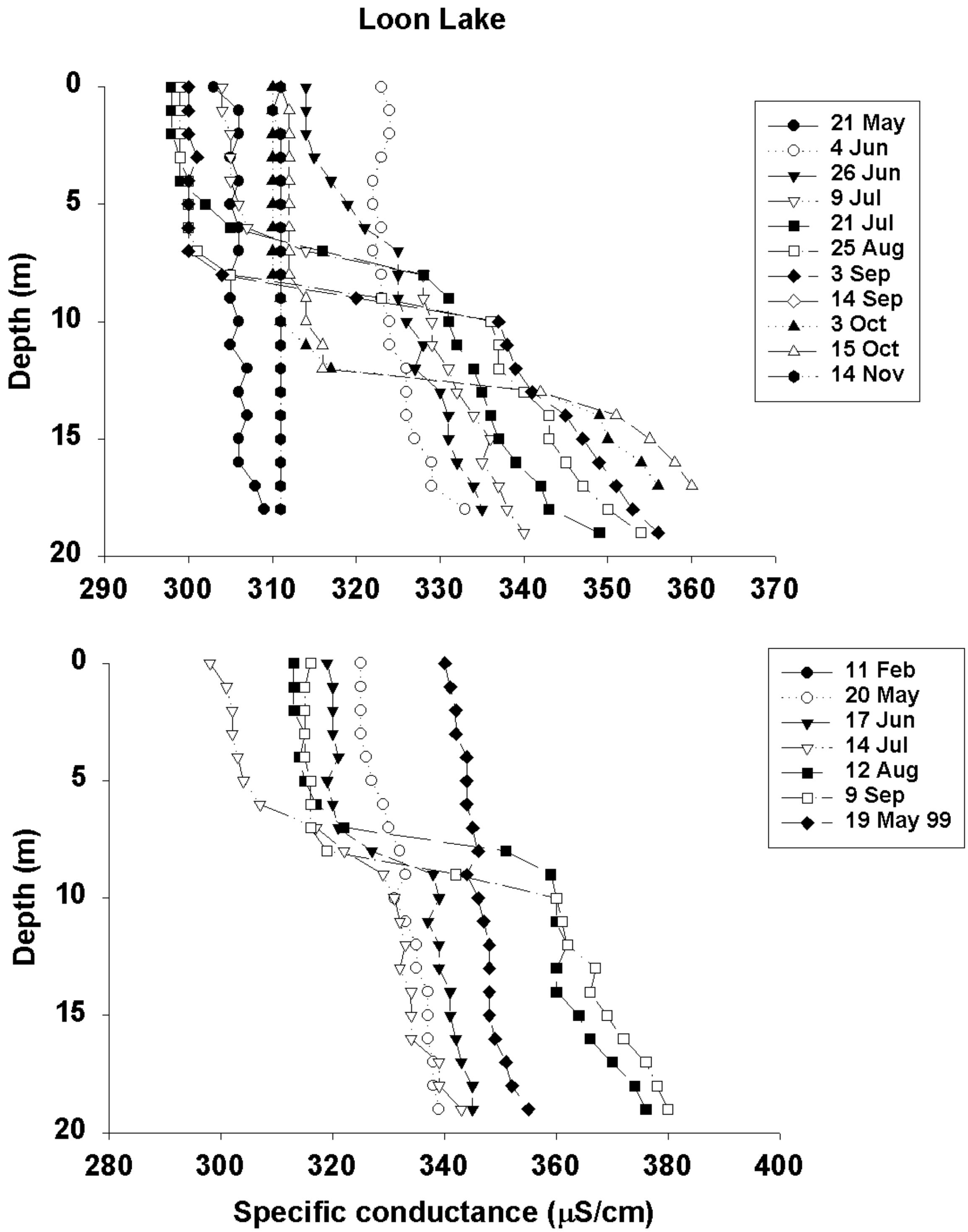


Figure 6-7. Total dissolved solids profiles for Loon Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

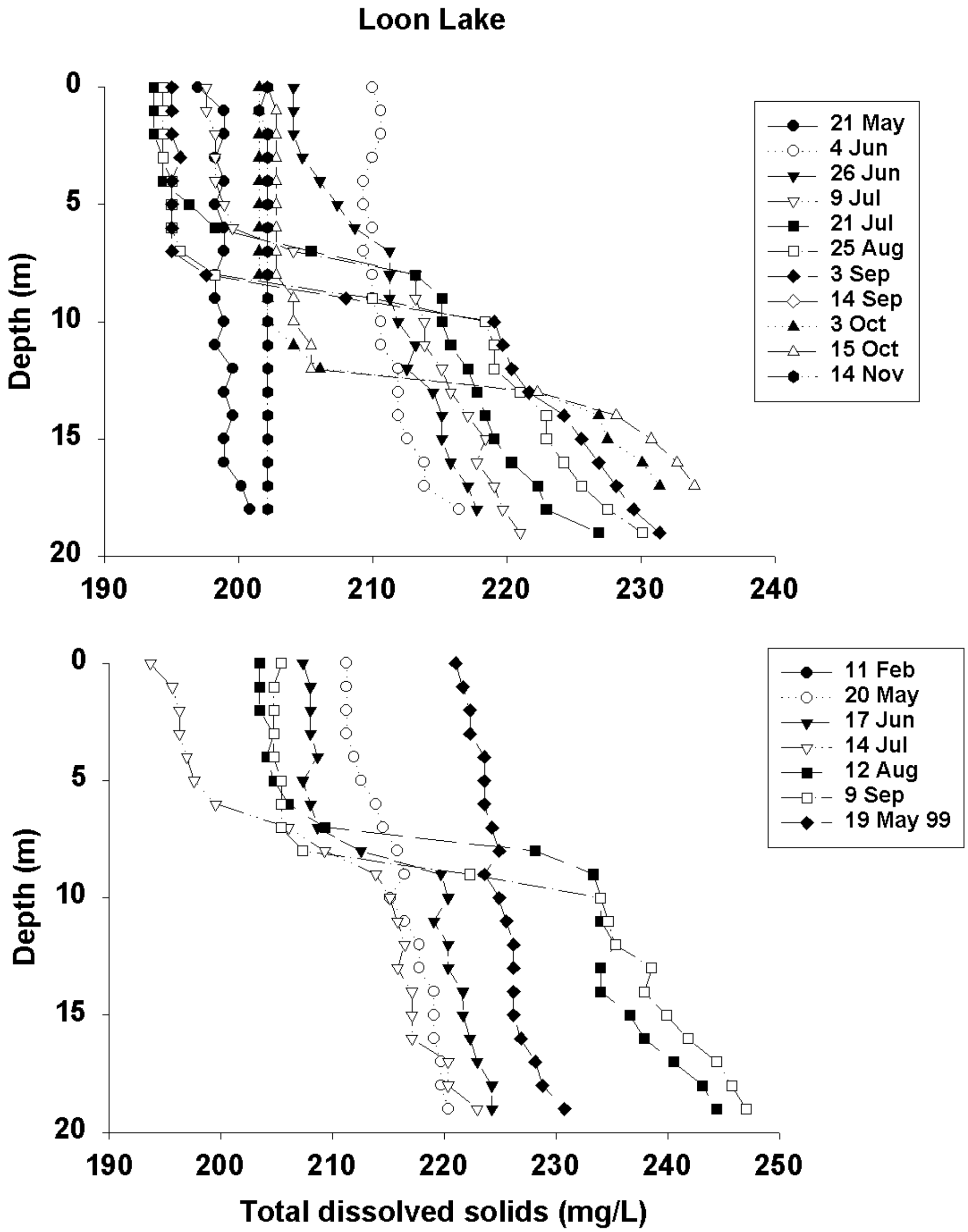


Table 6-1. Means, standard errors (SE), maximums (Max), and minimums (Min) for 1997, 1998, and for 1997-1999 in Loon Lake, Sleeping Bear Dunes National Lakeshore, Michigan. NH4 = ammonium, Ca=calcium, Cl=chloride, Mg=magnesium, N-NO3+NO2=nitrogen as nitrate+nitrite, SiO2=silica, SO4=sulfate, and P=phosphorus.

	Littoral Alkalinity mg/L	NH_4 mg/L	Ca mg/L	Cl mg/L	Color	Hardness Mg/L	Mg mg/L	$N\text{-}NO_3$+NO_2 mg/L	SiO_2 mg/L	SO_4 mg/L	Total P mg/L	Turbidity NTU
97Mean	142.40	0.05	54.90	5.69	5.63	125.89	9.70	0.08	3.20	10.18	0.04	2.86
97SE	3.82	0.00	15.44	0.21	1.57	25.08	2.81	0.03	0.80	0.29	0.02	1.46
97Max	155.00	0.06	92.60	6.20	10.00	158.41	13.00	0.16	4.00	11.10	0.13	8.60
97Min	134.00	0.03	13.80	5.10	2.50	51.34	4.10	0.05	1.60	9.60	0.01	0.70
98Mean	135.20	0.04	40.34	7.38	8.00	151.13	12.24	0.04	4.64	10.38	0.04	1.45
98SE	2.94	0.01	2.28	0.94	1.22	6.04	0.14	0.01	0.42	0.17	0.01	0.64
98Max	144.00	0.05	46.70	9.80	10.00	167.26	12.50	0.05	5.90	10.90	0.10	3.90
98Min	128.00	0.01	34.50	5.20	5.00	134.33	11.70	0.02	3.50	9.90	0.02	0.54
Mean	139.91	0.05	47.52	6.60	7.25	142.97	11.50	0.08	4.33	10.25	0.04	2.02
SE	2.57	0.00	7.01	0.49	0.95	10.82	0.94	0.02	0.46	0.15	0.01	0.72
Max	155.00	0.06	92.60	9.80	10.00	170.47	13.20	0.29	6.20	11.10	0.13	8.60
Min	128.00	0.01	13.80	5.10	2.50	51.34	4.10	0.01	1.60	9.60	0.01	0.54

	Limnetic Epilimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3$+NO_2	SiO_2	SO_4	Total P	Turbidity
97Mean	148.80	0.05	58.08	5.72	12.50	146.62	12.17	0.07	3.80	9.56	0.03	3.02
97SE	5.73	0.01	12.46	0.19	7.50	10.38	0.15	0.03	0.25	1.06	0.01	1.09
97Max	167.00	0.09	89.60	6.20	35.00	165.61	12.40	0.19	4.10	11.10	0.06	7.00
97Min	135.00	0.03	26.90	5.20	5.00	117.41	11.90	0.05	3.30	5.50	0.02	0.78
98Mean	136.60	0.05	40.52	5.78	7.00	111.97	2.22	0.06	4.64	10.36	0.03	1.41
98SE	3.44	0.00	1.98	0.16	1.22	5.46	0.58	0.01	0.45	0.16	0.00	0.70
98Max	148.00	0.05	46.00	6.20	10.00	126.80	3.10	0.12	6.10	10.80	0.03	4.20
98Min	127.00	0.05	35.80	5.40	5.00	100.36	0.00	0.05	3.60	10.00	0.02	0.54
Mean	143.00	0.05	49.05	5.86	9.50	132.57	6.99	0.07	4.36	9.95	0.03	2.07
SE	3.41	0.00	6.00	0.16	2.93	8.81	1.94	0.01	0.29	0.47	0.00	0.62
Max	167.00	0.09	89.60	7.00	35.00	179.37	15.30	0.19	6.10	11.10	0.06	7.00
Min	127.00	0.03	26.90	5.20	5.00	100.63	0.00	0.01	3.30	5.50	0.02	0.54

	Limnetic Hypolimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3$+NO_2	SiO_2	SO_4	Total P	Turbidity
97Mean	158.40	0.26	64.04	5.68	21.25	155.47	11.33	0.09	13.50	8.02	0.04	2.92
97SE	6.45	0.13	13.20	0.21	9.44	9.94	1.17	0.02	0.81	1.53	0.01	0.86
97Max	172.00	0.73	99.90	6.12	40.00	181.39	12.70	0.14	14.50	11.00	0.07	6.00
97Min	134.00	0.05	37.40	5.00	5.00	136.44	9.00	0.02	11.90	2.50	0.01	1.10
98Mean	162.00	0.18	48.58	5.88	31.00	170.23	11.88	0.04	14.44	6.78	0.03	1.88
98SE	3.10	0.13	0.77	0.16	4.85	2.49	0.17	0.01	0.79	0.64	0.01	0.14
98Max	174.00	0.72	50.50	6.20	40.00	175.51	12.30	0.05	16.70	8.30	0.04	2.20
98Min	158.00	0.05	46.00	5.30	15.00	161.40	11.30	0.02	11.80	4.50	0.01	1.50
Mean	159.45	0.21	55.80	5.88	25.50	166.51	12.14	0.07	14.01	7.56	0.03	2.25
SE	3.19	0.08	6.12	0.15	4.56	5.29	0.59	0.01	0.51	0.75	0.01	0.43
Max	174.00	0.73	99.90	6.90	40.00	192.07	15.90	0.14	16.70	11.00	0.07	6.00
Min	134.00	0.05	37.40	5.00	5.00	136.44	9.00	0.02	11.80	2.50	0.01	0.70

Table 6-2. Values from the littoral zone profiles. Means (SE) are given. Most profiles had 1-4 readings. ND=no data available, Sp. Cond.=specific conductance and Cond.=conductance.

Date	5/21/97	6/4/97	6/26/97	7/9/97	7/21/97	8/25/97
Temperature(C)	23.37 (0.04)	ND	23.37 (0.05)	20.13 (0)	22.61 (0.00)	20.28 (0.04)
DO(mg/L)	98.4 (1.2)	ND	98.40 (1.2)	97.3 (0.1)	94.8 (0.5)	92.8 (0.1)
DO(%saturation)	8.38 (0.09)	ND	8.38 (0.09)	8.825 (0.01)	8.19 (0.05)	8.39 (0.01)
pH	8.68 (0.01)	ND	8.68 (0.01)	8.675 (0.01)	8.49 (0.01)	8.78 (0.03)
TDS (mg/L)	204 (2)	ND	204 (2)	198 (0)	194 (0)	194 (1)
Sp. Cond. (μS/cm)	314 (0)	ND	314 (0)	305 (0)	299 (0)	299 (2)
Cond. (μS/cm)	304 (0)	ND	304 (0)	276 (0)	285 (0)	272 (2)

Date	9/3/97	9/14/97	10/3/97	10/15/97	11/14/97	2/11/98
Temperature(C)	19.41 (0.01)	20.20 (1.06)	14.86 (0.01)	14.13 (0.01)	ND	ND
DO(mg/L)	87.45 (0.65)	71.05 (33.15)	83.1 (0.09)	82.26 (0.48)	ND	ND
DO(%saturation)	8.0 (0.1)	6.4 (3.1)	8.3 (0.0)	8.4 (0.0)	ND	ND
pH	8.54 (0.01)	7.58 (0.92)	ND	8.41 (0.00)	8.52 (0.01)	ND
TDS (mg/L)	195 (0)	ND	201 (0)	202 (0)	ND	ND
Sp. Cond. (μS/cm)	300 (0)	ND	310 (0)	311 (0)	ND	ND
Cond. (μS/cm)	268 (0)	ND	250 (0)	248 (0)	ND	ND

Date	5/20/98	6/17/98	7/14/98	8/12/98	9/9/98	5/19/99
Temperature(C)	20.74 (0.08)	22.56 (0.07)	24.80 (0.00)	23.71 (0.01)	20.97 (0.01)	15.973 (0.01)
DO(mg/L)	104.66 (0.49)	108.26 (0.12)	106.03 (0.21)	99.53 (0.26)	91.86 (0.13)	94.8 (0.26)
DO(%saturation)	9.3 (0.0)	9.3 (0.0)	8.7 (0.0)	8.4 (0.0)	8.1 (0.0)	9.3 (0.0)
pH	8.42 (0.03)	8.56 (0.01)	8.72 (0.03)	8.46 (0)	8.45 (0.00)	8.60 (0.01)
TDS (mg/L)	211 (0)	208 (0)	198 (0)	203 (0)	206 (0)	222 (0)
Sp. Cond. (μS/cm)	325 (0)	321 (1)	305 (0)	313 (0)	318 (1)	341 (0)
Cond. (μS/cm)	298 (1)	305 (2)	304 (0)	305 (0)	293 (1)	283 (0)

North Bar Lake

North Bar Lake stratified around early June in 1997 and was already stratified in mid-May in 1998, as was evident from the clinograde curves in the temperature profiles (Figure 6-8). The positive heterograde dissolved oxygen curves indicated some elevated primary production occurring on a couple dates in June and July in both years and also in May in 1998 (Figure 6-9). The oxygen saturation curves also indicated the higher production at the epilimnion-metalimnion interface (Figure 6-10). The lake underwent fall overturn by October in 1997, and sampling did reveal a strong overturn event in 1998 because sampling continued only into September.

The water chemistry for the three zones is summarized by year in Table 6-3, and further data is given in Table 6-4. No parameters measured in the epilimnion were significantly different between 1997 and 1998 (all t-test P-values > 0.10; P-critical = 0.007 after Bonferonni adjustment). In general, within-year variation was minimal as suggested by SEM's of <20% of the means for all parameters with the exception of turbidity in 1997 (SEM 67% of mean).

The water chemistry in the hypolimnion showed similar trends as the epilimnion (Table 6-3). No parameters were significantly different between 1997 and 1998 (all t-test P-values > 0.22; P-critical = 0.007 after Bonferonni adjustment), although nitrogen as nitrate+nitrite (P = 0.009) and silica (P= 0.041) were slightly higher in 1997 than in 1998. Ammonia was detected on a few dates, but data were not sufficient for statistical testing.

Water samples from the littoral zone showed similar trends as was observed in the hypolimnion. Only silica was marginally significantly different (P = 0.017, mean 1998 > mean 1997), whereas all other parameters were not significantly different (all P-values > 0.09). Ammonia was only detected in a few samples but was consistently below detection limit in other samples, which indicates very low concentrations.

When parameters were compared between epilimnion and hypolimnion (paired tests), none were significantly different (P > 0.07). As expected there was also no significant difference between the epilimnion and littoral water samples for all parameters, although turbidity was marginally higher in the epilimnion than in the littoral (P = 0.04). Numerous parameters (mean alkalinity hypolimnion > littoral; mean chloride hypolimnion < littoral; mean magnesium hypolimnion > littoral; mean sulfate hypolimnion < littoral) were marginally different between the hypolimnion and littoral habitats (P's = 0.045-0.01). Silica (P = 0.002) and calcium (P = 0.003) were significantly higher in the hypolimnion compared to the littoral, whereas nitrogen as nitrate+nitrite was significantly higher in the littoral than the hypolimnion (P = 0.005).

Limnology

The high calcium and magnesium concentrations were associated with the similarly high alkalinity and hardness. North Bar Lake is a hardwater lake and, consequently, had high specific conductance measurements. Calcium was higher in the hypolimnion than in the littoral, but the mean difference of about 5 mg/L is likely of no biological importance.

Although nitrogen tended to be higher in North Bar Lake compared to the other lakes studied, levels were not high enough to be of concern. Nitrogen levels coupled with total phosphorus concentrations would suggest that the lake is in no danger of eutrophication. The likely limiting nutrient of the lake is phosphorus, which was rarely observed above detection limits. Ammonia was low throughout the lake and indicates a non-polluted system.

The low silica levels in the littoral are consistent with high diatom production in this zone. The production appears high enough to cause a significant difference between the hypolimnion and littoral. Diatom production in the hypolimnion would not be expected to be high and thus higher dissolved silica concentrations would be expected to be relatively high. Sulfate does not change with depth, which suggests that redox potentials were not too low in the hypolimnion. In fact, the extent of stratification did not suggest a large anoxic zone in the lake, which could be responsible for keeping redox potentials from dropping and converting the sulfates into hydrogen sulfide.

The general conclusions about North Bar Lake are that it is a mesotrophic, hardwater lake. No problem conditions were suggested by any of the chemical parameters measured.

Figure 6-8. Temperature profiles for North Bar Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999

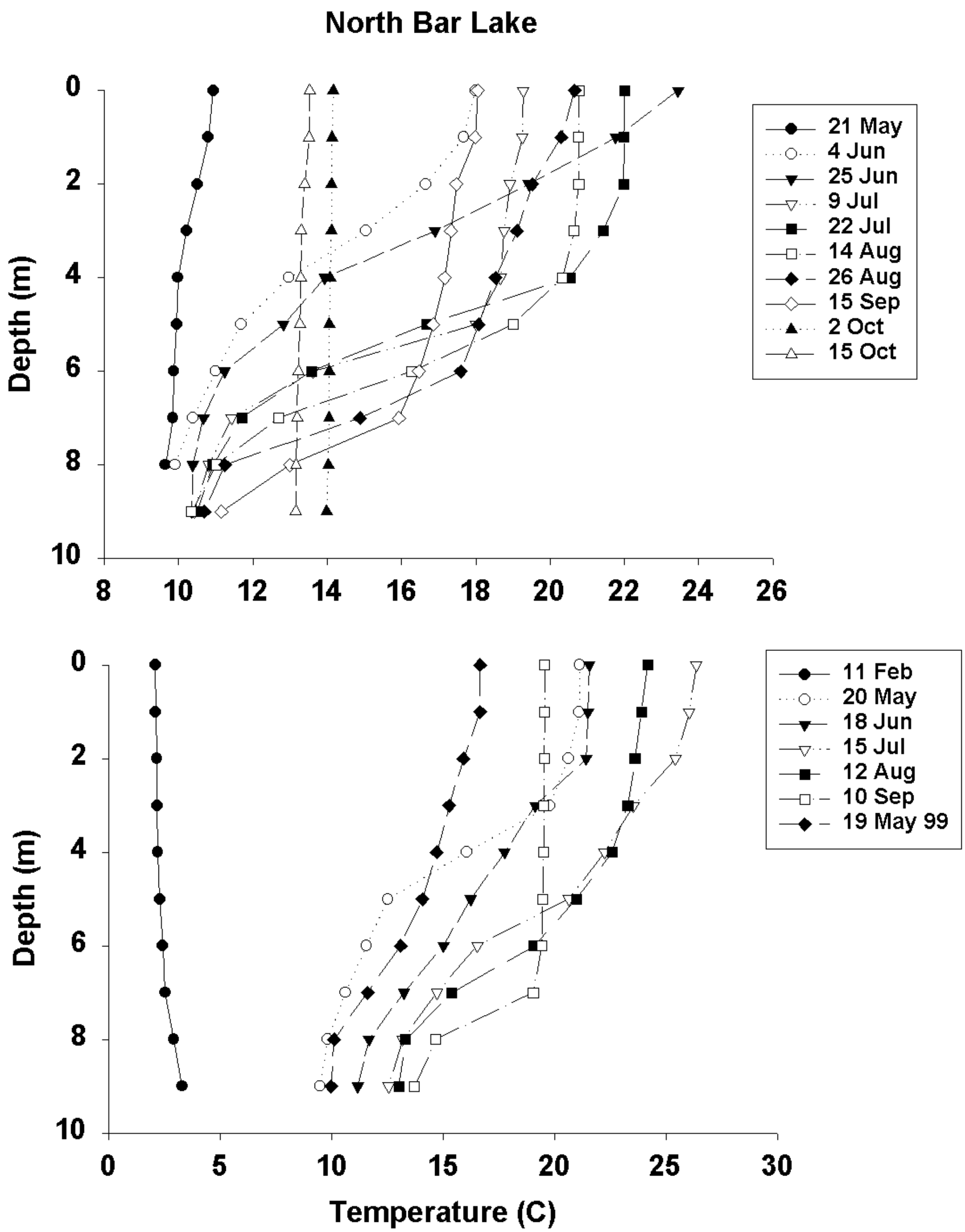


Figure 6-9. Dissolved oxygen profiles for North Bar Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999

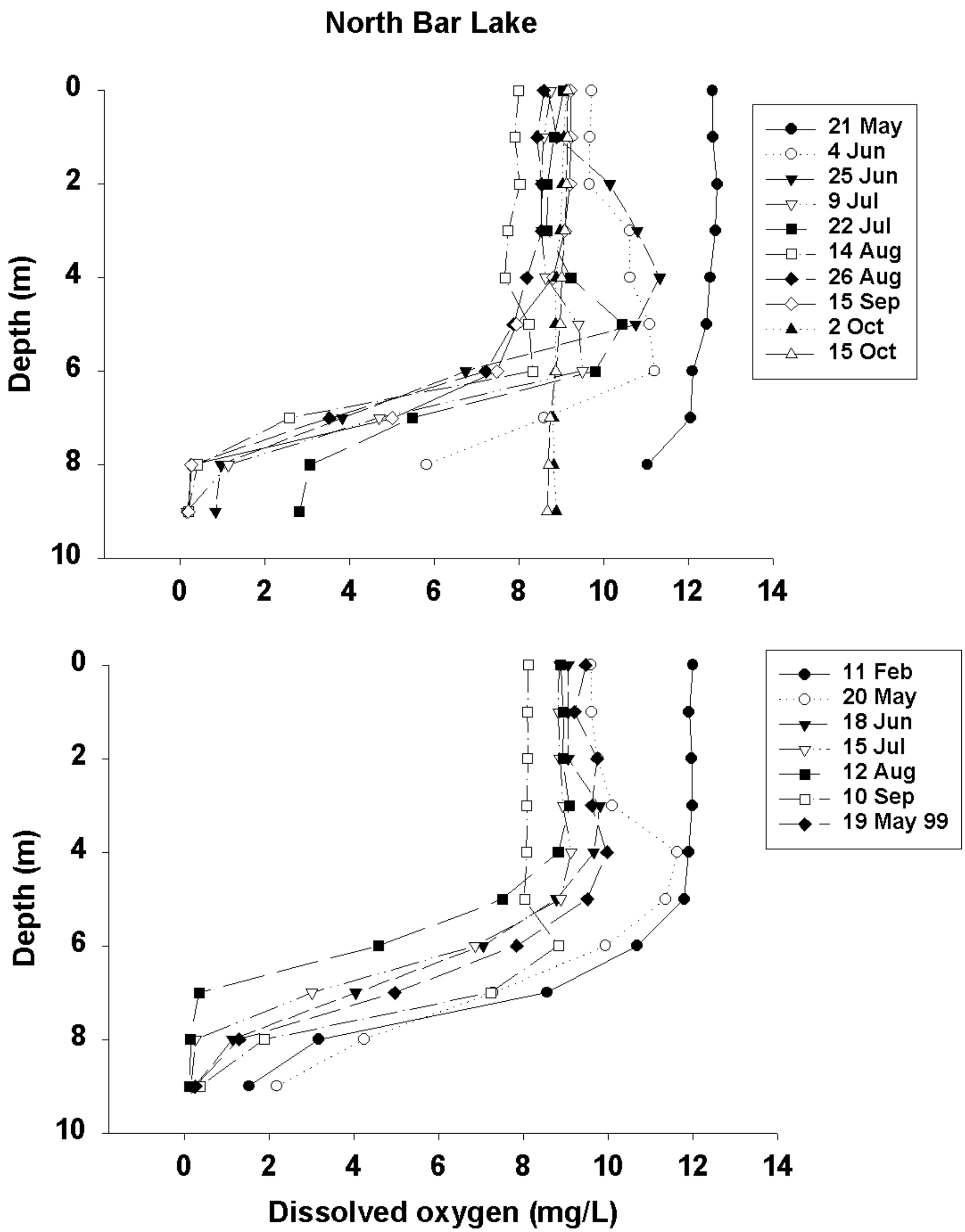


Figure 6-10. Oxygen saturation calculated from measured water temperature and dissolved oxygen in North Bar Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999

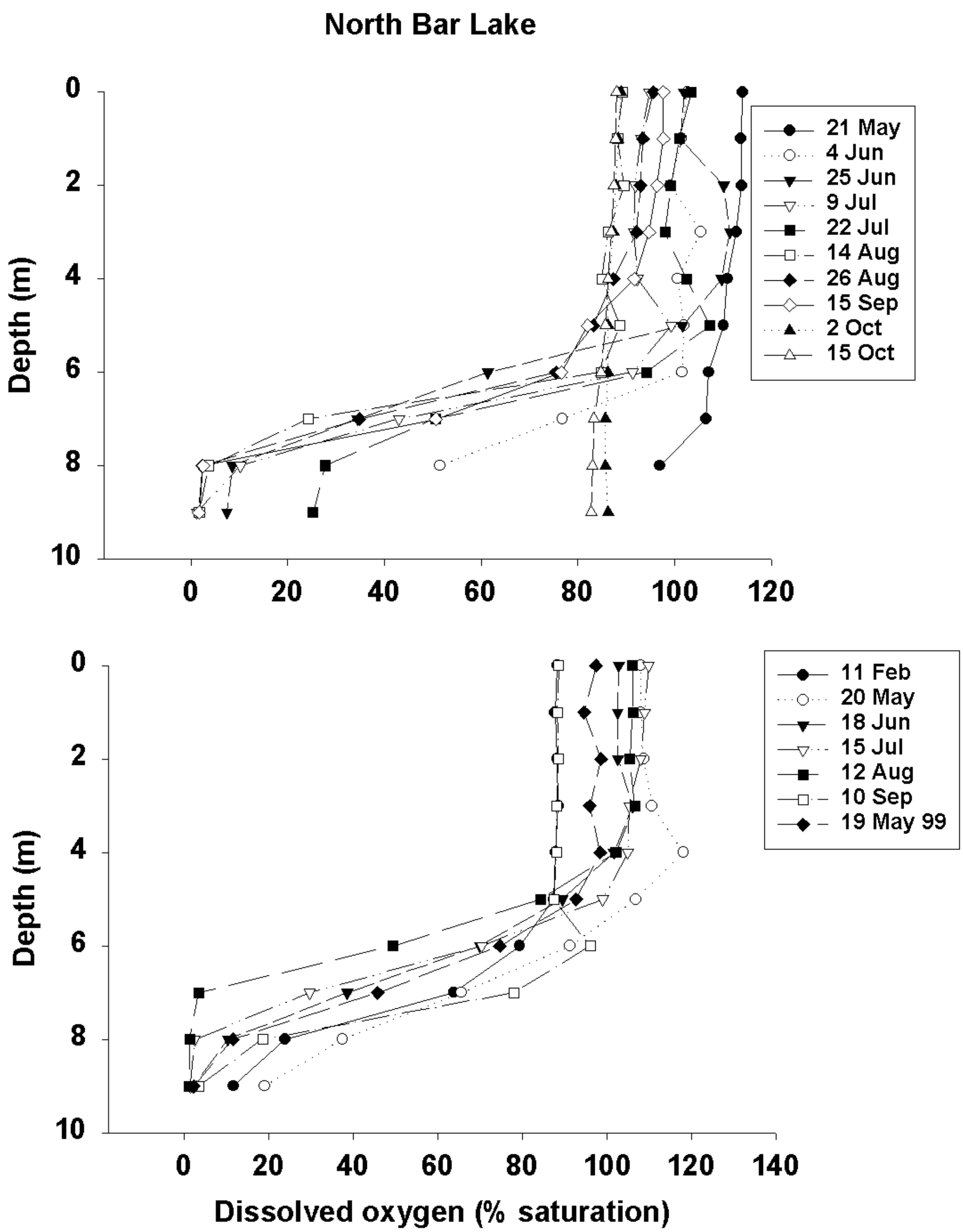


Figure 6-11. pH profiles for North Bar Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999

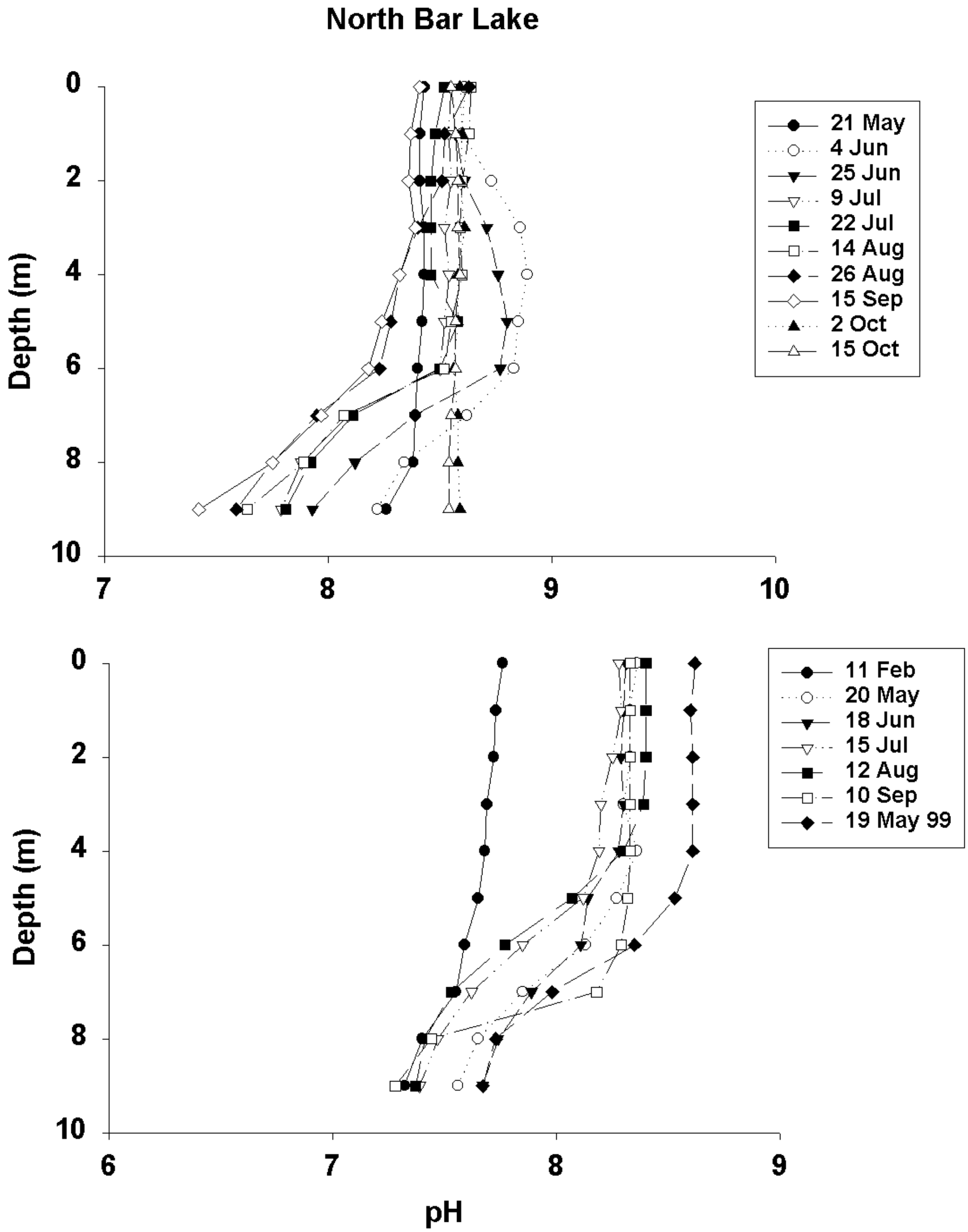


Figure 6-12. Conductivity profiles for North Bar Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999

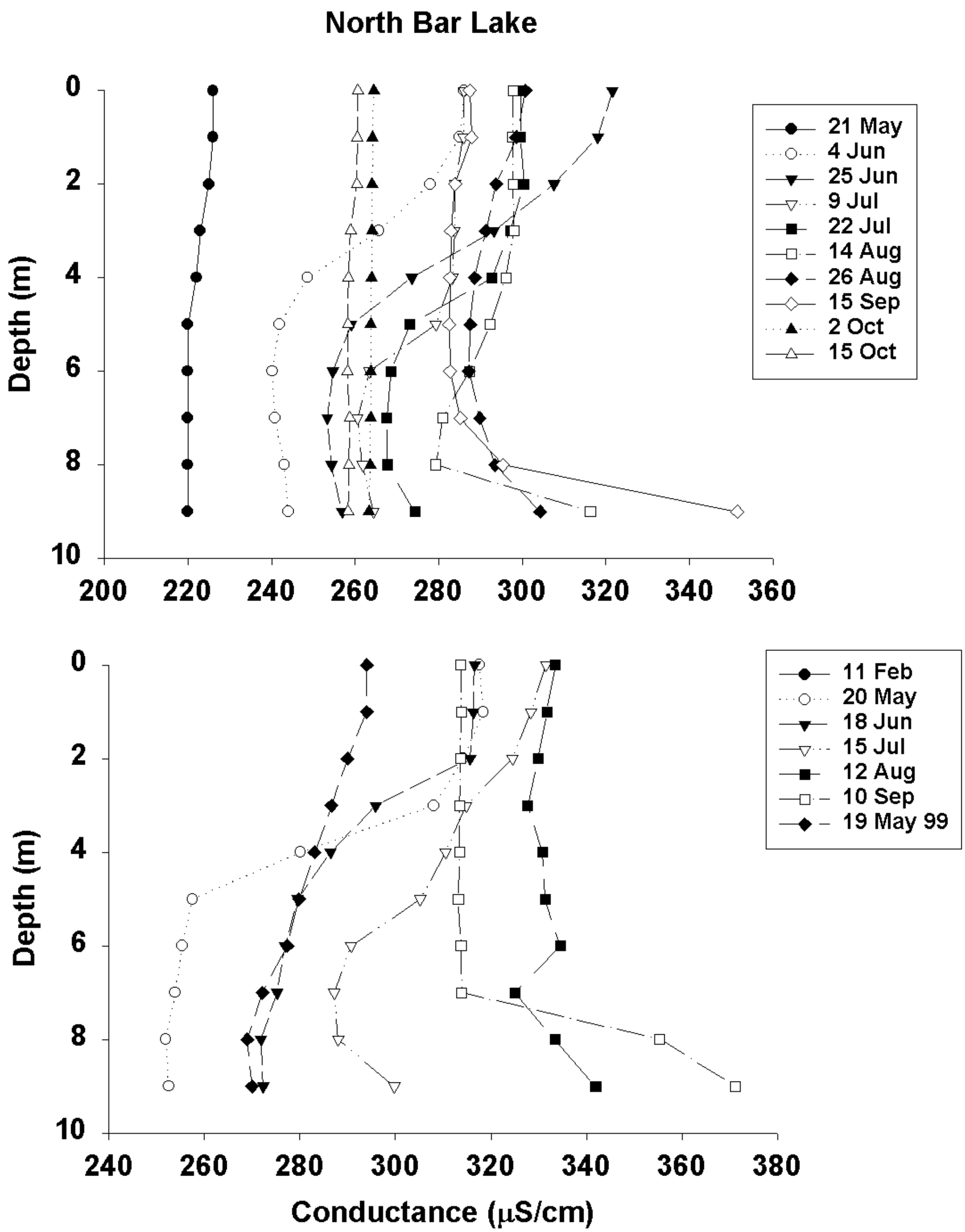


Figure 6-13. Specific conductivity profiles for North Bar Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999

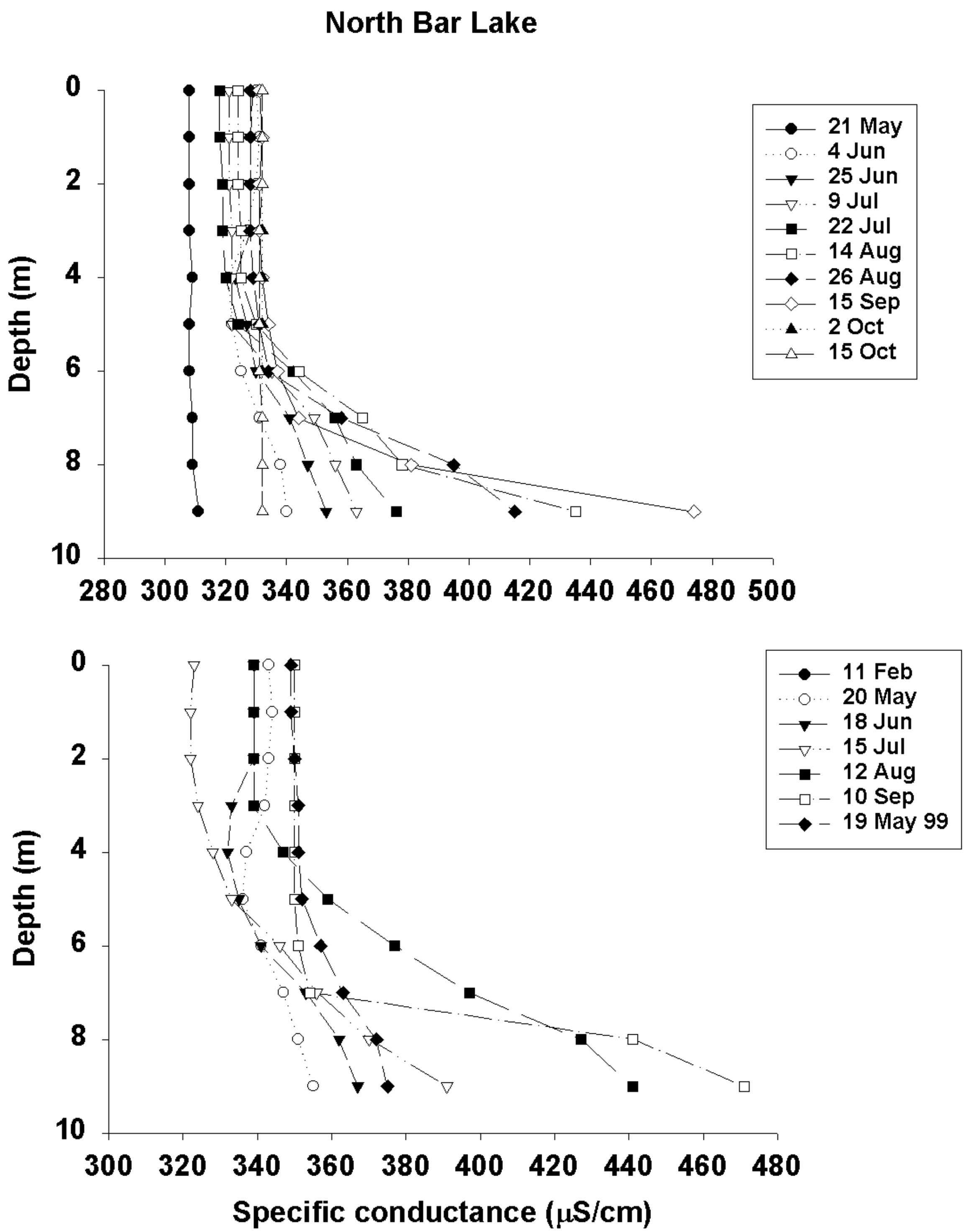


Figure 6-14. Total dissolved solids profiles for North Bar Lake, Sleeping Bear Dunes National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999

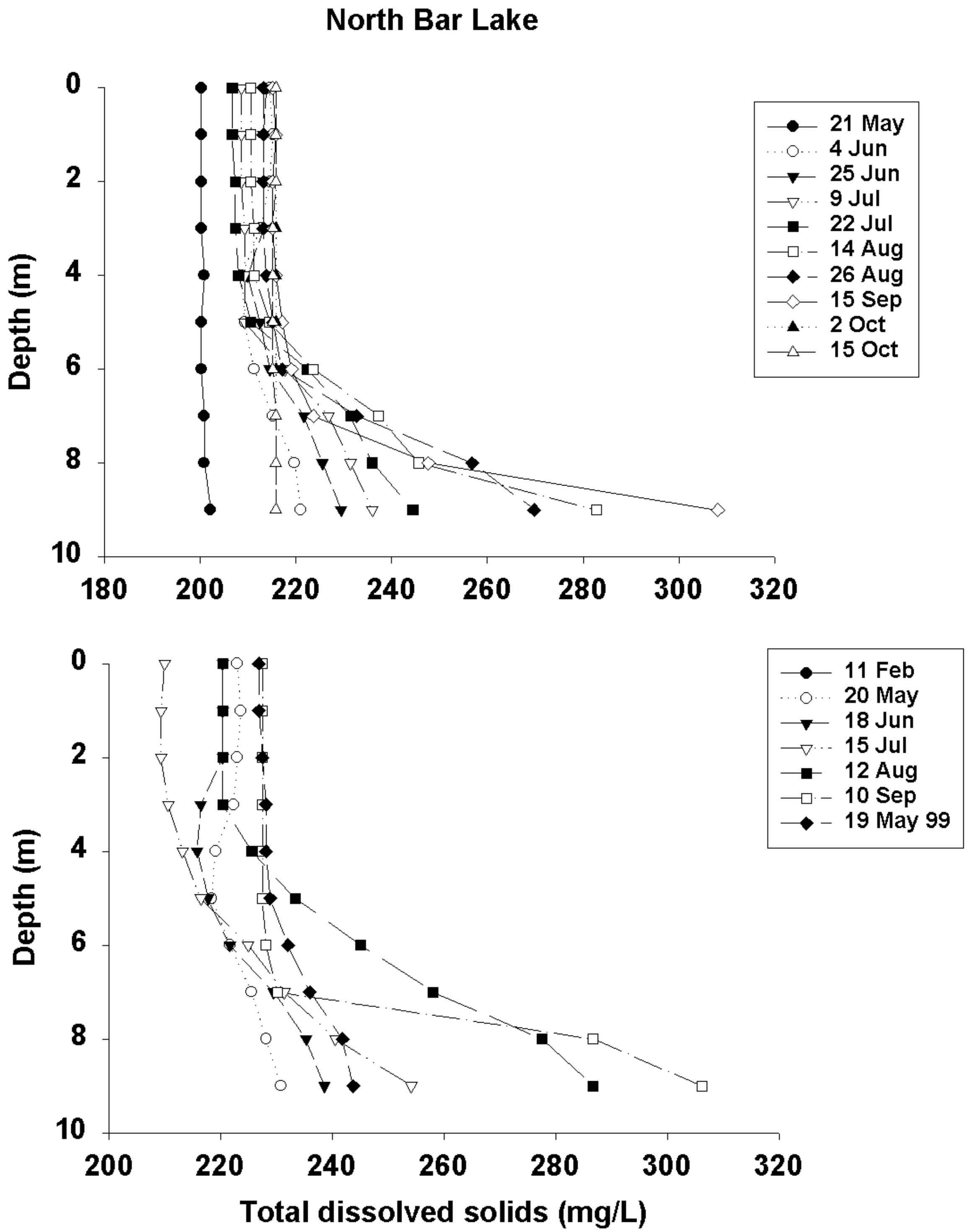


Table 6-3. Means, standard errors (SE), maximums (Max), and minimums (Min) for 1997, 1998, and for 1997-1999 in North Bar Lake, Sleeping Bear Dunes National Lakeshore, Michigan. NH4 = ammonium, Ca=calcium, Cl=chloride, Mg=magnesium, N-NO3+NO2=nitrogen as nitrate, SiO_2=silica, SO_4=sulfate, and P=phosphorus.

	Littoral Alkalinity mg/L	NH_4 mg/L	Ca mg/L	Cl mg/L	Color	Hardness mg/L	Mg mg/L	$N\text{-}NO_3{+}NO_2$ mg/L	SiO_2 mg/L	SO_4 mg/L	Total P mg/L	Turbidity NTU
97Mean	154.50	0.06	52.98	4.48	8.13	157.11	13.30	0.31	2.63	10.38	0.02	2.81
97SE	1.44	0.01	13.58	0.19	1.88	9.82	1.46	0.05	0.27	0.52	0.00	1.76
97Max	158.00	0.08	93.30	4.90	10.00	169.64	15.00	0.45	2.90	11.40	0.03	8.00
97Min	151.00	0.05	34.10	4.00	2.50	127.97	10.40	0.23	2.10	9.30	0.01	0.56
98Mean	148.80	0.17	44.88	5.34	4.50	172.68	14.72	0.22	5.52	11.56	0.03	0.97
98SE	3.34	0.10	1.41	0.47	0.50	3.56	0.22	0.03	0.65	0.36	0.00	0.21
98Max	159.00	0.57	48.80	7.00	5.00	181.98	15.60	0.30	7.90	12.60	0.03	1.50
98Min	141.00	0.02	41.70	4.10	2.50	163.84	14.40	0.16	4.00	10.60	0.02	0.46
Mean	152.00	0.11	48.35	4.96	6.00	167.31	14.32	0.26	4.47	10.87	0.02	1.69
SE	2.00	0.05	5.17	0.27	0.93	4.91	0.51	0.03	0.58	0.35	0.00	0.72
Max	159.00	0.57	93.30	7.00	10.00	181.98	15.60	0.45	7.90	12.60	0.03	8.00
Min	141.00	0.02	34.10	4.00	2.50	127.97	10.40	0.16	2.10	9.30	0.01	0.46

	Limnetic Epilimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3{+}NO_2$	SiO_2	SO_4	Total P	Turbidity
97Mean	161.50	0.05	57.48	4.73	11.25	167.89	14.10	0.19	7.00	10.60	0.04	3.79
97SE	7.42	0.00	13.16	0.66	3.15	7.66	0.85	0.05	4.27	1.97	0.01	2.54
97Max	183.00	0.05	96.40	6.60	20.00	188.12	15.00	0.31	15.50	15.70	0.08	8.80
97Min	149.00	0.05	40.10	3.60	5.00	151.19	12.40	0.05	2.10	6.20	0.02	0.57
98Mean	147.60	0.05	44.58	4.88	7.00	171.77	14.68	0.17	5.48	11.58	0.02	1.06
98SE	2.77	0.00	1.46	0.21	1.22	3.72	0.21	0.04	0.65	0.35	0.00	0.24
98Max	154.00	0.05	48.70	5.40	10.00	181.73	15.50	0.30	7.90	12.60	0.03	1.60
98Min	141.00	0.03	41.10	4.20	5.00	161.51	14.30	0.05	4.00	10.70	0.01	0.46
Mean	154.20	0.05	49.91	4.79	8.50	169.95	14.30	0.20	6.04	10.97	0.03	1.95
SE	3.74	0.00	5.28	0.26	1.50	3.36	0.33	0.03	1.30	0.77	0.01	0.87
Max	183.00	0.05	96.40	6.60	20.00	188.12	15.50	0.34	15.50	15.70	0.08	8.80
Min	141.00	0.03	40.10	3.60	5.00	151.19	12.40	0.05	2.10	6.20	0.01	0.46

	Limnetic Hypolimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3{+}NO_2$	SiO_2	SO_4	Total P	Turbidity
97Mean	167.25	0.15	58.13	4.45	10.00	168.37	13.57	0.23	7.63	9.98	0.03	1.94
97SE	6.56	0.07	14.99	0.45	2.04	12.54	1.44	0.06	3.68	1.21	0.00	0.89
97Max	181.00	0.33	102.00	5.70	15.00	193.60	15.30	0.35	14.90	13.40	0.04	4.50
97Min	151.00	0.05	36.50	3.70	5.00	135.20	10.70	0.08	3.00	7.70	0.02	0.54
98Mean	174.80	0.42	52.52	4.38	21.00	192.99	15.02	0.05	17.22	8.48	0.03	2.05
98SE	8.65	0.32	1.06	0.18	5.10	3.16	0.16	0.01	1.91	0.80	0.01	0.57
98Max	207.00	1.70	56.10	4.80	35.00	203.91	15.50	0.10	23.30	10.10	0.04	3.60
98Min	157.00	0.05	49.60	3.90	10.00	185.21	14.70	0.02	11.70	5.60	0.02	0.36
Mean	171.10	0.27	54.27	4.42	15.50	181.08	14.31	0.12	13.23	9.05	0.03	2.04
SE	4.89	0.16	5.62	0.19	3.11	6.25	0.51	0.04	2.17	0.63	0.00	0.42
Max	207.00	1.70	102.00	5.70	35.00	203.91	15.50	0.35	23.30	13.40	0.04	4.50
Min	151.00	0.05	36.50	3.70	5.00	135.20	10.70	0.02	3.00	5.60	0.02	0.36

Table 6-4. Values from the littoral zone profiles in North Bar Lake. Means (SE) are given. Most profiles had 1-4 readings. ND=no data available, Sp. Cond.=specific conductance and Cond.=conductance.

Date	5/21/97	6/4/97	6/25/97	7/9/97	7/22/97	8/14/97
Temperature(C)	ND	ND	25.19 (0.14)	19.29 (0.05)	21.75 (0.06)	20.72 (0)
DO(mg/L)	ND	ND	8.44 (0.19)	8.53 (0.17)	8.87 (0.11)	8.20 (0)
DO(%saturation)	ND	ND	102.5 (2.2)	92.5 (2.0)	101.1 (1.4)	91.4 (0)
pH	ND	ND	8.52 (0.05)	8.52 (0.01)	8.38 (0.00)	8.74 (0)
TDS (mg/L)	ND	ND	212 (1)	208 (0)	208 (0)	210 (0)
Sp. Cond. (μS/cm)	ND	ND	327 (1)	320 (0)	321 (0)	324 (0)
Cond. (μS/cm)	ND	ND	328 (2)	285 (1)	301 (0)	297 (0)

Date	8/26/97	9/15/97	10/2/97	10/15/97	2/11/98	5/20/98
Temperature(C)	19.58 (0.21)	17.83 (0.03)	14.39 (0.03)	13.63 (0.06)	ND	21.04 (0.16)
DO(mg/L)	8.5 (0.04)	9.13 (0.02)	9.04 (0.01)	9.42 (0.03)	ND	9.63 (0.02)
DO(%saturation)	92.6 (0.2)	96.2 (0.3)	88.6 (0.08)	90.8 (0.2)	ND	108.2 (0.6)
pH	8.31 (0.01)	8.43 (0.01)	8.57 (0.00)	8.53 (0.01)	ND	8.35 (0.02)
TDS (mg/L)	212 (0)	216 (0)	215 (0)	215 (0)	ND	223 (0)
Sp. Cond. (μS/cm)	327 (0)	332 (0)	332 (0)	331 (0)	ND	343 (0)
Cond. (μS/cm)	293 (1)	287 (0)	265 (0)	260 (0)	ND	317 (1)

Date	6/18/98	7/15/98	8/12/98	9/10/98	5/19/99
Temperature(C)	21.28)	25.785 (0.29)	23.91 (0.26)	19.49 (0.01)	16.64 (0.09)
DO(mg/L)	9.12 (0.02)	9.13 (0.07)	9.18 (0.09)	8.16 (0.01)	9.21 (0.16)
DO(%saturation)	103.0 (0.5)	112.3 (0.5)	109.1 (0.6)	88.9 (0.1)	94.7 (1.8)
pH	8.35 (0.00)	8.33 (0.01)	8.42 (0.01)	8.38 (0.00)	8.65 (0.01)
TDS (mg/L)	221 (0)	211 (1)	225 (1)	227 (0)	225 (0)
Sp. Cond. (μS/cm)	340 (0)	324 (1)	346 (1)	350 (0)	347 (0)
Cond. (μS/cm)	316 (1)	329 (2)	339 (1)	313 (0)	292 (1)

Round Lake

Both clinograde temperature (Figure 6-15) and dissolved oxygen (Figure 6-16) curves indicated that Round Lake was stratified by May in both 1997 and 1998. Overturn occurred by September. Both the dissolved oxygen (Figs. 6-16) and oxygen saturation (Figure 6-17) curves suggest some primary production in the water column, corresponding to the metalimnion area, around the June sampling.

No between-year comparisons could be made because chemistry data was collected from Round Lake only in 1998 (see Tables 6-5 and 6-6).

When parameters were compared between epilimnion and hypolimnion (paired tests), all parameters showed no significant differences (all P-values > 0.17). As expected there was also no significant difference between the epilimnion and littoral water samples for all parameters. Turbidity was only marginally higher in the littoral than in the hypolimnion (P = 0.05).

Limnology

Round Lake is an alkaline, hardwater lake. Calcium and magnesium, which were some of the highest measured in this study, contribute to the hardness. Chloride ion concentration was also the highest of the lakes studied. Concentrations were about 10 mg/L above the average freshwater concentration but were still not high enough to cause alarm. Specific conductance was high, which reflects the presence of high cation concentrations.

Nutrient concentrations were low throughout the lake. Nitrate, nitrite, ammonia, and total phosphorus were mostly below detection limits. Thus, it is likely that the lake is not in a eutrophic state, but more likely is oligotrophic to slightly mesotrophic.

Round Lake does stratify during summer, although oxygen is present throughout most of the summer even at the deeper zones. Silica concentrations remained the same through the water column indicating that diatom production was evenly distributed or that production was uniformly low. Despite stratification, sulfates remained the same in the hypolimnion as in the overlying water. This would be the case if the oxygen concentration in the hypolimnion was still sufficient to keep the redox potential high, which in turn would prevent sulfates from being converted to hydrogen sulfides.

In general, Round Lake is in good condition based on the water chemistry data.

Figure 6-15. Temperature profiles for Round Lake, Sleeping Bear Dunes National Lakeshore, Michigan, 1998/1999.

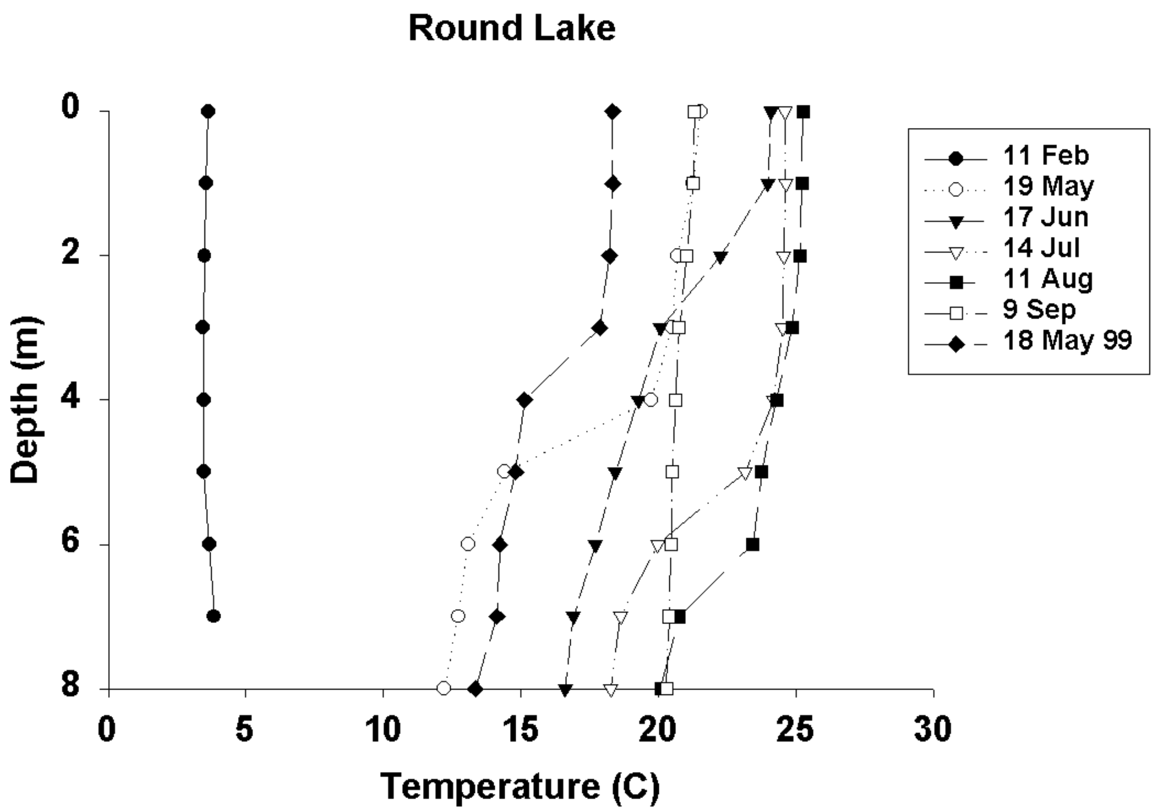


Figure 6-16. Dissolved oxygen profiles for Round Lake, Sleeping Bear Dunes National Lakeshore, Michigan, 1998/1999.

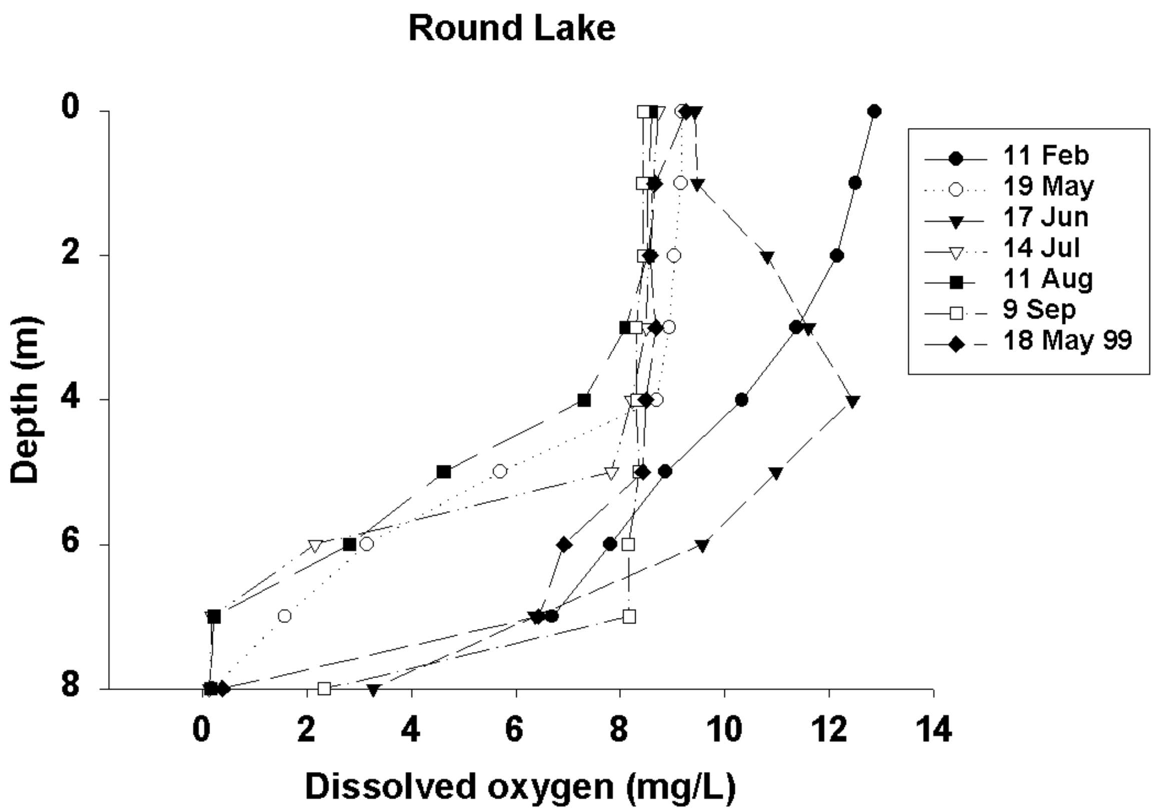


Figure 6-17. Oxygen saturation calculated from measured water temperature and dissolved oxygen in Round Lake, Sleeping Bear Dunes National Lakeshore, Michigan, 1998/1999.

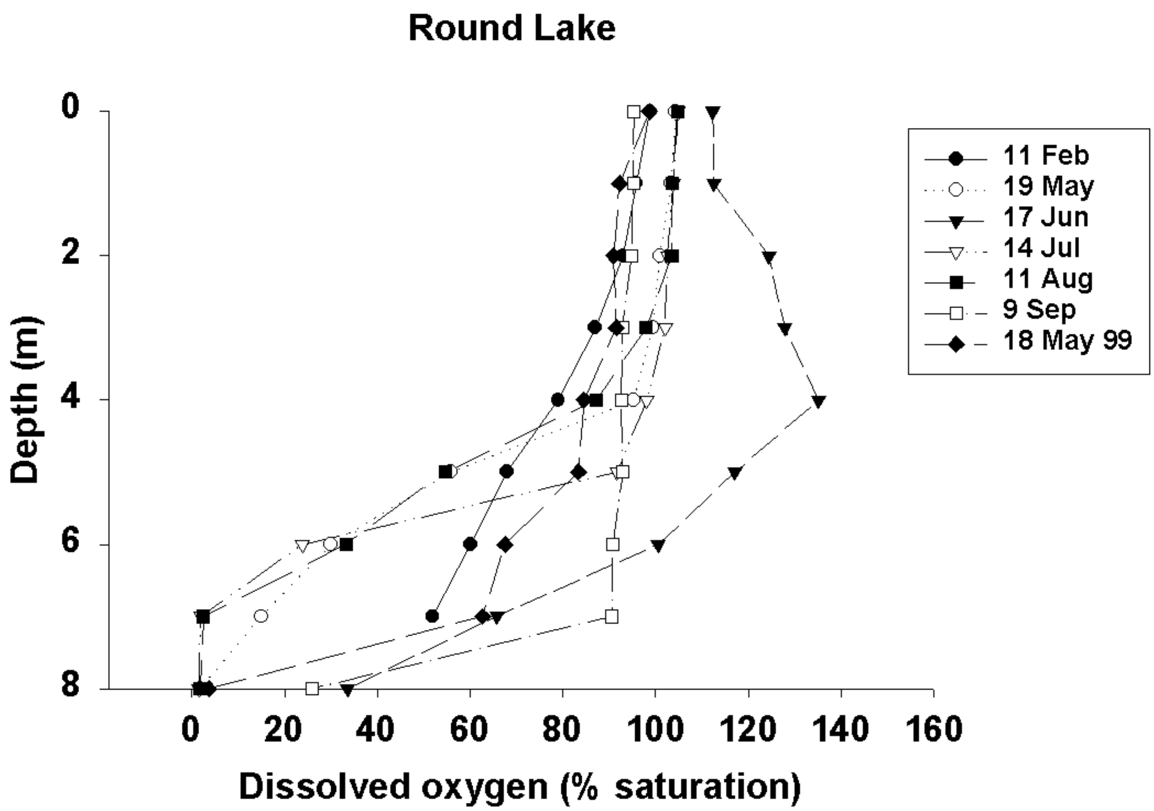


Figure 6-18. pH profiles for Round Lake, Sleeping Bear Dunes National Lakeshore, Michigan, 1998/1999.

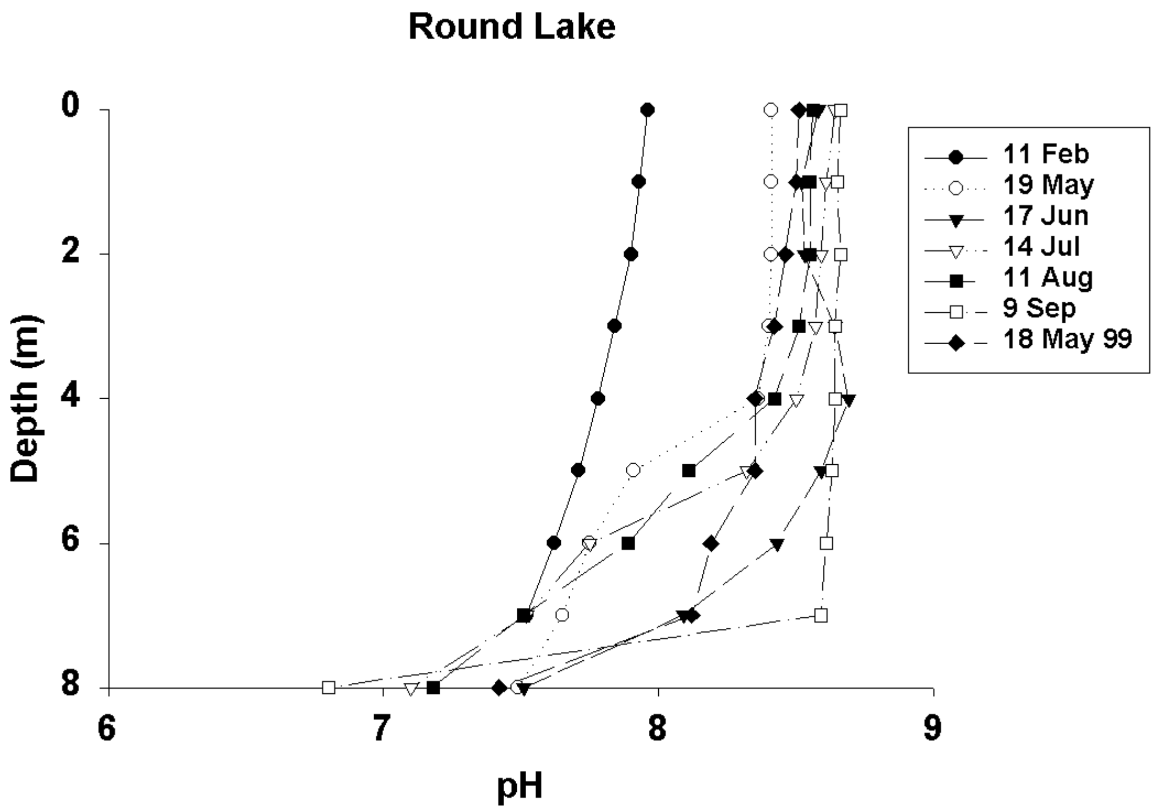


Figure 6-19. Conductivity profiles for Round Lake, Sleeping Bear Dunes National Lakeshore, Michigan, 1998/1999.

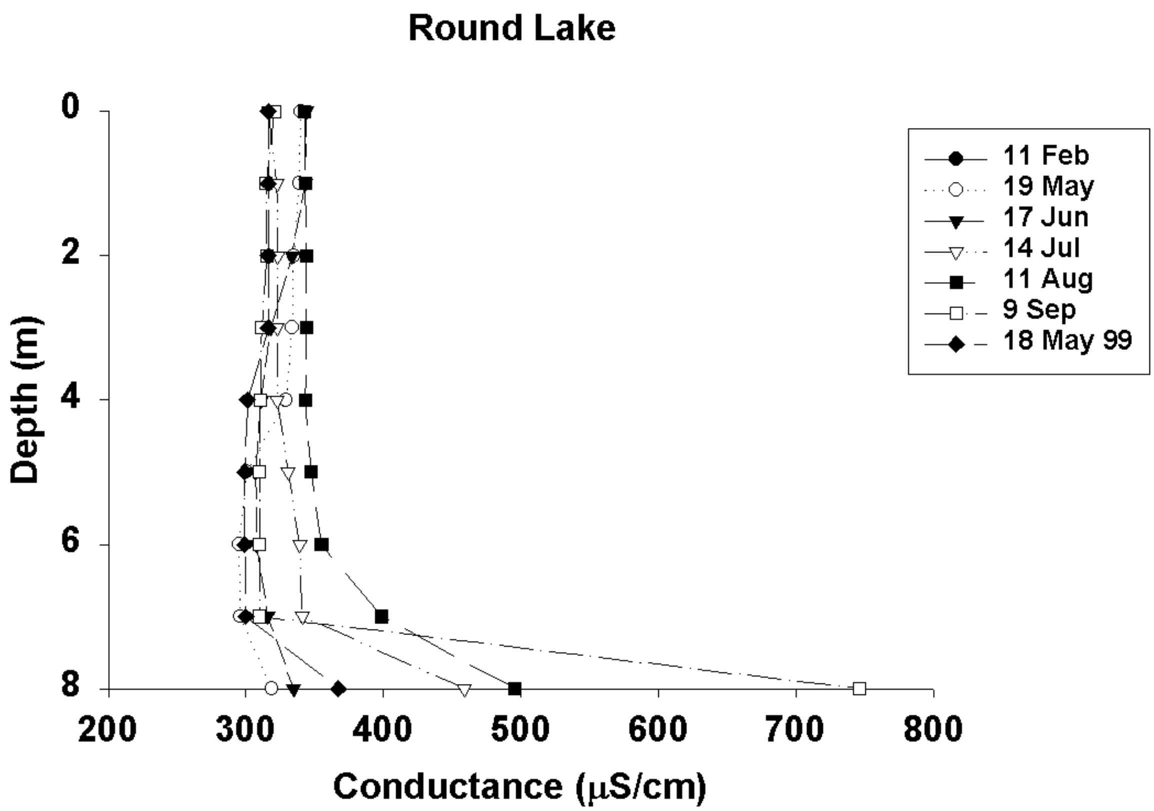


Figure 6-20. Specific conductivity profiles for Round Lake, Sleeping Bear Dunes National Lakeshore, Michigan, 1998/1999.

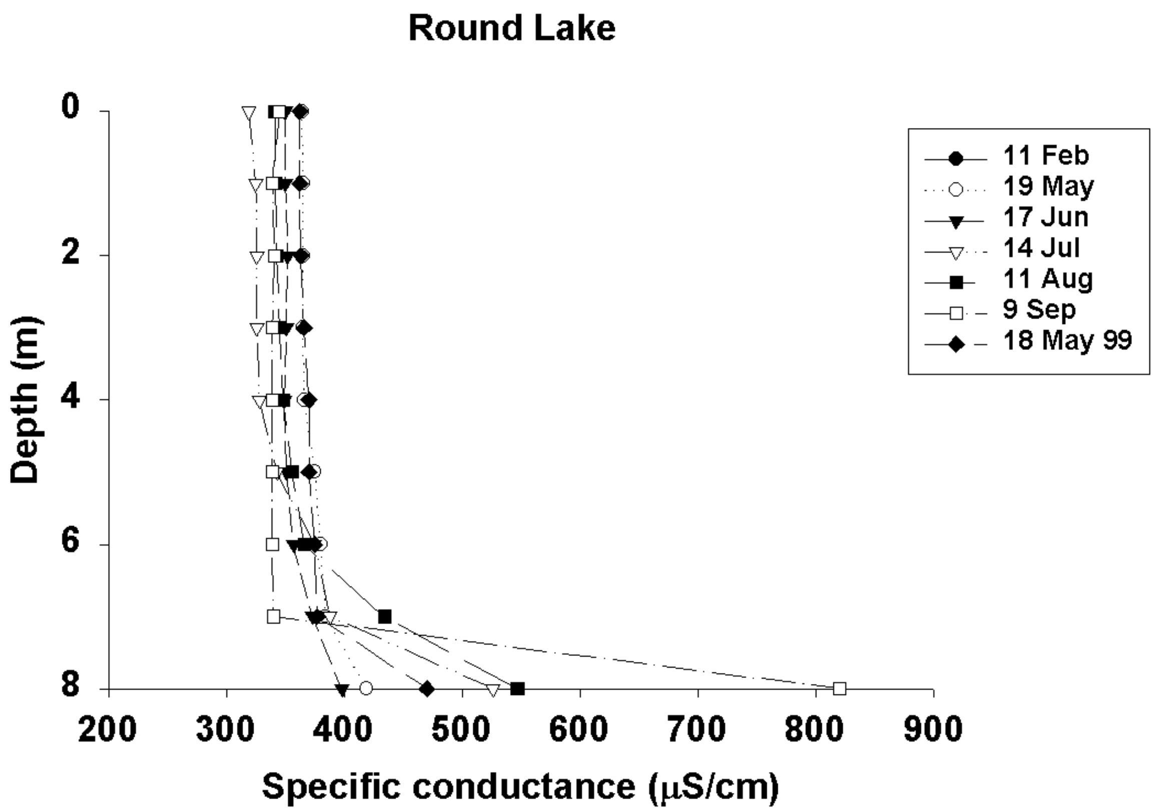


Figure 6-21. Total dissolved solids profiles for Round Lake, Sleeping Bear Dunes National Lakeshore, Michigan, 1998/1999.

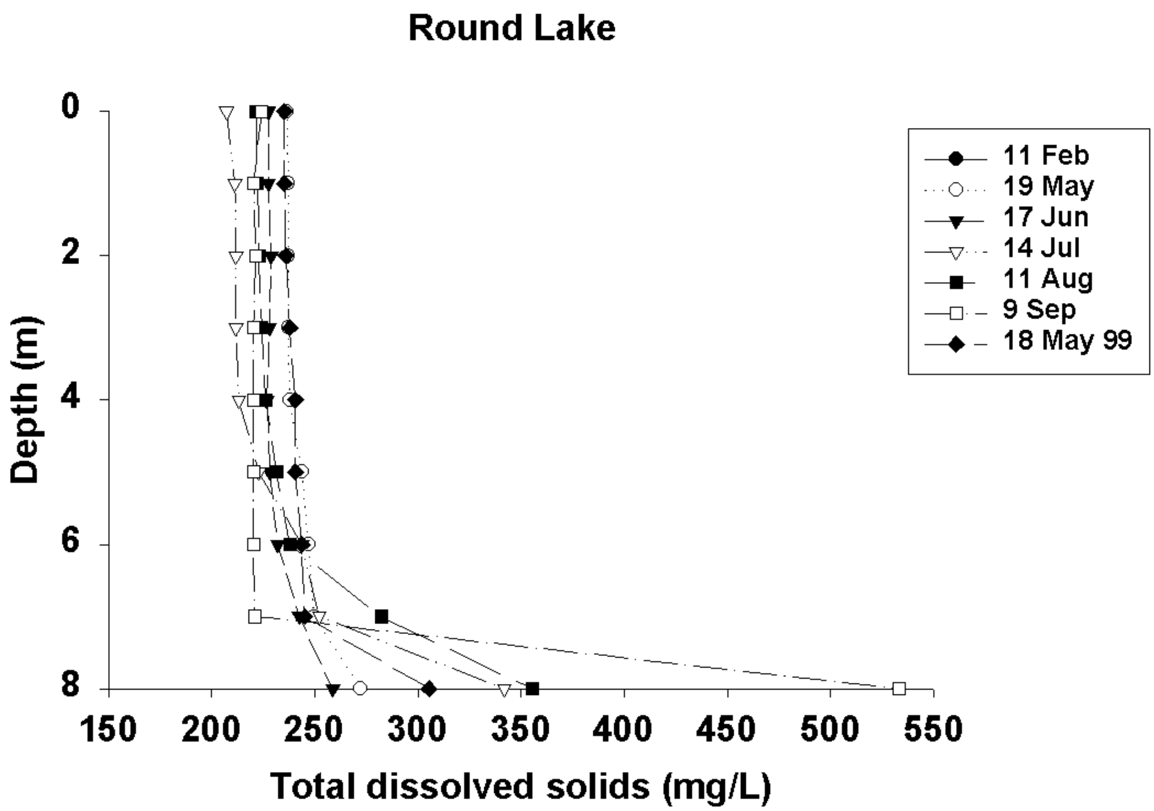


Table 6-5. Means, standard errors (SE), maximums (Max), and minimums (Min) for 1998, and for 1998-1999 in Round Lake, Sleeping Bear Dunes National Lakeshore, Michigan. NH4 = ammonium, Ca=calcium, Cl=chloride, Mg=magnesium, N-NO3+NO2=nitrogen as nitrate+nitrite, SiO_2=silica, SO_4=sulfate, and P=phosphorus.

	Littoral Alkalinity mg/L	NH_4 mg/L	Ca mg/L	Cl mg/L	Color	Hardness mg/L	Mg mg/L	$N\text{-}NO_3+NO_2$ mg/L	SiO_2 Mg/L	SO_4 mg/L	Total P mg/L	Turbidity NTU
98Mean	131.00	0.07	32.22	17.44	10	147.08	16.18	0.04	7.38	10.72	0.06	0.75
98SE	5.19	0.03	2.76	0.46	0.00	6.86	0.12	0.01	1.16	0.22	0.03	0.09
98Max	148.00	0.17	40.20	18.70	10.00	165.86	16.40	0.05	10.00	11.20	0.18	1.00
98Min	121.00	0.02	25.30	16.30	10.00	128.65	15.90	0.02	4.10	10.00	0.00	0.56
Mean	132.17	0.06	32.85	17.67	10.00	147.08	16.28	0.05	6.57	10.55	0.05	0.69
SE	4.39	0.02	2.34	0.44	0.00	6.89	0.14	0.00	1.25	0.25	0.03	0.09
Max	148.00	0.17	40.20	18.80	10.00	165.86	16.80	0.05	10.00	11.20	0.18	1.00
Min	121.00	0.02	25.30	16.30	10.00	128.65	15.90	0.02	2.50	9.70	0.00	0.41

	Limnetic Epilimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3+NO_2$	SiO_2	SO_4	Total P	Turbidity
98Mean	131.80	0.05	32.66	17.00	10.00	149.42	16.48	0.04	7.74	10.60	0.06	0.69
98SE	4.60	0.00	2.25	0.34	0.00	4.48	0.30	0.01	1.37	0.28	0.04	0.09
98Max	147.00	0.05	39.20	17.90	10.00	161.71	17.30	0.05	10.80	11.40	0.21	0.97
98Min	124.00	0.04	27.40	15.90	10.00	139.66	15.50	0.02	4.00	10.00	0.00	0.45
Mean	133.83	0.05	33.10	17.28	10.00	149.42	16.47	0.04	6.85	10.45	0.05	0.64
SE	4.27	0.00	1.89	0.40	0.00	4.48	0.24	0.01	1.43	0.28	0.03	0.09
Max	147.00	0.05	39.20	18.70	10.00	161.71	17.30	0.05	10.80	11.40	0.21	0.97
Min	124.00	0.04	27.40	15.90	10.00	139.66	15.50	0.02	2.40	9.70	0.00	0.40

	Limnetic Hypolimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3+NO_2$	SiO_2	SO_4	Total P	Turbidity
98Mean	138.20	0.07	35.26	16.76	11.00	155.09	16.28	0.04	8.62	10.62	0.04	0.63
98SE	7.07	0.04	3.38	0.31	1.00	8.07	0.26	0.01	1.06	0.50	0.01	0.08
98Max	161.00	0.21	45.40	17.80	15.00	176.78	16.90	0.05	11.40	11.50	0.08	0.89
98Min	121.00	0.02	25.70	16.00	10.00	130.06	15.40	0.02	5.22	8.70	0.00	0.46
Mean	139.17	0.07	35.92	17.17	10.83	155.09	16.50	0.05	7.79	10.50	0.03	0.60
SE	5.85	0.03	2.83	0.48	0.83	8.07	0.31	0.00	1.20	0.43	0.01	0.07
Max	161.00	0.21	45.40	19.20	15.00	176.78	17.60	0.05	11.40	11.50	0.08	0.89
Min	121.00	0.02	25.70	16.00	10.00	130.06	15.40	0.02	3.60	8.70	0.00	0.44

Table 6-6. Values from the littoral zone profiles in Round Lake. Means (SE) are given. Most profiles had 1-4 readings. ND=no data available, Sp. Cond.=specific conductance and Cond.=conductance.

Date	2/11/98	5/19/98	6/17/98	7/14/98	8/11/98	9/9/98	5/18/99
Temperature(C)	ND	20.55 (0.02)	23.20 (0.43)	24.37 (0.01)	25.15 (0.03)	21.11 (0.01)	18.28 (0.03)
DO(mg/L)	ND	8.96 (0.04)	10.43 (0.88)	7.98 (0.51)	8.08 (0.28)	8.50 (0.04)	8.58 (0.09)
DO(%saturation)	ND	99.8 (0.5)	122.1 (9.3)	95.5 (6.0)	98.2 (3.5)	95.7 (0.6)	91.3 (1.1)
pH	ND	8.42 (0.01)	8.58 (0.02)	8.49 (0.03)	8.58 (0.01)	8.68 (0.01)	8.54 (0.01)
TDS (mg/L)	ND	237 (0)	227 (0)	211 (0)	223 (1)	219 (0)	235 (0)
Sp. Cond. (μS/cm)	ND	365 (0)	350 (1)	325 (0)	344 (1)	337 (0)	363 (1)
Cond. (μS/cm)	ND	334 (0)	338 (3)	321 (0)	345 (1)	312 (1)	316 (0)

Beaver Lake

Beaver Lake stratified in 1997 by mid-July and remained stratified until overturn occurred in early September (Figure 6-22). During stratification, dissolved oxygen remained above 2 mg/L to a depth of about 8 m (Figure 6-23). Similarly, oxygen saturation was high until about 6-8 m during stratification (Figure 6-24). When the lake was not stratified, oxygen saturation ranged from 80 to 110%. Based on temperature profiles, the lake did not stratify for any long period of time in 1998 (Figure 6-22). However, oxygen profiles indicated that the lake was stratified as early as 2 June and remained unmixed until the end of August (Figure 6-22), with oxygen saturation also showing a typical summer clinograde curve (Figure 6-24).

The water chemistry in the epilimnion above the limnetic zone showed no surprises (Tables 6-7 and 6-8). No parameters were significantly different between years (all t-test P-values > 0.009; P-critical = 0.007 after Bonferonni adjustment). Magnesium and silica could not be compared because silica was not added to the sampling regime until August 1997 (thus leaving an n=2). In general, within-year variation was minimal as suggested by SEM of <20% of the means for parameters.

The water chemistry in the hypolimnion did not show much between-year variation (Table 6-7). Only sulfur was significantly higher in 1998 compared to 1997 (P = 0.001), whereas all other parameters were not significantly different (all P-values > 0.05). Again, parameters added later in the season (silica) had fewer replicates, which prevented statistical testing. Chloride showed a temporal trend toward increasing concentration each year. Turbidity varied temporally as indicated by the SEM being about 40% of the mean for all years combined (Table 6-7).

Water samples from the littoral zone showed similar trends as was observed in the hypolimnion. Only sulfur differed significantly between 1997 and 1998 (P = 0.002, mean 1998> mean 1997), whereas all other parameters were statistically similar (all P-values > 0.02, except magnesium and silica, which were not tested). Chloride concentration increased temporally each year as observed in the other sites, and silica also increased over time in each year.

When parameters were compared between epilimnion and hypolimnion (paired tests), no parameter showed significant differences. Silica was only slightly different (P = 0.044). There was also no significant difference between the epilimnion and littoral water samples for all parameters. Silica was the only parameter that showed even a slight difference between the hypolimnion and littoral samples (P = 0.032).

Limnology

The low nitrogen concentrations in the epilimnion (both limnetic and littoral) could suggest that productivity was high or that nitrogen-loading rates were very low. Low nitrogen concentrations in highly productive lakes are due to the rapid uptake of NH_4^+ and NO_3^- by plants during active photosynthesis. The low nitrogen in the hypolimnion, however, is not characteristic of a stratified, productive lake. Beaver Lake is polymictic, which probably prevents the hypolimnetic waters from becoming anoxic. Because of the frequent mixing of the water column, aerobic processes likely dominate the hypolimnetic microbial community. This condition would prevent buildup of ammonia in the hypolimnion, which is consistent with our results. The low N-NO_3^-, NO_2^- in the hypolimnion is suggestive of a eutrophic lake where the waters were anoxic and denitrification was intense. This condition is not likely the case in Beaver Lake, as was indicated by the oxygen profiles. The nitrogen data coupled with the low total phosphorus suggests that Beaver Lake is oligotrophic and does not receive excessive inputs of either phosphorus or nitrogen.

Beaver Lake is mildly alkaline. Hardness, magnesium, and calcium (major cations affecting hardness) suggest the lake is hard. Specific conductance was also high throughout the study.

Silica also tended to increase with time in the epilimnion and littoral in 1998 (no trends could be observed in 1997). This may suggest an abundance of diatoms in the epilimnion during summer months. Sulfates were within the range of typical lakes; that is, the sulfate concentrations in Beaver Lake do not suggest any unusual conditions. The comparable sulfate concentration in the hypolimnion and epilimnion is indicative of oligotrophic lakes (eutrophic, stratified lakes tend to have lower sulfate but higher hydrogen sulfide in the hypolimnetic water), although the polymictic nature of Beaver Lake also may influence the sulfate vertical profile. Chloride tended to increase slightly over time each year indicating that it was being concentrated, possibly due to evapotranspiration processes in the lake. Specific conductivity was in the range expected based on the cation concentrations.

The final evaluation of the lake based on the water chemistry data alone is that the lake is in good condition. The polymictic status prevents nutrients from being locked into the sediments. Nutrient concentrations were low in the water column so the lake should be considered oligotrophic to possibly mesotrophic.

Figure 6-22. Temperature profile for Beaver Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999

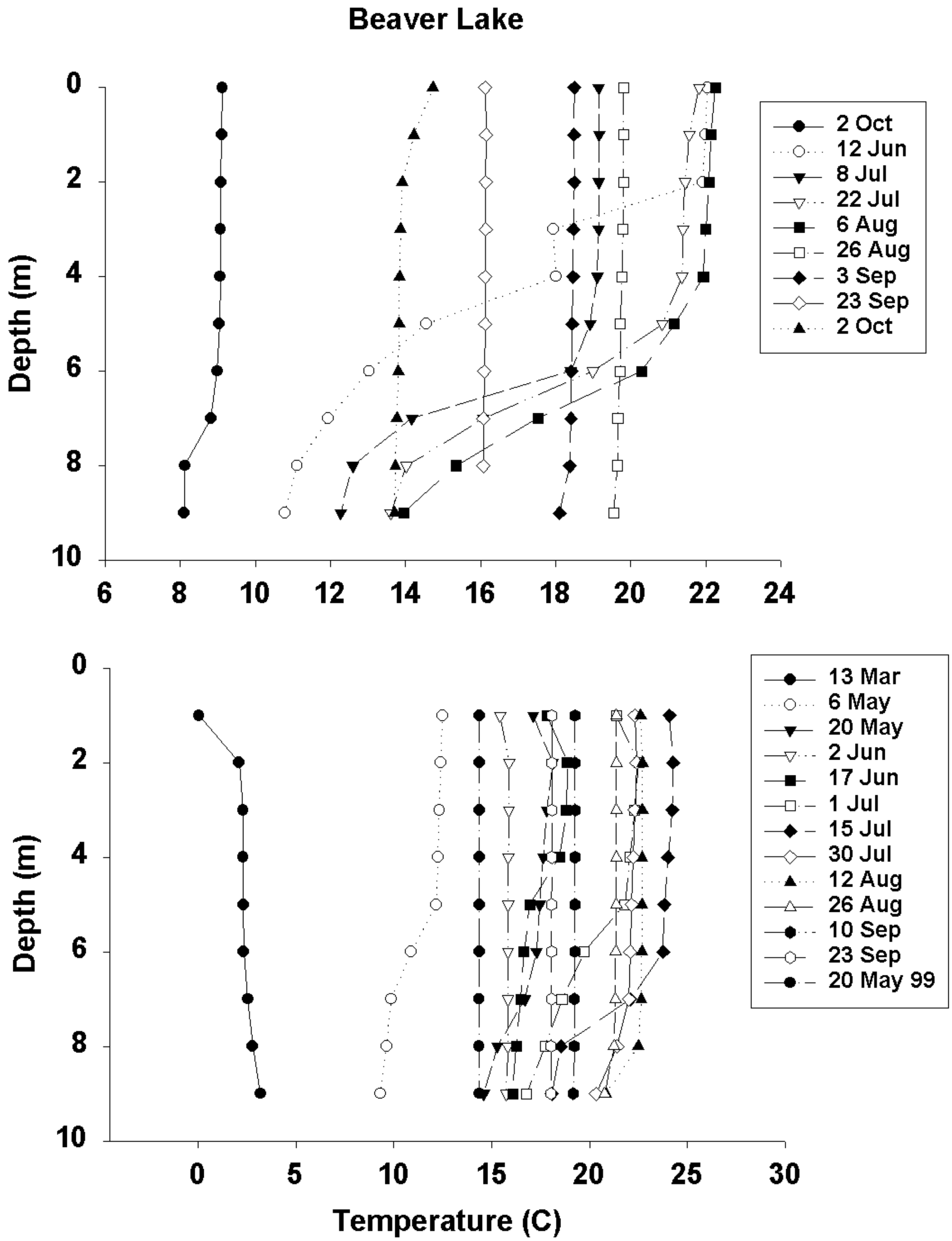


Figure 6-23. Dissolved oxygen profile for Beaver Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999

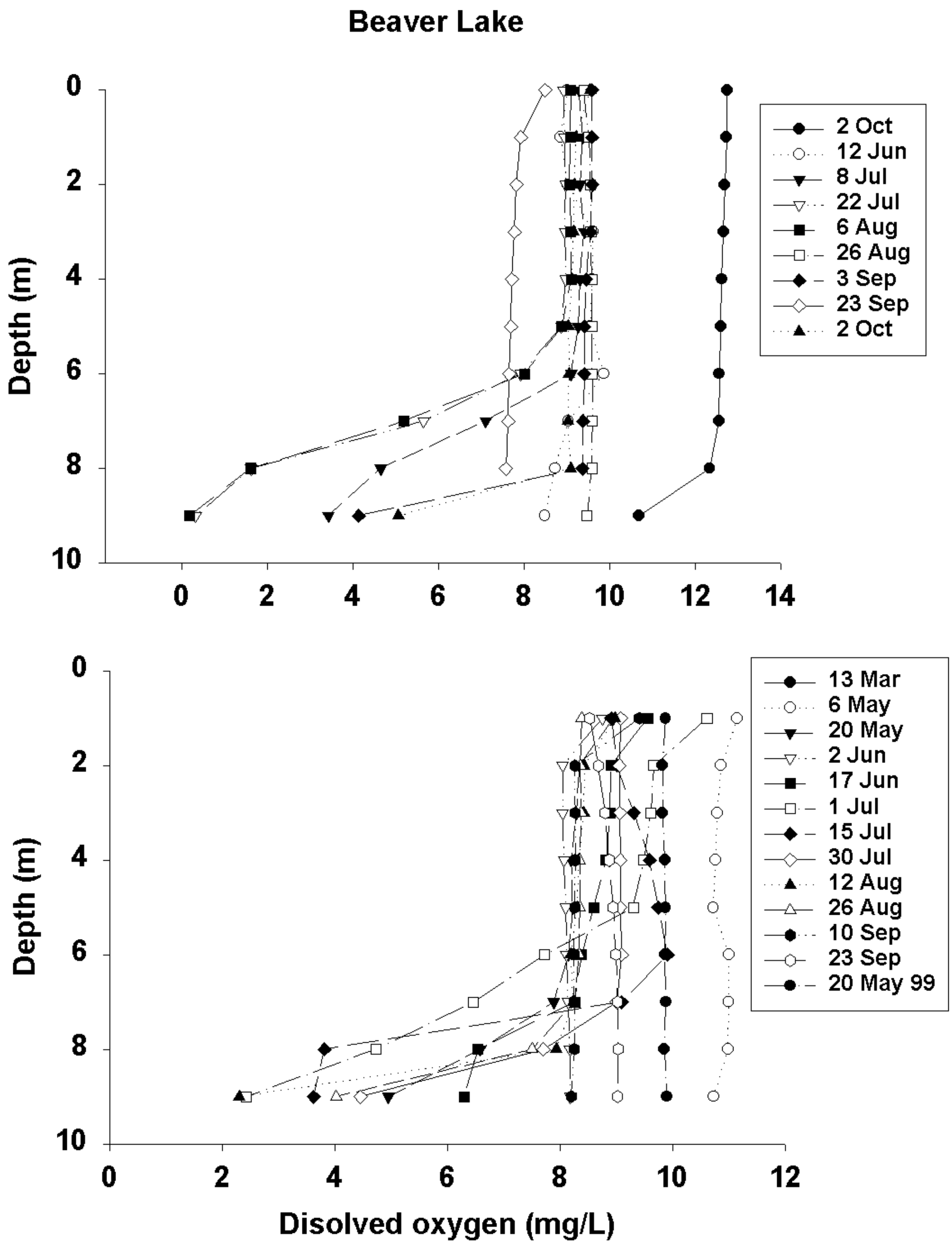


Figure 6-24. Oxygen saturation calculated from measured water temperature and dissolved oxygen for Beaver Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999

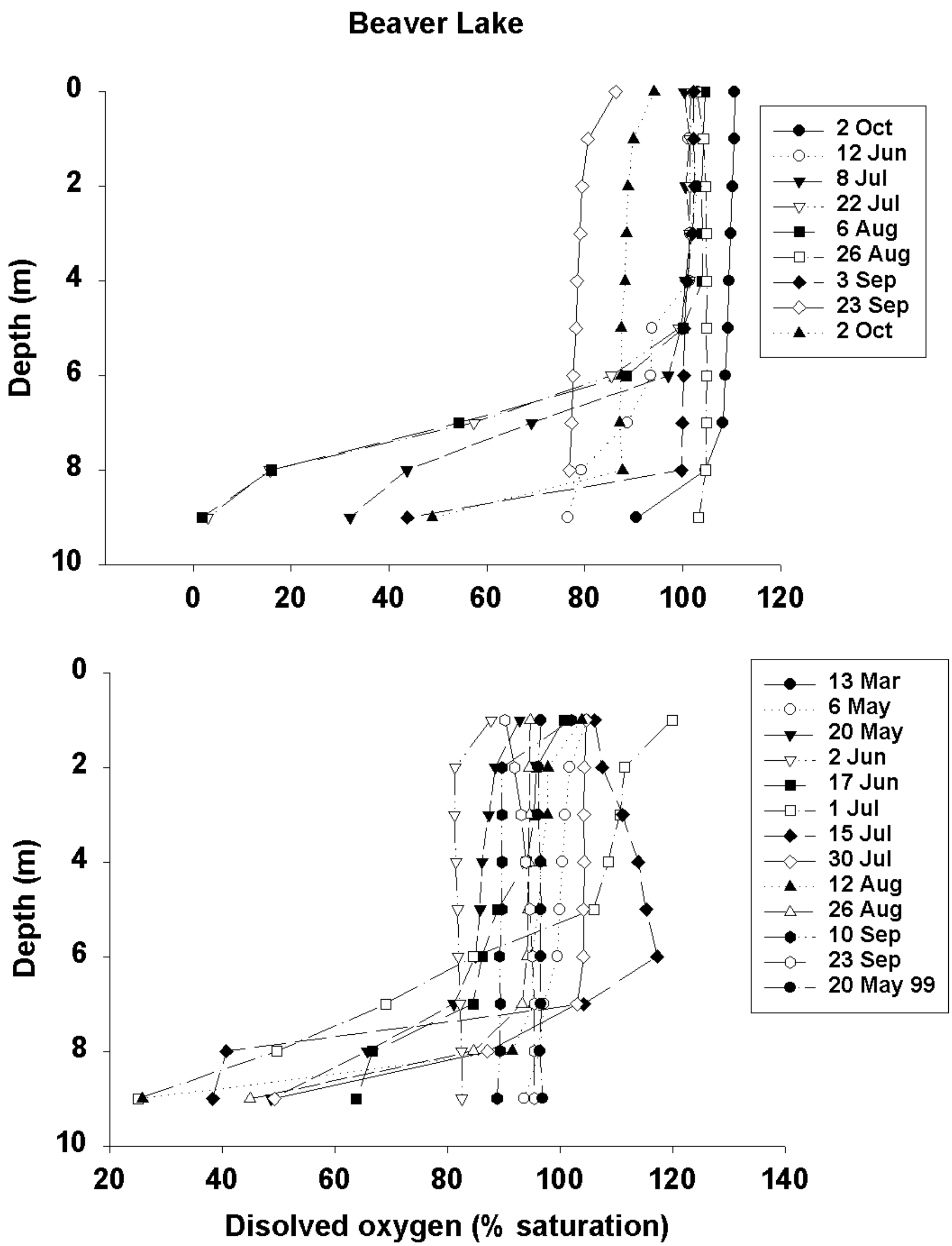


Figure 6-25. pH profiles for Beaver Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

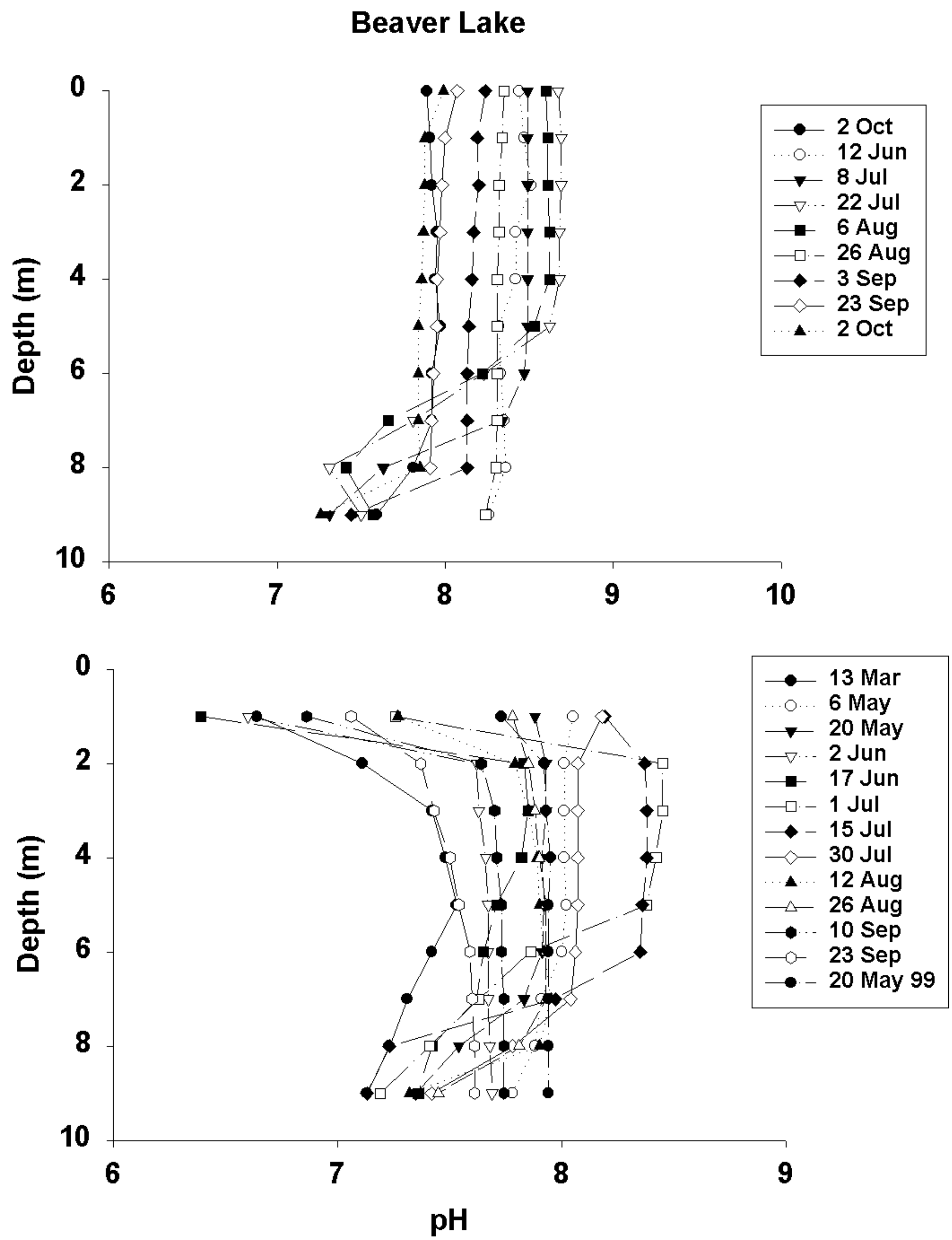


Figure 6-26. Conductivity profiles for Beaver Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

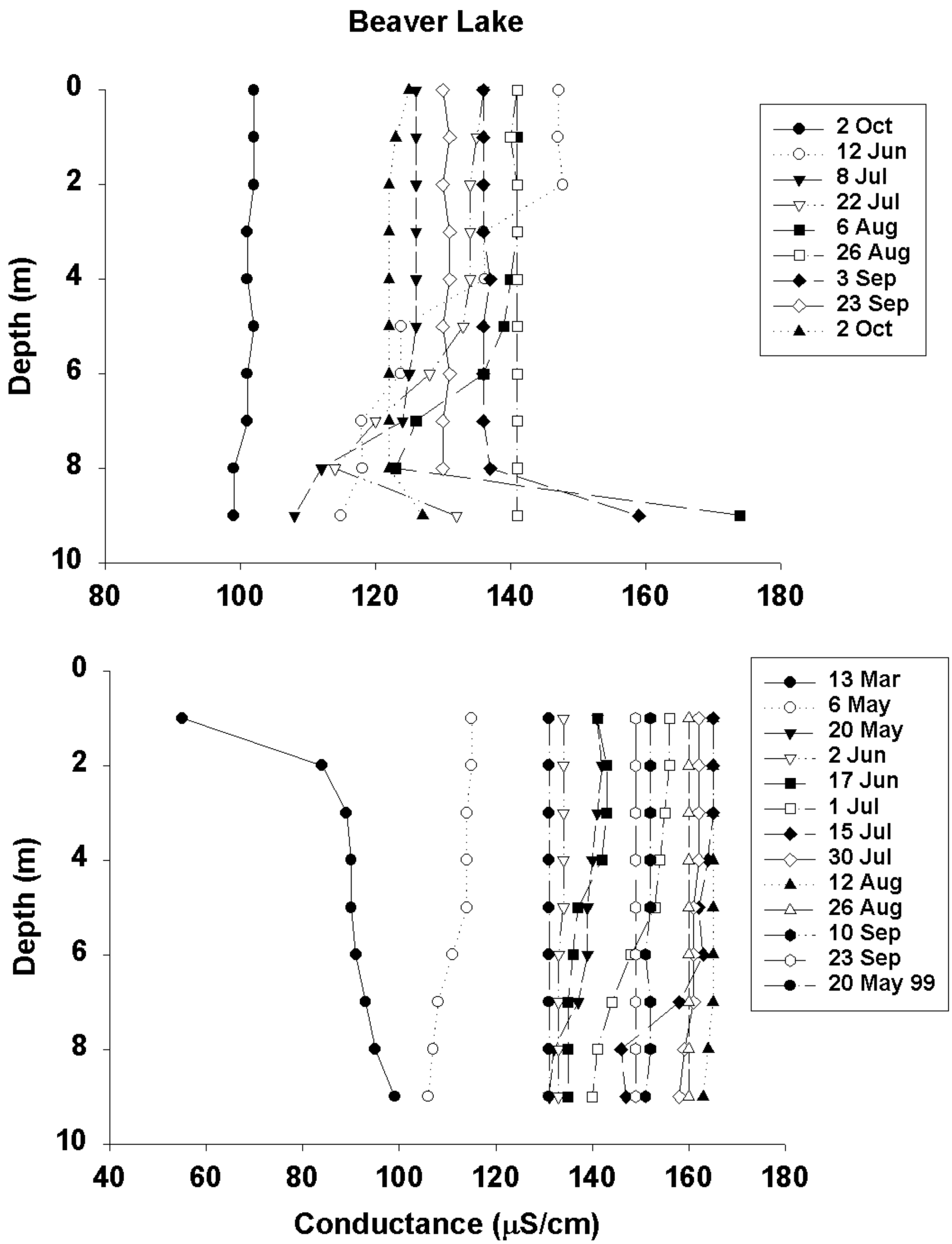


Figure 6-27. Specific conductivity profiles for Beaver Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

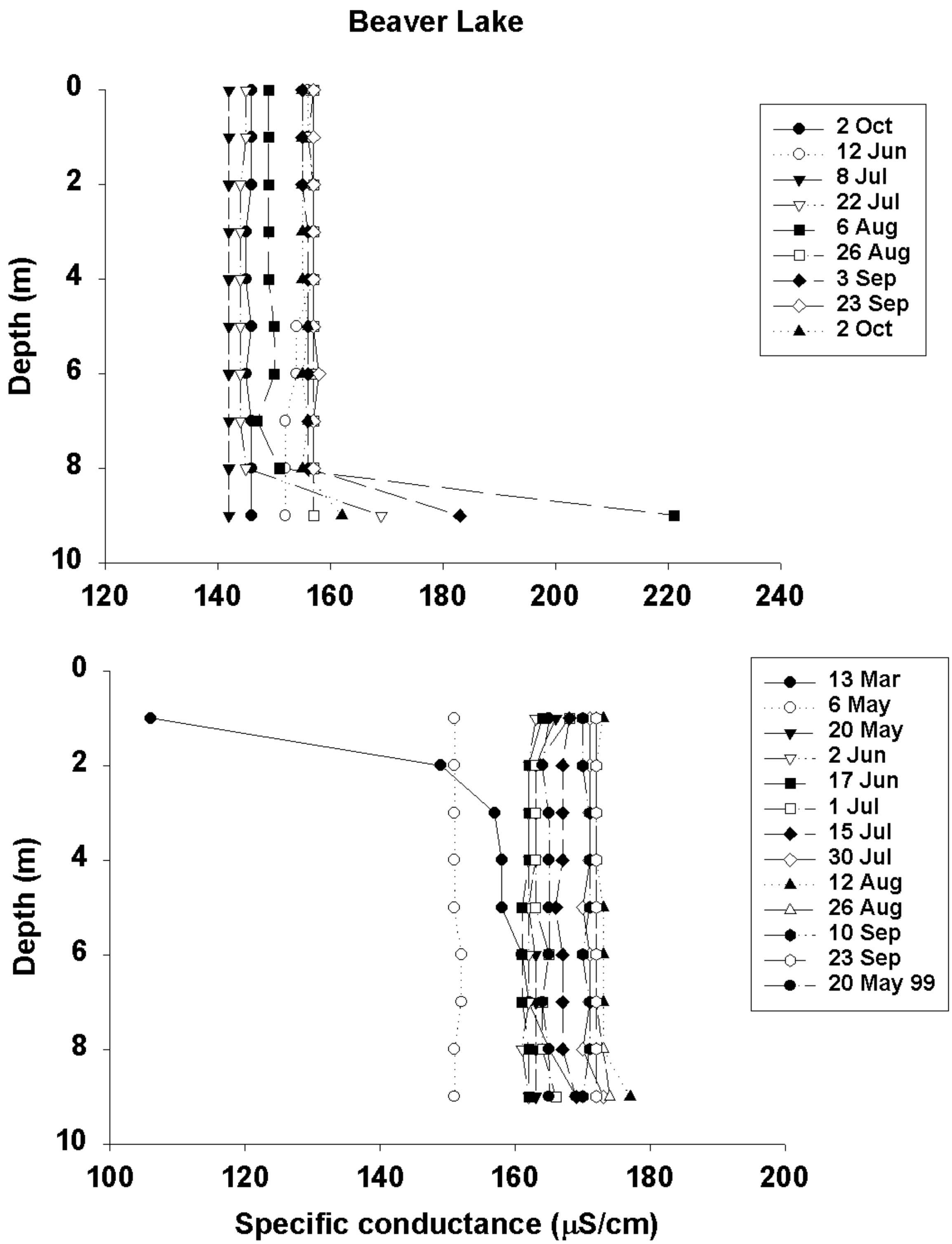


Figure 6-28. Total dissolved solids profiles for Beaver Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

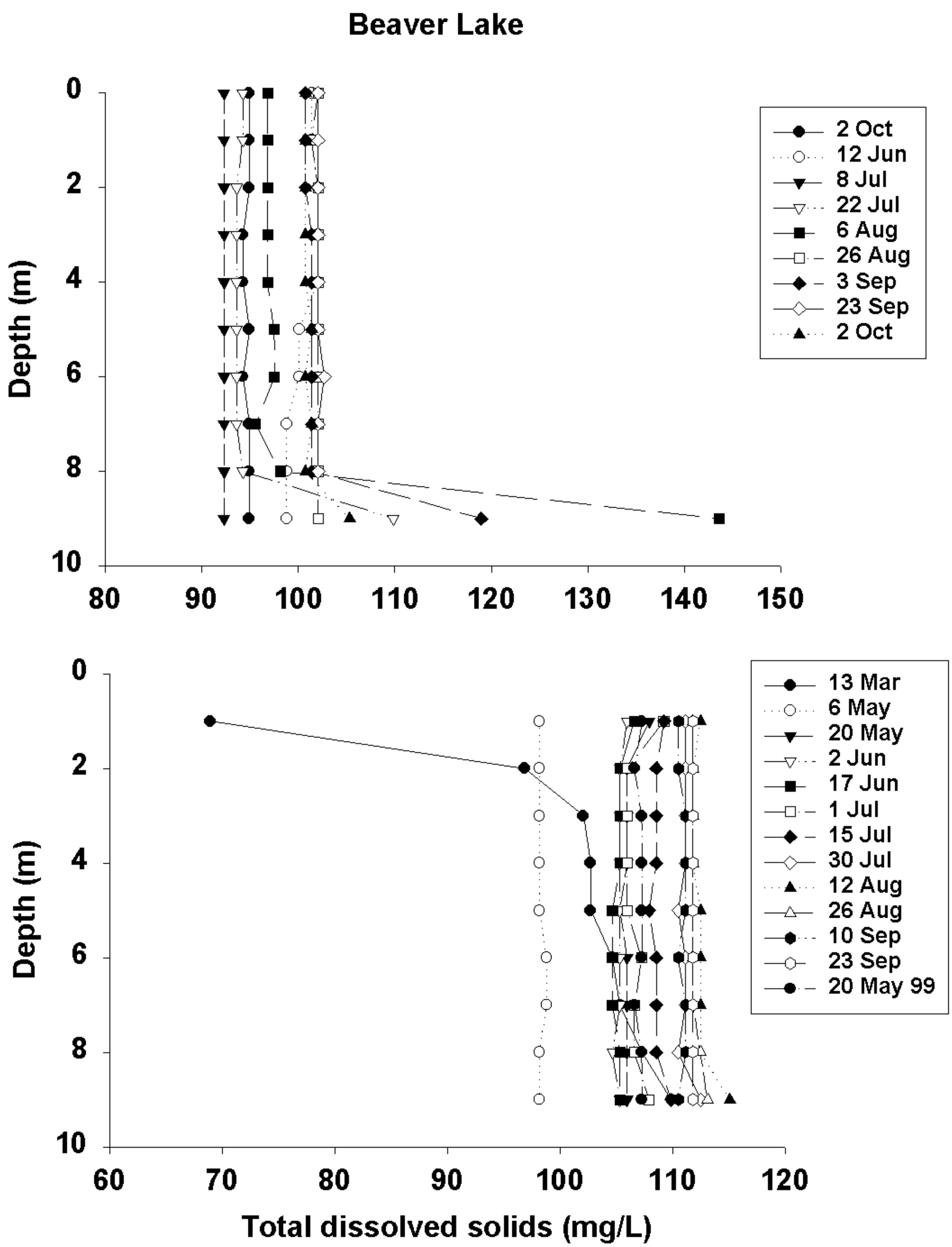


Table 6-7. Means, standard errors (SE), maximums (Max), and minimums (Min) for 1997, 1998, and for 1997-1999 in Beaver Lake, Pictured Rocks National Lakeshore, Michigan. NH4 = ammonium, Ca=calcium, Cl=chloride, Mg=magnesium, N-NO3+NO2=nitrogen as nitrate+nitrite, SiO_2=silica, SO_4=sulfate, and P=phosphorus.

	Littoral Alkalinity (mg/L)	NH_4 (mg/L)	Ca (mg/L)	Cl (mg/L)	Color	Hardness (mg/L)	Mg (mg/L)	$N\text{-}NO_3+NO_2$ (mg/L)	SiO_2 (mg/L)	SO_4 (mg/L)	Total P (mg/L)	Turbidity (NTU)
97Mean	74.48	0.06	40.60	0.64	8.75	82.78	5.50	0.07	8.70	6.23	0.02	0.53
97SE	0.59	0.00	8.74	0.13	2.39	3.59	0.00	0.01	0.20	0.19	0.00	0.13
97Max	76.10	0.07	62.20	0.99	15.00	89.82	5.50	0.10	8.90	6.53	0.03	0.90
97Min	73.30	0.05	25.90	0.42	5.00	74.00	5.50	0.05	8.50	5.70	0.01	0.33
98Mean	77.42	0.05	25.48	0.83	10.00	85.28	5.26	0.04	10.98	7.56	0.02	0.28
98SE	0.79	0.01	0.17	0.10	0.00	0.61	0.08	0.01	0.80	0.15	0.00	0.07
98Max	80.50	0.07	26.00	1.10	10.00	87.57	5.50	0.05	13.90	8.00	0.03	0.41
98Min	76.20	0.02	25.10	0.56	10.00	84.09	5.10	0.02	9.40	7.10	0.01	0.05
Mean	76.01	0.05	31.53	0.81	9.50	84.39	5.35	0.06	10.46	7.10	0.02	0.38
SE	0.64	0.00	4.04	0.10	0.90	1.42	0.07	0.01	0.62	0.27	0.00	0.07
Max	80.50	0.07	62.20	1.40	15.00	89.82	5.50	0.10	13.90	8.30	0.03	0.90
Min	73.30	0.02	25.10	0.42	5.00	74.00	5.10	0.02	8.50	5.70	0.01	0.05

	Limnetic Epilimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3+NO_2$	SiO_2	SO_4	Total P	Turbidity
97Mean	74.03	0.05	38.28	0.64	8.75	83.58	5.45	0.08	8.65	6.22	0.04	0.63
97SE	0.77	0.00	8.02	0.13	2.39	3.24	0.05	0.01	0.25	0.20	0.02	0.14
97Max	76.30	0.06	59.90	0.99	15.00	88.57	5.50	0.10	8.90	6.60	0.09	0.90
97Min	73.00	0.05	25.80	0.43	5.00	74.10	5.40	0.04	8.40	5.70	0.01	0.35
98Mean	78.06	0.05	25.72	0.86	10.00	86.21	5.34	0.04	11.12	7.60	0.02	0.29
98SE	0.79	0.00	0.22	0.11	0.00	0.94	0.11	0.01	0.87	0.14	0.00	0.06
98Max	80.10	0.05	26.20	1.20	10.00	88.48	5.60	0.05	14.30	7.90	0.03	0.43
98Min	76.30	0.05	25.00	0.57	10.00	83.02	5.00	0.02	9.60	7.10	0.00	0.05
Mean	76.13	0.05	30.72	0.82	9.50	85.17	5.39	0.06	10.53	7.11	0.03	0.42
SE	0.80	0.00	3.58	0.10	0.90	1.34	0.07	0.01	0.67	0.27	0.01	0.08
Max	80.10	0.06	59.90	1.40	15.00	88.57	5.60	0.12	14.30	8.20	0.09	0.90
Min	73.00	0.05	25.00	0.43	5.00	74.10	5.00	0.02	8.40	5.70	0.00	0.05

	Limnetic Hypolimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3+NO_2$	SiO_2	SO_4	Total P	Turbidity
97Mean	74.47	0.05	37.90	0.72	13.33	84.82	5.60	0.07	10.00	6.11	0.03	0.91
97SE	0.87	0.00	11.01	0.14	4.41	5.62	0.10	0.01	0.70	0.24	0.01	0.45
97Max	76.20	0.06	59.90	0.98	20.00	92.89	5.70	0.10	10.70	6.52	0.05	1.80
97Min	73.50	0.05	26.00	0.51	5.00	74.00	5.50	0.05	9.30	5.70	0.02	0.32
98Mean	77.65	0.14	25.83	0.77	12.50	86.41	5.33	0.06	11.48	7.45	0.02	0.29
98SE	0.83	0.10	0.25	0.09	2.50	0.87	0.09	0.01	0.66	0.10	0.01	0.09
98Max	80.10	0.44	26.30	1.00	15.00	88.07	5.50	0.09	13.10	7.70	0.04	0.43
98Min	76.50	0.02	25.30	0.56	5.00	84.18	5.10	0.05	10.10	7.20	0.01	0.05
Mean	76.29	0.10	31.00	0.75	12.86	85.73	5.42	0.07	10.98	6.87	0.02	0.56
SE	0.85	0.06	4.83	0.07	2.14	2.20	0.08	0.01	0.55	0.29	0.01	0.22
Max	80.10	0.44	59.90	1.00	20.00	92.89	5.70	0.10	13.10	7.70	0.05	1.80
Min	73.50	0.02	25.30	0.51	5.00	74.00	5.10	0.05	9.30	5.70	0.01	0.05

Table 6-8. Values from the littoral zone profiles. Means (SE) are given. Most profiles had 1-4 readings. ND=no data available, Sp. Cond.=specific conductance and Cond.=conductance.

Date	5/24/97	6/1/00	4/21/00	1/0/00	4/8/00	8/26/97	9/3/97	9/23/97
Temperature(C)	ND	21.98 (0.13)	19.45 (0.01)	23.05 (0.16)	23.06 (0.31)	18.93 (0.03)	18.93 (0.01)	16.31 (0.01)
DO(mg/L)	ND	9.45 (0.08)	9.35 (0)	9.04 (0.03)	9.27 (0.03)	9.56 (0.07)	9.46 (0.01)	8.62 (0.11)
DO(%saturation)	ND	108.1 (1.2)	101.7 (0.0)	105.5 (0.7)	108.2 (0.9)	103.0 (0.9)	101.8 (0.2)	87.9 (1.2)
pH	ND	8.45 (0.03)	8.51 (0)	8.71 (0.01)	8.61 (0.01)	8.29 (0.03)	8.21 (0.01)	8.09 (0)
TDS (mg/L)	ND	101 (0)	91 (0)	94 (0)	96 (0)	101 (0)	102 (0)	101 (0)
Sp. Cond. (μS/cm)	ND	155 (0)	141 (1)	145 (0)	149 (0)	156 (0)	157 (0)	156 (1)
Cond. (μS/cm)	ND	146 (1)	126 (1)	139 (0)	143 (1)	138 (0)	139 (0)	130 (1)

Date	10/2/97	3/13/98	5/6/98	5/20/98	6/2/98	6/17/98	7/1/98
Temperature(C)	14.55 (0.29)	ND	12.58 (0.51)	17.20 (0.03)	15.84 (0.07)	19.62 (0.11)	22.11 (0.86)
DO(mg/L)	10.01 (0.22)	ND	10.56 (0.08)	8.42 (0.05)	8.73 (0.08)	9.24 (0.05)	9.48 (0.01)
DO(%saturation)	98.3 (2.7)	ND	99.3 (0.29)	87.6 (0.4)	88.1 (0.7)	100.9 (0.2)	108.7 (1.7)
pH	7.97 (0.01)	ND	8.14 (0.01)	8.14 (0.03)	7.80 (0.02)	8.12 (0.01)	8.34 (0.15)
TDS (mg/L)	100 (0)	ND	99 (0)	104 (0)	104 (0)	107 (1)	109 (2)
Sp. Cond. (μS/cm)	155 (0)	ND	152 (1)	161 (1)	161 (1)	166 (1)	168 (3)
Cond. (μS/cm)	124 (1)	ND	116 (2)	137 (0)	133 (1)	149 (0)	159 (1)

Date	7/15/98	7/30/98	8/12/98	8/26/98	9/10/98	9/23/98	5/20/99
Temperature(C)	24.60 (0.41)	22.03 (0.67)	23.53 (0.05)	21.41 (0.06)	19.62 (0.26)	18.21 (0.14)	14.27 (0.10)
DO(mg/L)	10.01 (0.09)	9.51 (0.28)	9.03 (0.01)	8.66 (0.01)	9.27 (0.77)	9.40 (0.04)	96.80 (0.59)
DO(%saturation)	120.3 (0.2)	108.8 (1.8)	106.3 (0.1)	98.0 (0.1)	101.2 (7.9)	99.7 (0.1)	105.3 (0)
pH	8.43 (0.02)	8.02 (0.08)	8.17 (0.01)	8.04 (0.01)	7.84 (0.01)	7.85 (0.02)	9.91 (0.04)
TDS (mg/L)	109 (0)	111 (0)	111 (0)	111 (0)	110 (0)	111 (1)	105 (0)
Sp. Cond. (μS/cm)	168 (0)	171 (1)	171 (0)	172 (0)	170 (1)	171 (1)	162 (0)
Cond. (μS/cm)	166 (1)	161 (1)	166 (1)	160 (1)	153 (1)	149 (1)	129 (0)

Grand Sable Lake

Grand Sable Lake demonstrated very strong clinograde curves in temperature throughout the entire sampling season in 1997 (Figure 6-29). However, the dissolved oxygen curves were not clear (Figure 6-30). These inconsistencies may be caused by inconsistent meter readings, but assuming the data are accurate, there may have been periods of high primary production at depths. This explanation is not likely because the oxygen saturation data do not indicate high productivity (Figure 6-31). Alternatively, there could have been some mixing of the water layers enough to redistribute the oxygen without greatly affecting the temperature profile. In 1998, the temperature and dissolved oxygen profiles were similar, both showing typical stratified clinograde curves (Figs. 6-30 & 6-31). The lake stratified by early June and remained stratified throughout the sampling dates.

The water chemistry is summarized in Tables 6-9 and 6-10. No parameters were significantly different between years in the epilimnion (all t-test P-values > 0.05; P-critical = 0.007 after Bonferonni adjustment). In general, within-year variation was minimal as suggested by SEM of <20% of the means for all parameters

The water chemistry in the hypolimnion did not show much between-year variation (Table 6-9). Only alkalinity was significantly higher in 1998 compared to 1997 (P = 0.002), whereas all other parameters were not significantly different (all P-values > 0.04).

Water samples from the littoral zone showed similar trends as were observed in the hypolimnion. No parameter was statistically different (all P-values > 0.02).

When parameters were compared between epilimnion and hypolimnion (paired tests), no parameter showed significant differences. As expected there was also no significant difference between the epilimnion and littoral water samples for all parameters. Alkalinity was only slightly different (P = 0.045). Silica was the only parameter that showed a significant difference between the hypolimnion and littoral samples (P = 0.004, mean hypolimnion > mean littoral).

Limnology

Grand Sable Lake is a dimictic, holomictic lake. This mixis state would influence the nitrogen concentrations in eutrophic lakes: ammonia would build up and nitrate would be denitrified intensively so as to be absent in the hypolimnion. Although nitrates are in low concentration in the hypolimnion in Grand Sable Lake, some spikes were observed. Ammonia was below detection in the hypolimnion, which suggests oligotrophic conditions based on the mixis state. As in Beaver Lake, the low total phosphorus and nitrogen in Grand Sable Lake indicates an oligotrophic condition with apparently minimal external loading rates of nutrients.

Grand Sable Lake is less alkaline and hard than Beaver Lake but can still be considered alkaline and hard. The slight difference in the alkalinity between the epilimnion and littoral is of no limnological concern and in fact may be spurious. Specific conductance was consistent through the water column, and the values indicate hard water.

Sulfate concentrations did not indicate unusual extremes in Grand Sable Lake, although low sulfate levels in the hypolimnion would be expected from a stratified lake, which we did not observe. This lack of gradient in sulfate concentration, again, suggests an oligotrophic condition in the lake.

The significantly higher silica concentration in the hypolimnion compared with the epilimnion is typical of both oligotrophic and eutrophic lakes. This difference is due to diatoms using the silicates to form their frustules. Where diatom production is high, such as in the epilimnion of a productive lake, the dissolved silica concentrations will be low. It appears, then, that the diatoms in the epilimnion of Grand Sable Lake are using the silica and it follows that the unproductive hypolimnion will have a higher silica concentration, as was observed.

The final evaluation of Grand Sable Lake based solely on water chemistry is that the lake is in good condition. Nutrient concentrations were low throughout the water column indicating no eutrophication problems.

Figure 6-29. Temperature profile for Grand Sable Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

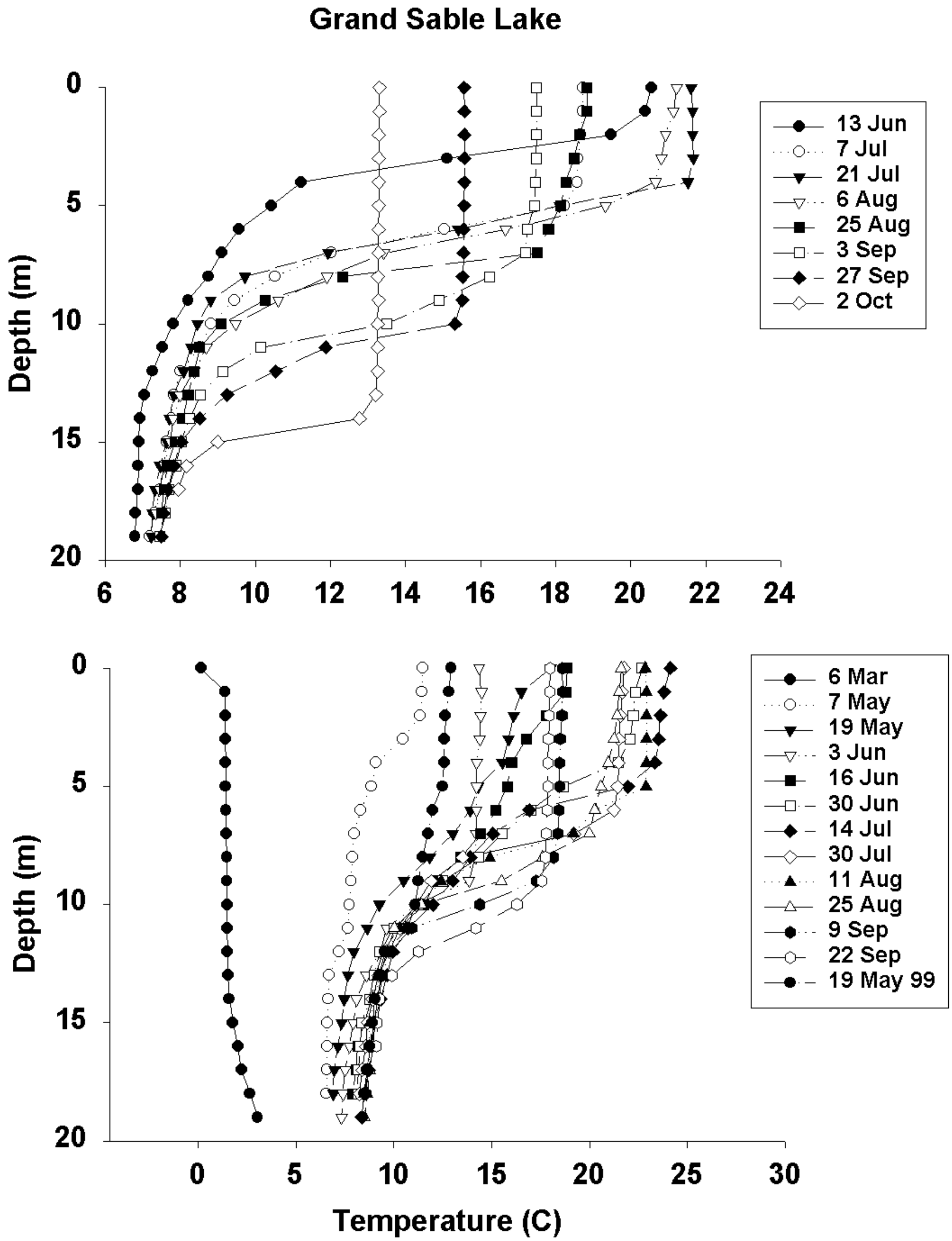


Figure 6-30. Dissolved oxygen profile for Grand Sable Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

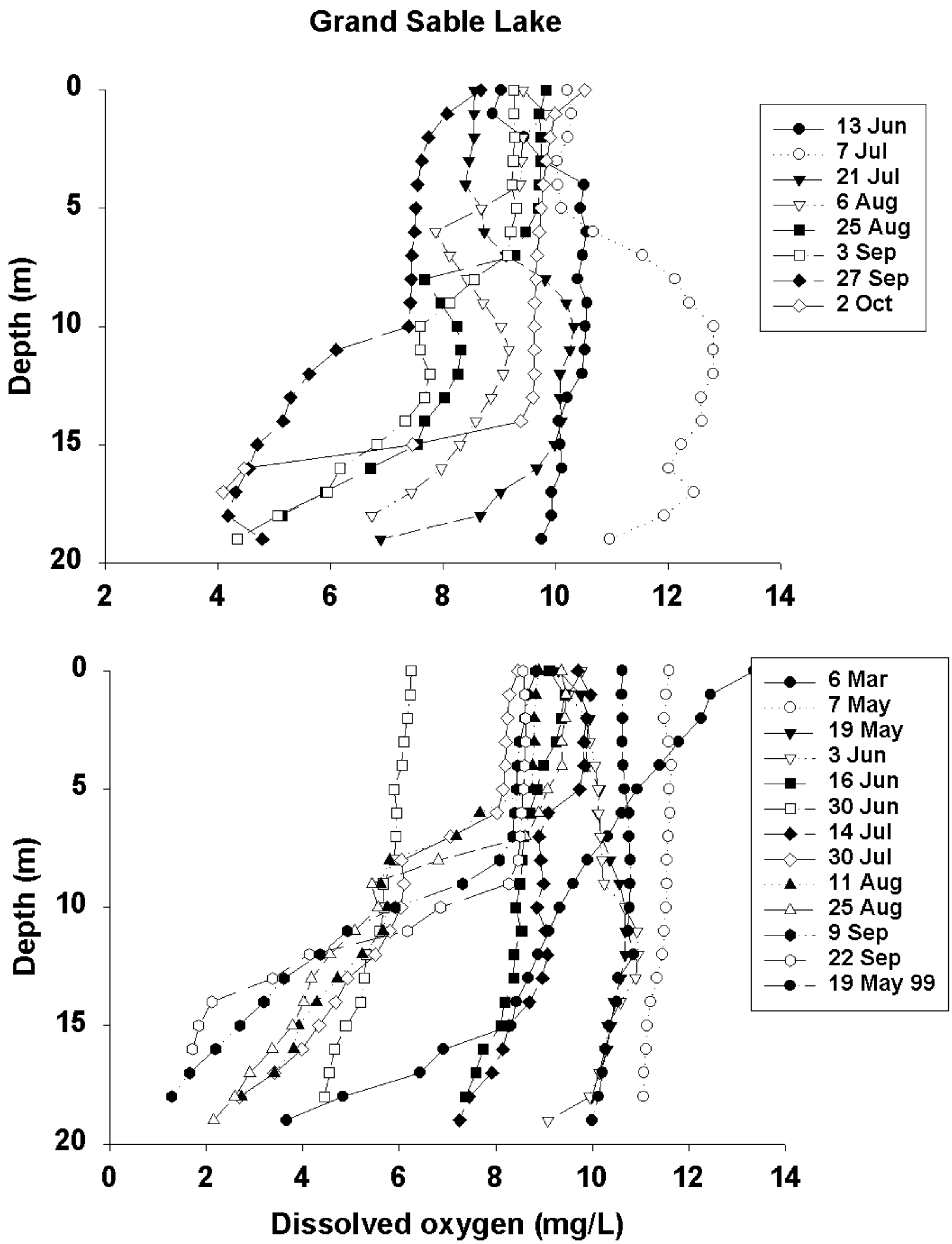


Figure 6-31. Oxygen saturation calculated from measured water temperature and dissolved oxygen in Grand Sable Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

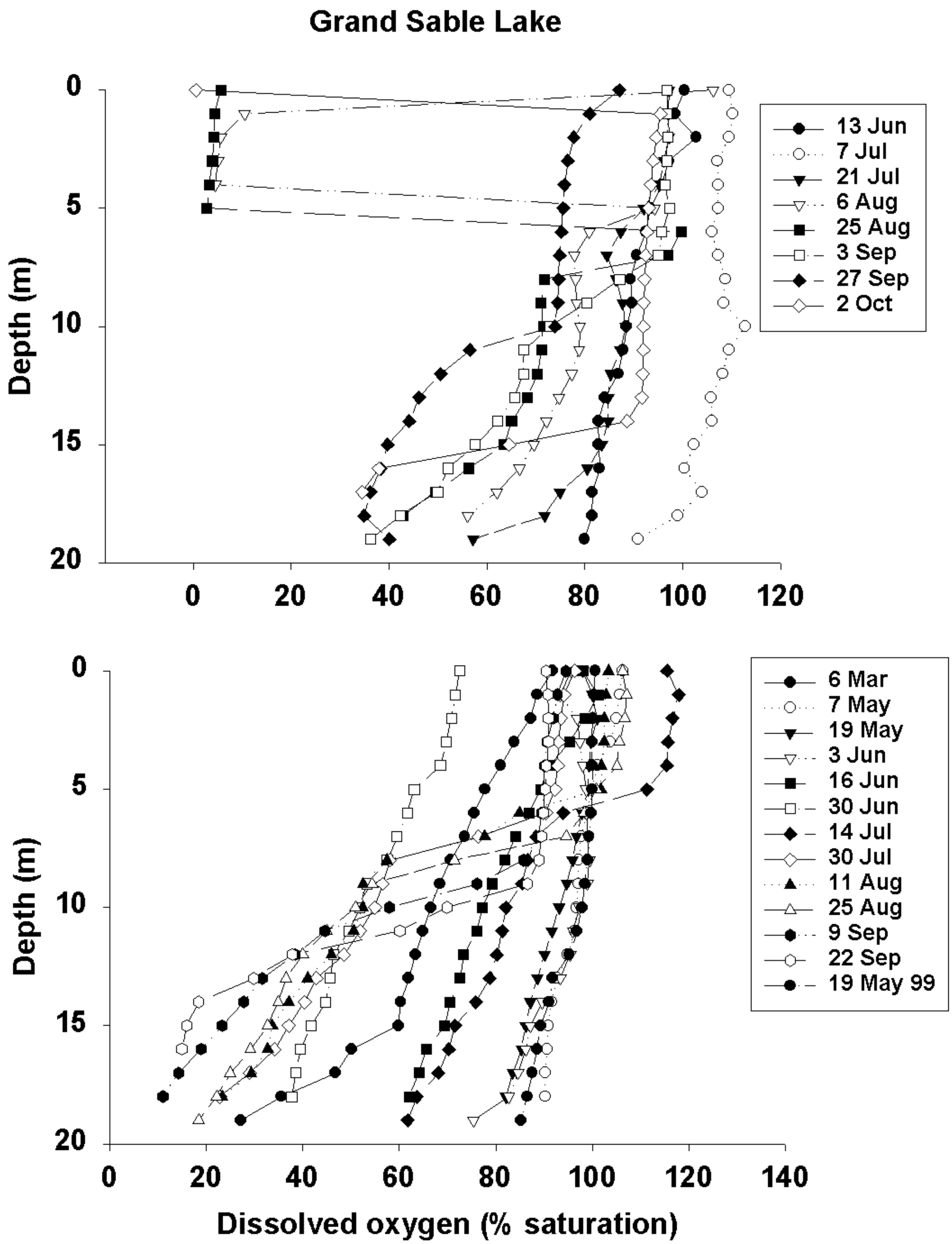


Figure 6-32. pH profiles for Grand Sable Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

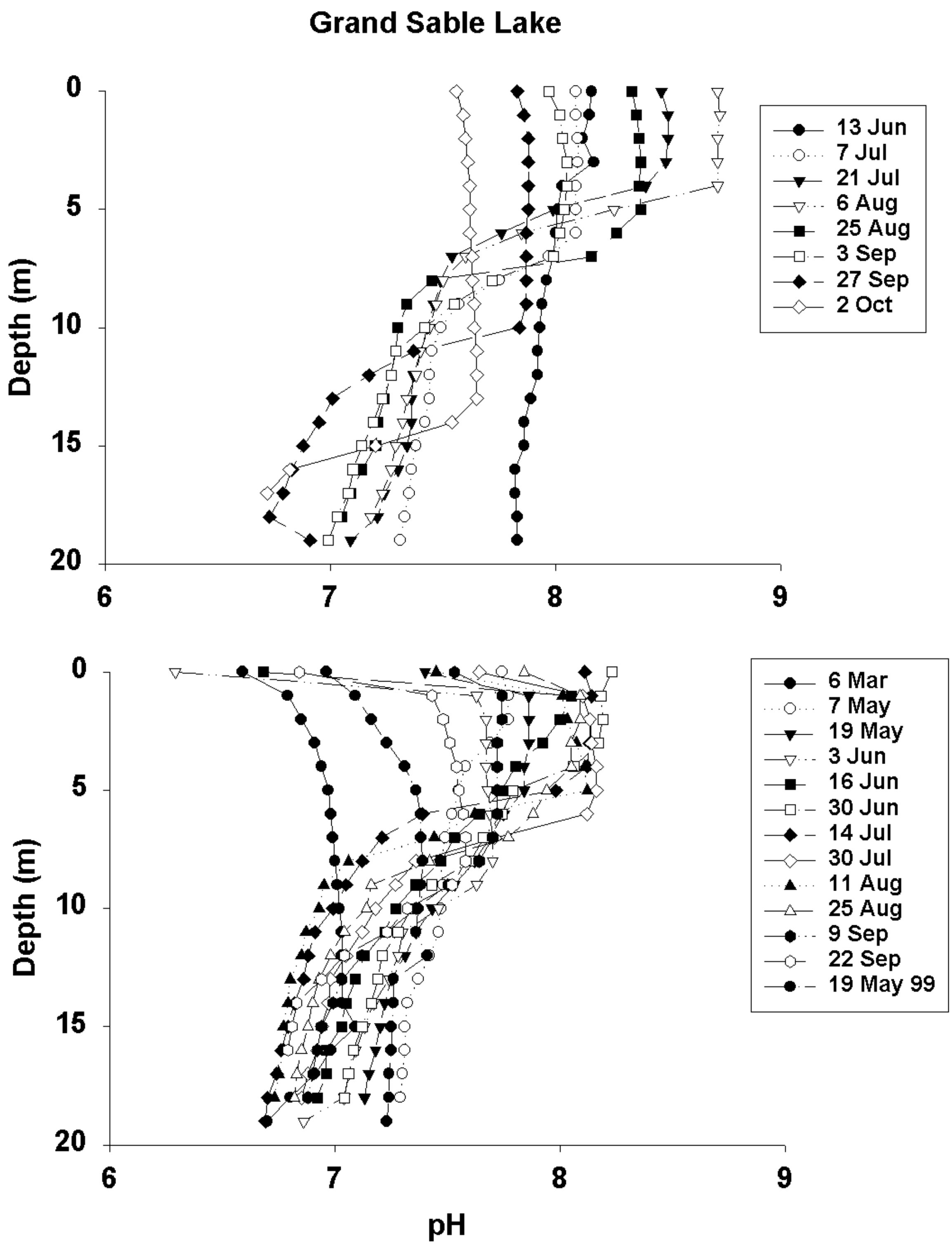


Figure 6-33. Conductivity profiles for Grand Sable Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

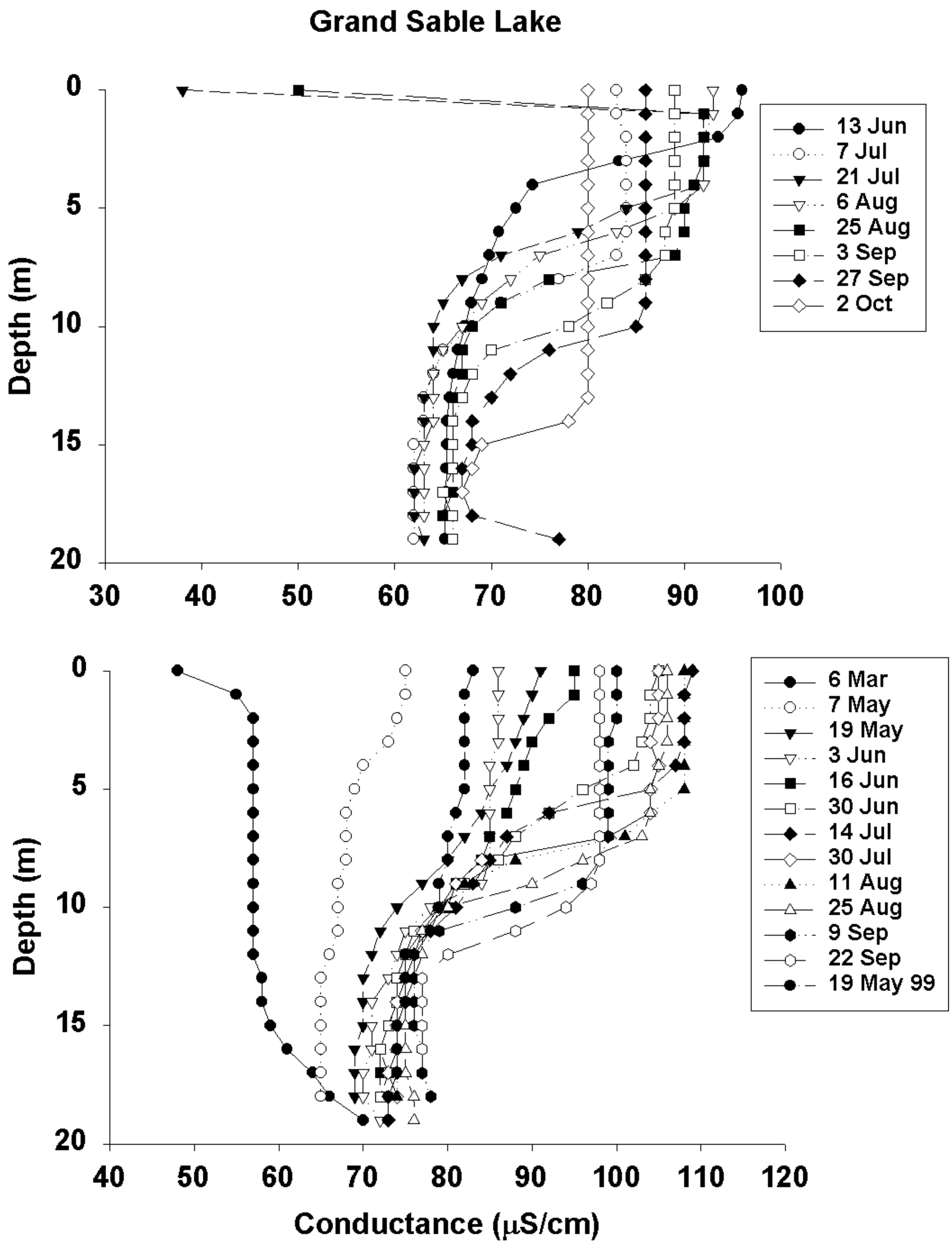


Figure 6-34. Specific conductivity profiles for Grand Sable Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

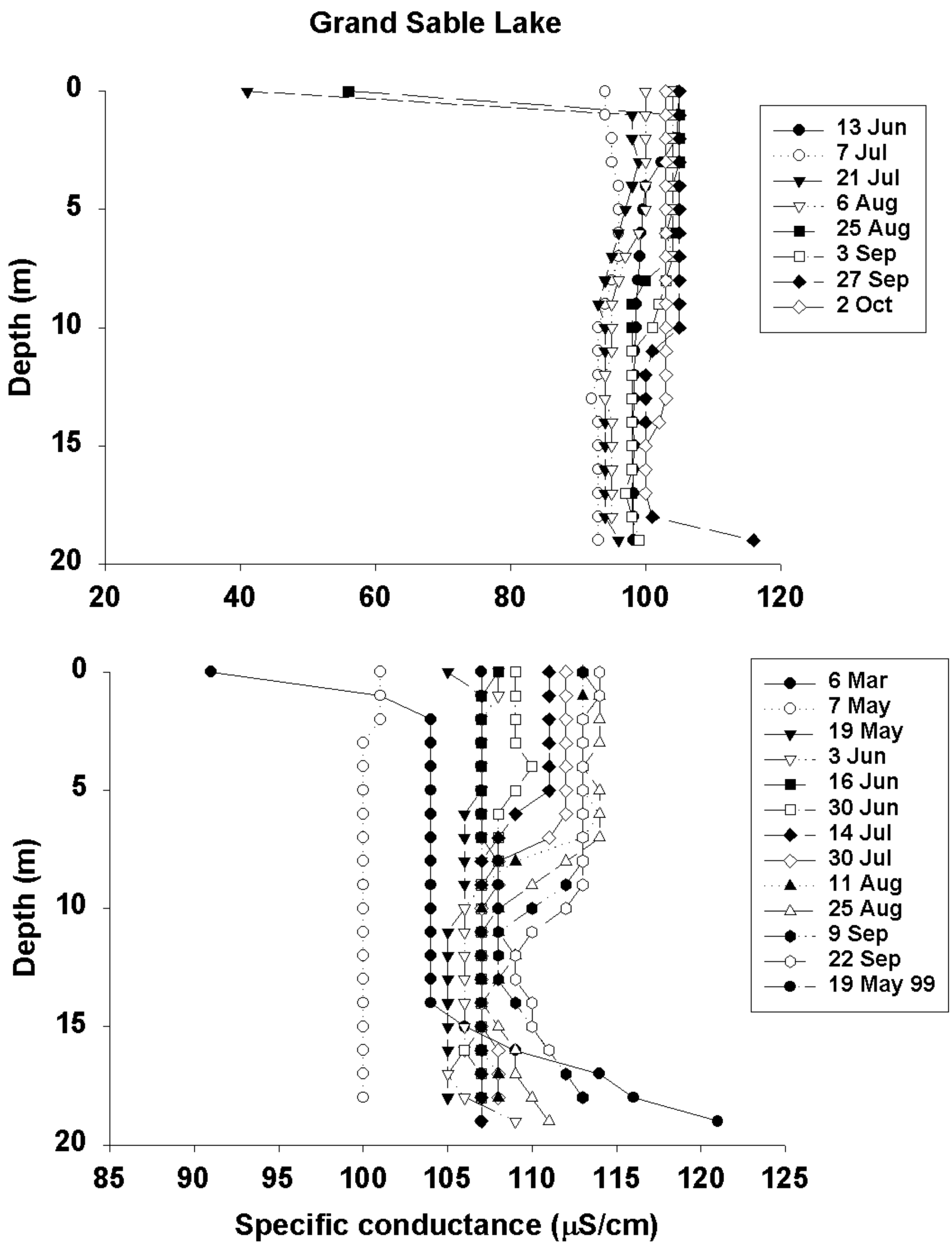


Figure 6-35. Total dissolved solids profiles for Grand Sable Lake, Pictured Rocks National Lakeshore, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

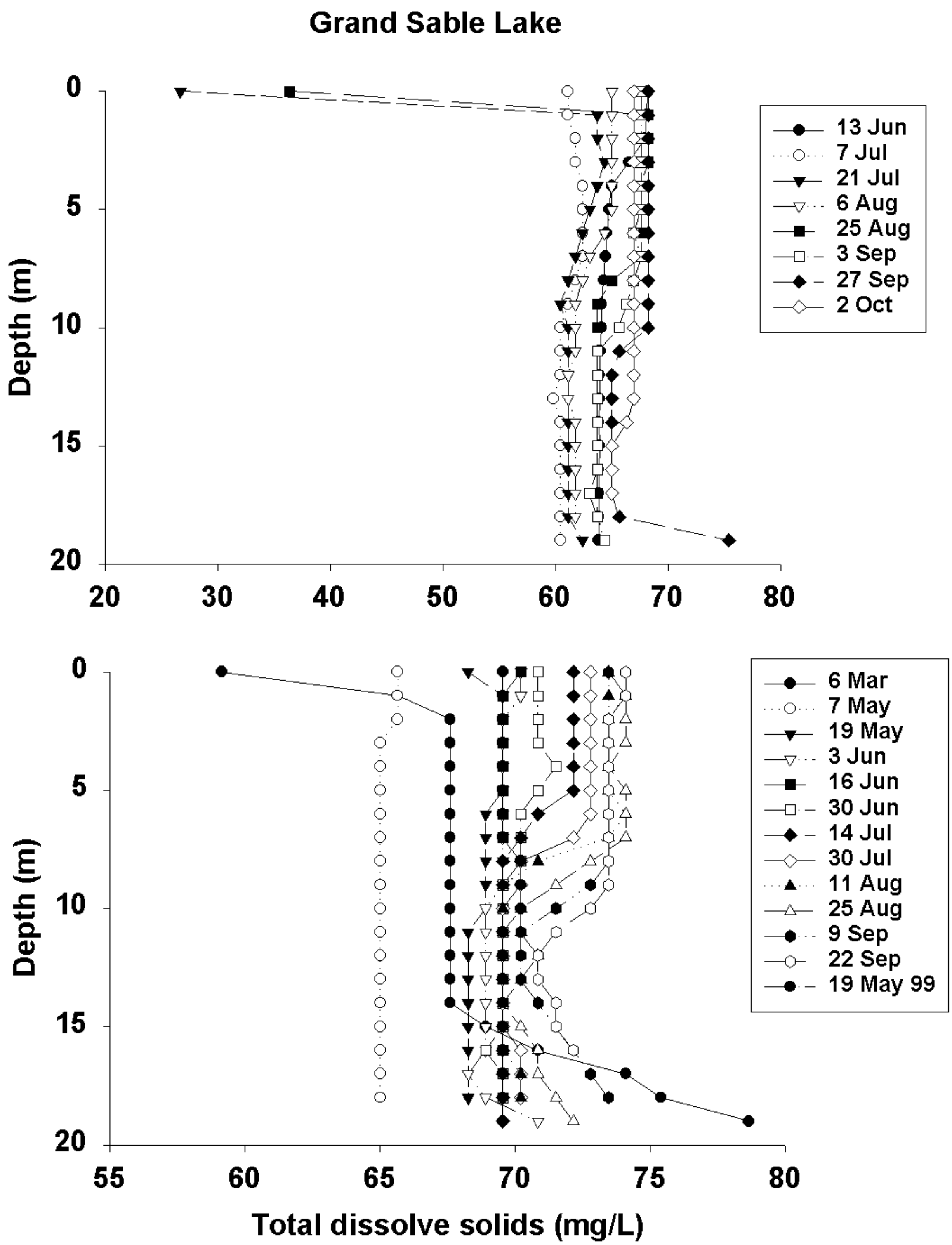


Table 6-9. Means, standard errors (SE), maximums (Max), and minimums (Min) for 1997, 1998, and for 1997-1999 inGrand Sable Lake, Pictured Rocks National Lakeshore, Michigan. NH4 = ammonium, Ca=calcium, Cl=chloride, Mg=magnesium, N-NO3+NO2=nitrogen as nitrate+nitrite, SiO_2=silica, SO_4=sulfate, and P=phosphorus.

Littoral	Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3+NO_2$	SiO_2	SO_4	Total P	Turbidity
	mg/L	mg/L	mg/L	mg/L		mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	NTU
97Mean	48.90	0.05	18.30	0.53	27.50	49.79	5.00	0.08	4.43	4.57	0.02	0.64
97SE	0.54	0.00	4.00	0.04	7.50	6.88	0.21	0.02	1.42	0.11	0.01	0.11
97Max	50.20	0.06	30.20	0.61	35.00	61.69	5.40	0.12	6.10	4.87	0.04	0.90
97Min	48.00	0.05	13.20	0.41	5.00	30.00	4.70	0.05	1.60	4.40	0.01	0.38
98Mean	51.28	0.24	14.04	0.85	29.00	55.40	4.94	0.09	5.46	4.78	0.03	0.40
98SE	0.57	0.19	0.41	0.11	2.45	1.65	0.16	0.05	0.09	0.14	0.00	0.02
98Max	52.80	0.99	15.30	1.10	35.00	60.85	5.50	0.28	5.80	5.10	0.03	0.44
98Min	49.70	0.05	12.70	0.53	25.00	50.65	4.60	0.02	5.30	4.30	0.03	0.33
Mean	50.24	0.15	15.73	0.77	29.00	53.23	4.99	0.08	5.20	4.71	0.03	0.50
SE	0.50	0.09	1.63	0.09	3.06	2.79	0.11	0.02	0.46	0.09	0.00	0.06
Max	52.80	0.99	30.20	1.30	35.00	61.69	5.50	0.28	6.20	5.10	0.04	0.90
Min	48.00	0.05	12.70	0.41	5.00	30.00	4.60	0.02	1.60	4.30	0.01	0.33

Limnetic Epilimnion	Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3+NO_2$	SiO_2	SO_4	Total P	Turbidity
97Mean	49.18	0.05	18.85	0.51	28.75	52.554	4.80	0.09	5.67	4.54	0.03	0.76
97SE	0.65	0.00	5.48	0.06	8.00	4.34	0.58	0.01	0.56	0.12	0.01	0.15
97Max	50.70	0.05	35.00	0.61	40.00	63.09	5.80	0.12	6.50	4.84	0.05	1.20
97Min	48.00	0.05	10.70	0.41	5.00	42.37	3.80	0.05	4.60	4.30	0.02	0.52
98Mean	51.78	0.04	13.88	0.85	30.00	54.75	4.88	0.04	5.58	4.76	0.02	0.41
98SE	0.80	0.01	0.42	0.11	2.24	1.80	0.19	0.01	0.15	0.13	0.00	0.06
98Max	54.40	0.05	15.00	1.10	35.00	59.28	5.30	0.05	6.10	5.00	0.03	0.61
98Min	49.60	0.02	12.40	0.52	25.00	48.26	4.20	0.02	5.30	4.30	0.02	0.28
Mean	50.81	0.05	15.88	0.79	30.00	54.03	4.89	0.06	5.69	4.68	0.03	0.55
SE	0.63	0.00	2.17	0.10	3.16	1.85	0.20	0.01	0.20	0.09	0.00	0.08
Max	54.40	0.05	35.00	1.30	40.00	63.09	5.80	0.12	6.50	5.00	0.05	1.20
Min	48.00	0.02	10.70	0.41	5.00	42.37	3.80	0.01	4.60	4.30	0.02	0.28

Limnetic Hypolimnion	Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3+NO_2$	SiO_2	SO_4	Total P	Turbidity
97Mean	46.58	0.05	17.95	0.53	35.00	52.99	4.50	0.19	5.13	4.57	0.03	1.21
97SE	0.39	0.00	5.20	0.05	10.00	4.92	0.53	0.06	1.77	0.11	0.01	0.35
97Max	47.70	0.05	33.20	0.61	45.00	60.03	5.30	0.30	7.60	4.88	0.04	2.20
97Min	45.90	0.05	9.90	0.40	5.00	39.13	3.50	0.05	1.70	4.40	0.01	0.60
98Mean	51.24	0.05	13.84	0.86	37.00	54.41	4.82	0.15	7.20	4.76	0.03	1.10
98SE	0.82	0.00	0.42	0.11	2.55	1.53	0.13	0.11	0.41	0.13	0.00	0.63
98Max	53.90	0.05	14.90	1.10	45.00	59.03	5.30	0.57	8.60	5.00	0.03	3.60
98Min	48.80	0.05	12.40	0.51	30.00	49.49	4.50	0.02	6.10	4.30	0.02	0.41
Mean	49.26	0.05	15.48	0.77	36.00	53.95	4.74	0.15	6.46	4.71	0.03	1.10
SE	0.85	0.00	2.03	0.09	3.86	1.96	0.18	0.06	0.65	0.09	0.00	0.33
Max	53.90	0.05	33.20	1.30	45.00	60.03	5.30	0.57	8.60	5.00	0.04	3.60
Min	45.90	0.05	9.90	0.40	5.00	39.13	3.50	0.02	1.70	4.30	0.01	0.41

Table 6-10. Values from the littoral zone profiles in Grand Sable Lake. Means (SE) are given. Most profiles had 1-4 readings. ND=no data available, Sp. Cond.=specific conductance and Cond.=conductance.

Date	6/13/97	7/7/97	7/21/97	8/6/97	8/25/97	9/3/97	9/22/97
Temperature(C)	20.54 (0.01)	19.58 (0.40)	21.41 (0.01)	21.60 (0.125)	20.23 (0.27)	17.34 (0.01)	15.74 (0.01)
DO(mg/L)	8.89 (0.09)	10.49 (0.05)	8.57 (0.01)	9.39 (0.04)	9.73 (0.01)	9.34 (0)	8.93 (0.15)
DO(%saturation)	98.7 (0.9)	114.4 (0.2)	96.9 (0.1)	106.6 (0.2)	107.5 (0.3)	97.4 (0)	89.9 (1.6)
pH	8.18 (0.05)	8.13 (0.01)	8.51 (0.01)	8.72 (0.04)	8.44 (0.01)	8.11 (0)	7.98 (0.01)
TDS (mg/L)	67 (0)	62 (0)	63 (0)	65 (1)	68 (0)	67.6 (0)	67 (0)
Sp. Cond. (μS/cm)	104 (0)	96 (0)	98 (0)	100 (1)	105 (0)	104 (0)	104 (0)
Cond. (μS/cm)	95 (0)	86 (1)	92 (0)	93 (1)	95 (1)	89 (0)	86 (0)

Date	10/2/97	3/6/98	5/7/98	5/19/98	6/3/98	6/16/98	6/30/98
Temperature(C)	13.17 (0.01)	ND	11.85 (0.02)	17.87 (0.28)	14.43 (0.02)	18.69 (0.02)	23.03 (0.01)
DO(mg/L)	10.45 (0.21)	ND	11.52 (0.03)	10.10 (0.06)	10.35 (0.04)	9.24 (0.41)	6.15 (0.01)
DO(%saturation)	99.6 (2.0)	ND	106.5 (0.3)	106.4 (0.0)	101 (0.3)	99.0 (4.3)	71.7 (0.2)
pH	7.8 (0.0)	ND	7.8 (0.0)	8.1 (0.0)	7.8 (0.0)	8.29 (0.01)	8.37 (0.02)
TDS (mg/L)	66 (0)	ND	65 (1)	69 (0)	69 (0)	69 (0)	71 (0)
Sp. Cond. (μS/cm)	103 (0)	ND	100 (1)	107 (0)	107 (0)	107 (1)	110 (0)
Cond. (μS/cm)	80 (0)	ND	74.5 (1)	92 (1)	86 (0)	94 (1)	105 (1)

Date	7/14/98	7/30/98	8/11/98	8/25/98	9/9/98	9/22/98	5/19/99
Temperature(C)	25.08 (0.18)	21.73 (0.01)	22.81 (0.02)	21.59 (0.01)	19.31 (0.44)	18.01 (0.01)	13.70 (0.28)
DO(mg/L)	9.76 (0.05)	8.33 (0.03)	8.74 (0.01)	9.26 (0.04)	8.965 (0.1)	8.63 (0.01)	10.82 (0.02)
DO(%saturation)	118.3 (0.2)	94.8 (0.3)	101.6 (0.2)	105.1 (0.5)	97.2 (2.3)	91.2 (0.1)	104.4 (0.3)
pH	8.23 (0.01)	8.21 (0.01)	8.18 (0.01)	8.15 (0.01)	7.87 (0.02)	7.73 (0.01)	7.63 (0.06)
TDS (mg/L)	71 (0)	73 (0)	73 (0)	73 (0)	73.45 (0)	73 (0)	69 (0)
Sp. Cond. (μS/cm)	110 (0)	113 (0)	113 (0)	113 (1)	113 (0)	113 (0)	107 (0)
Cond. (μS/cm)	110 (1)	106 (0)	108 (0)	106 (0)	101 (1)	98 (0)	84 (1)

Sargent Lake

Sargent Lake was stratified during the entire sampling period in both 1997 and 1998, with exception of the February 1998 sampling date. The temperature (Figure 6-36) and oxygen profiles (Figs. 6-37 and 6-38) were clinograde except the positive heterograde curves in June 1997 and slightly positive heterograde curves in June 1998. these heterograde curves suggest high algal activity at or just above the metalimnion. Fall turnover must occur after the sampling period, and the lake becomes inversely stratified during winter months.

The water chemistry for Sargent Lake is summarized by year in Tables 6-11 and 6-12. Within the epilimnion, no parameters were significantly different between years (all t-test P-values > 0.10; P-critical = 0.007 after Bonferonni adjustment), although sulfate was slightly higher in 1998 than in 1997 (P = 0.013). In general, within-year variation was minimal as suggested by SEM's of <20% of the means for all parameters.

The water chemistry in the hypolimnion showed similar trends as the epilimnion (Table 6-11). No parameters were significantly different (all P-values > 0.03), although magnesium was marginally higher in 1997 than in 1998 (P = 0.04). Ammonia was below detection limits in all years except for a few dates, but in general ammonia was very low in the system.

Water samples from the littoral zone showed similar trends as observed in the other zones. No parameters were significantly different (all P-values > 0.11). Ammonia was only detected in three samples, but was consistently below detection limit in other samples, which indicates very low concentrations.

When parameters were compared between epilimnion and hypolimnion (paired tests), only marginally significant differences were detected (P = 0.03 for turbidity). All other parameters showed no significant differences. As expected there was also no significant difference between the epilimnion and littoral water samples for all parameters. Silica was significantly higher in the hypolimnion compared to the littoral (P < 0.001) and turbidity was marginally higher (P = 0.008).

Limnology

Sargent Lake was slightly alkaline as was indicated in the specific conductance measurements. The calcium and magnesium concentrations were at moderate levels, and together with the data on hardness indicate that the lake was only mildly hard. Chloride ion levels were low in comparison to typical lake levels, so there is no need for concern based on the data.

Nutrients in the lake were low. Nitrogen as nitrate, nitrite, and ammonia were usually below detection limits. Total phosphorus was also low throughout the study. Based on the water chemistry, there is no concern that Sargent Lake is in a state of eutrophication, and it could even be considered oligotrophic or slightly mesotrophic.

As in most of the other lakes involved in the study, Sargent Lake had higher silica in the hypolimnion than the littoral zone. Differences in silica usually result from differences in diatom production, and diatom production is the likely cause of the observed lower silica in the littoral. Data from vertical profiles was not complete, but they show that Sargent Lake stratifies during summer. However, oxygen is still present in low amounts fairly deep. The available oxygen and the low productivity in the epilimnion work to keep the redox potential from becoming too low. This potential results in a lack of conversion of sulfates to hydrogen sulfides in the hypolimnion and, thus, no change in sulfate concentration with depth was observed.

The general conclusions, based on the water chemistry, are that Sargent Lake is in good condition. Nutrient concentrations are low throughout the water column suggesting that the lake is oligotrophic.

Figure 6-36. Temperature profiles for Sargent Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

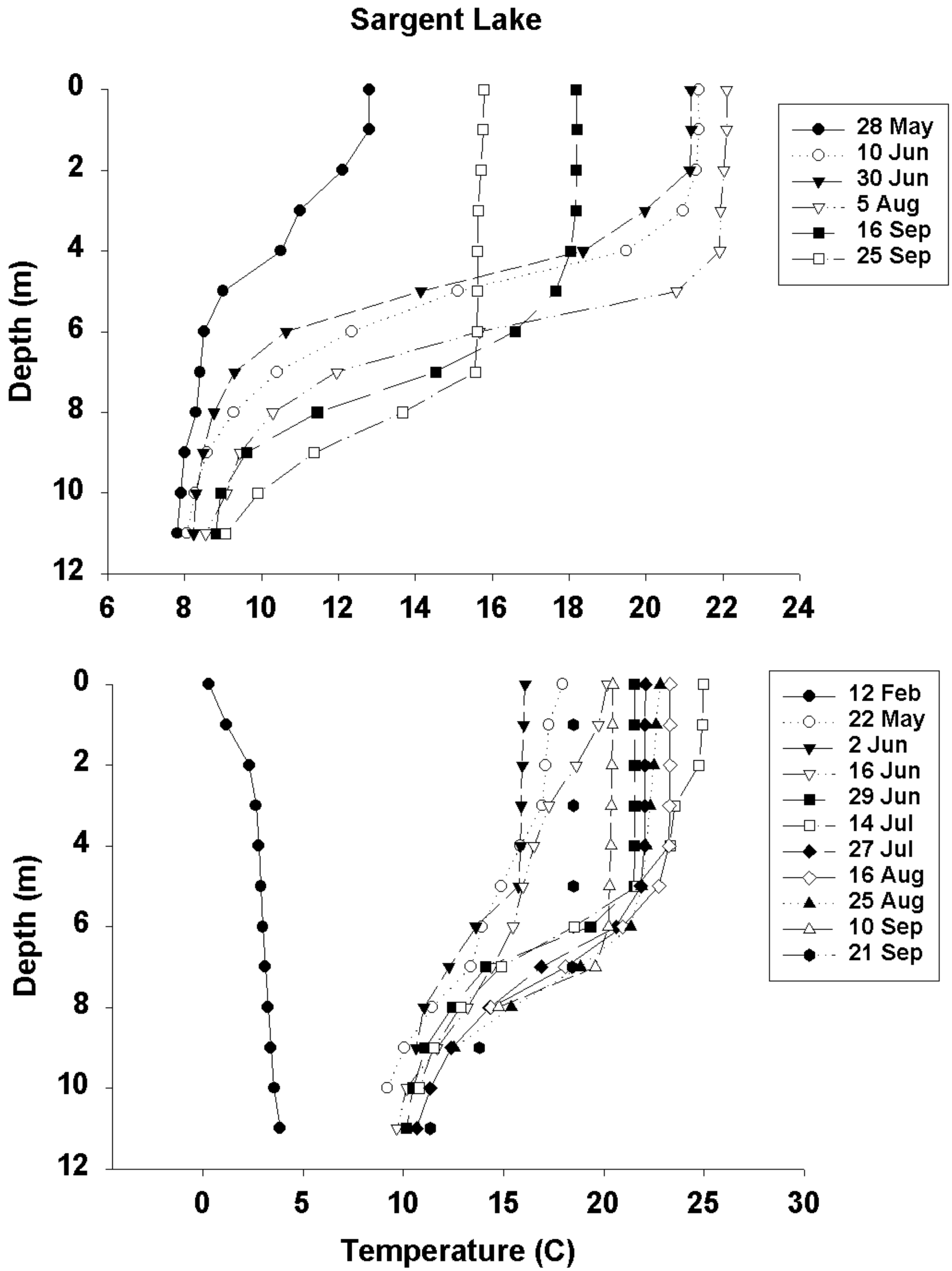


Figure 6-37. Dissolved oxygen profiles for Sargent Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

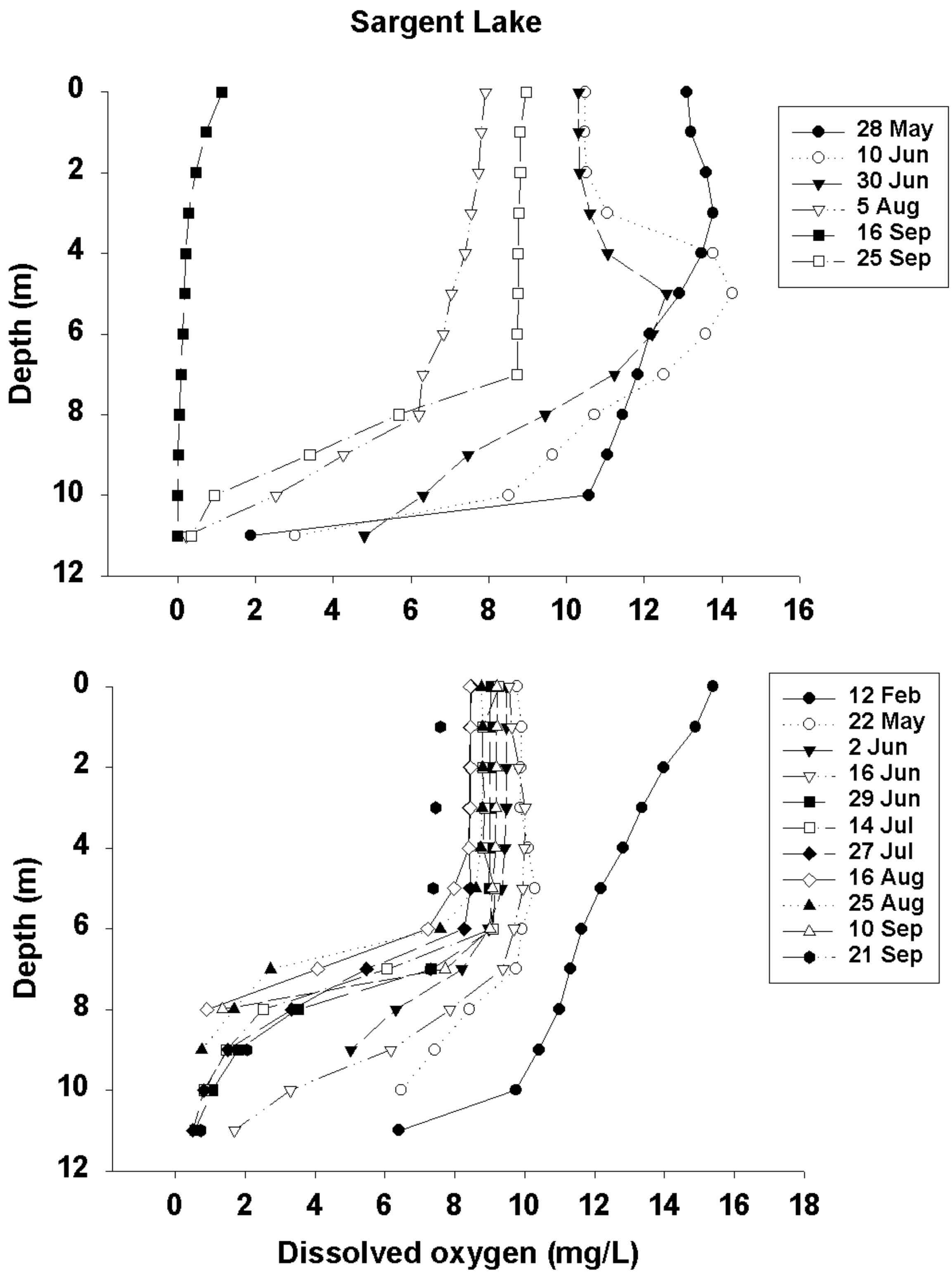


Figure 6-38. Oxygen saturation calculated from measured water temperature and dissolved oxygen in Sargent Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

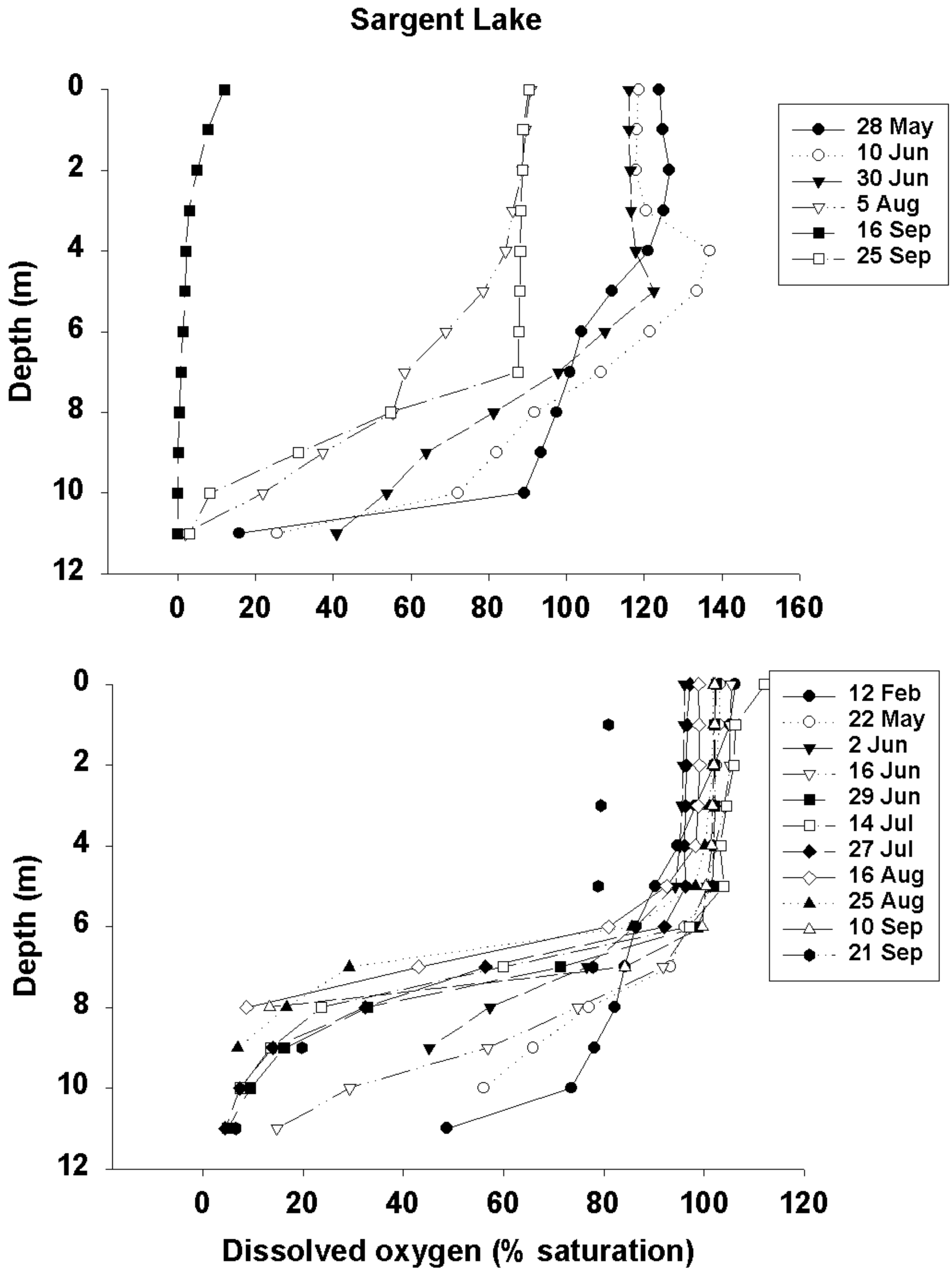


Figure 6-39. pH profiles for Sargent Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

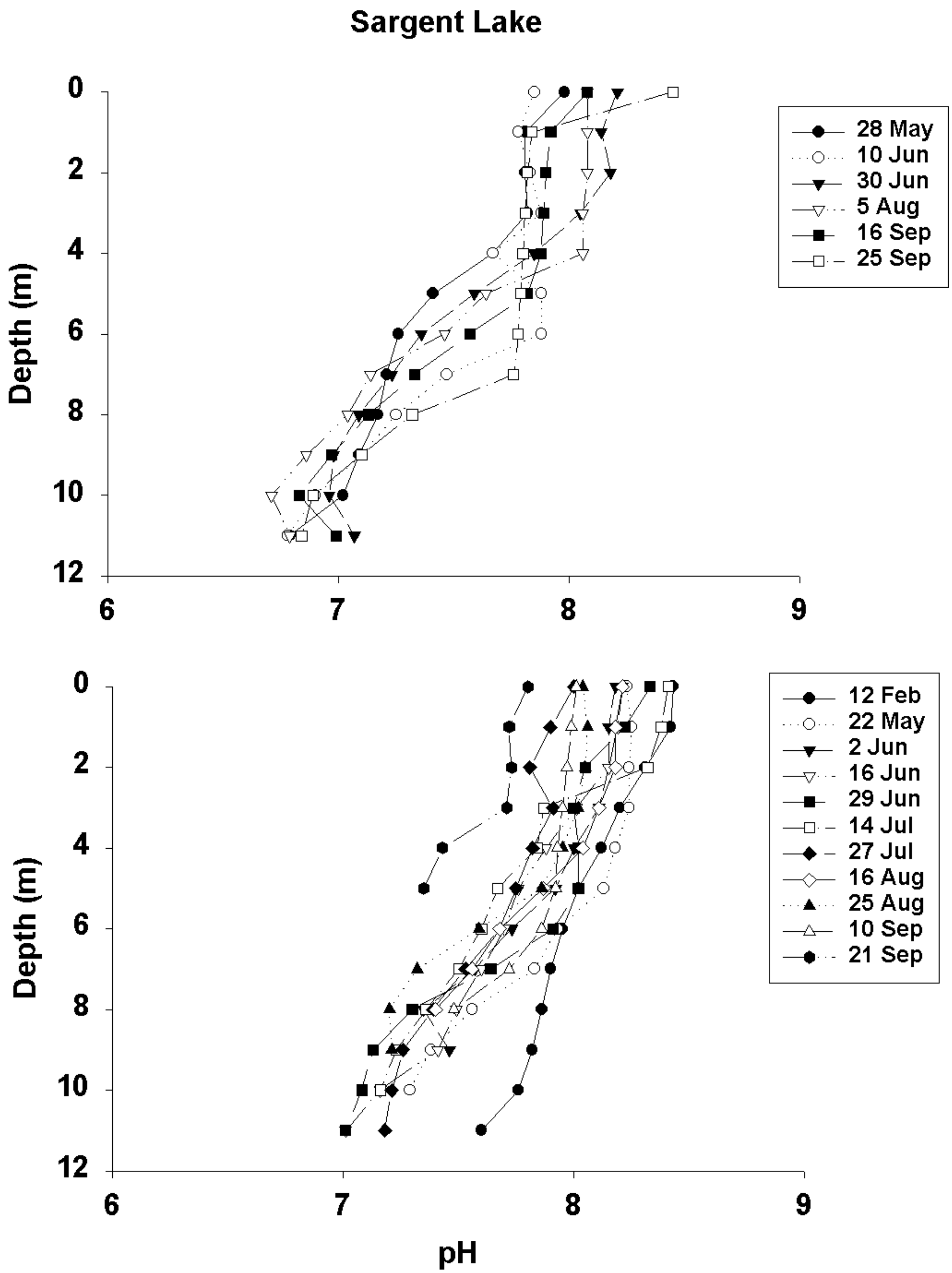


Figure 6-40. Conductivity profiles for Sargent Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

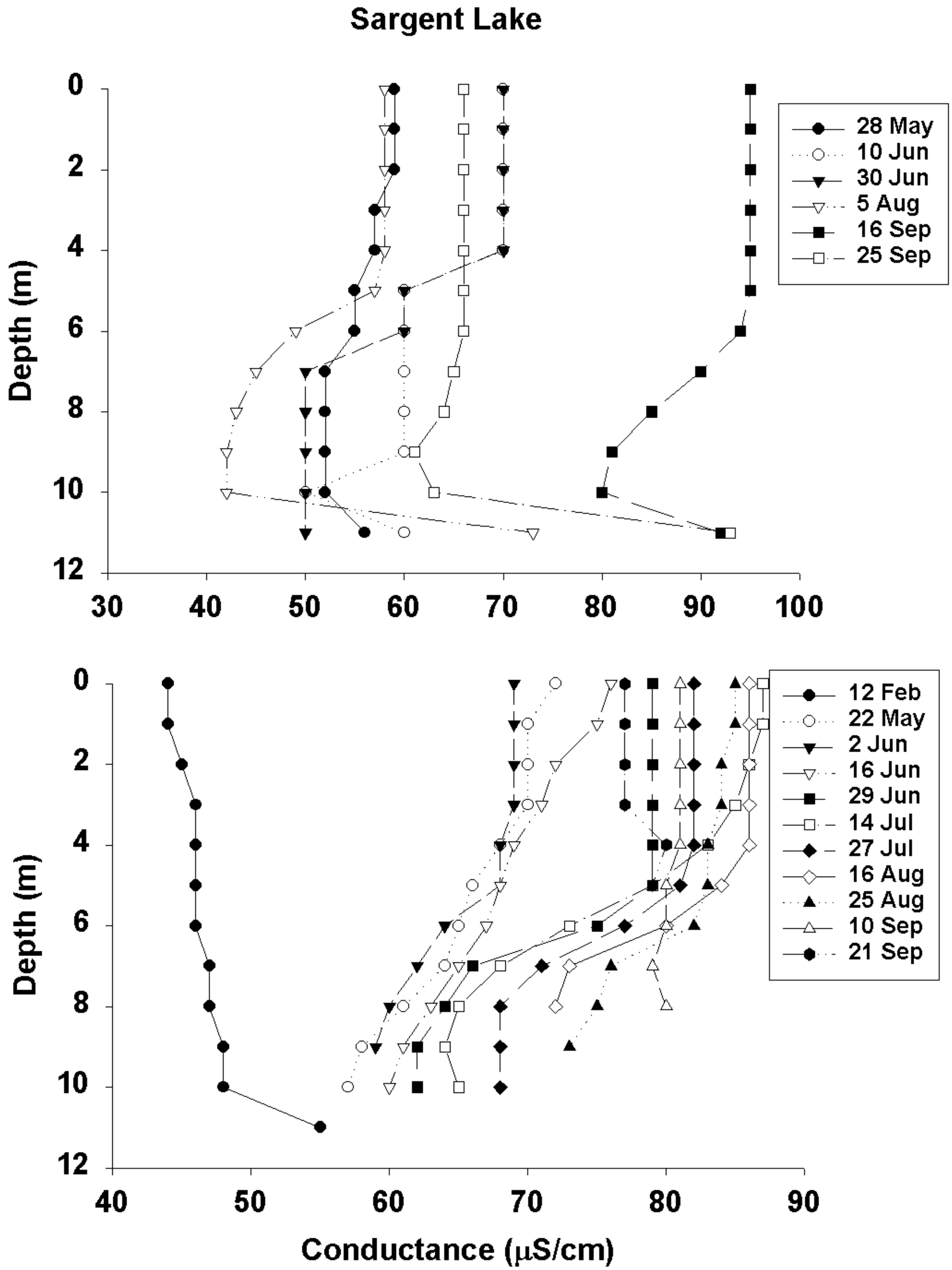


Figure 6-41. Specific conductivity profiles for Sargent Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

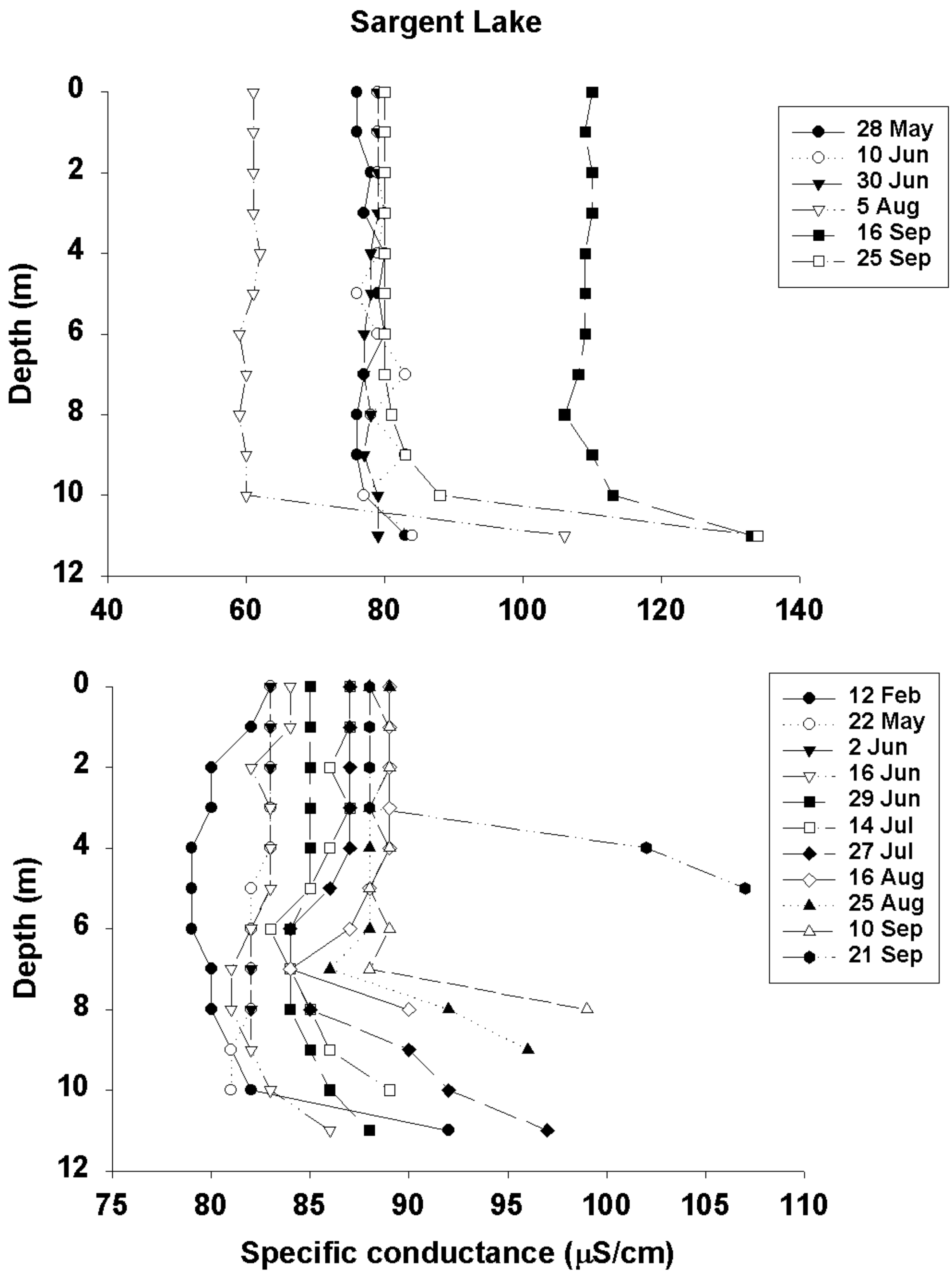


Figure 6-42. Total dissolved solids profiles for Sargent Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

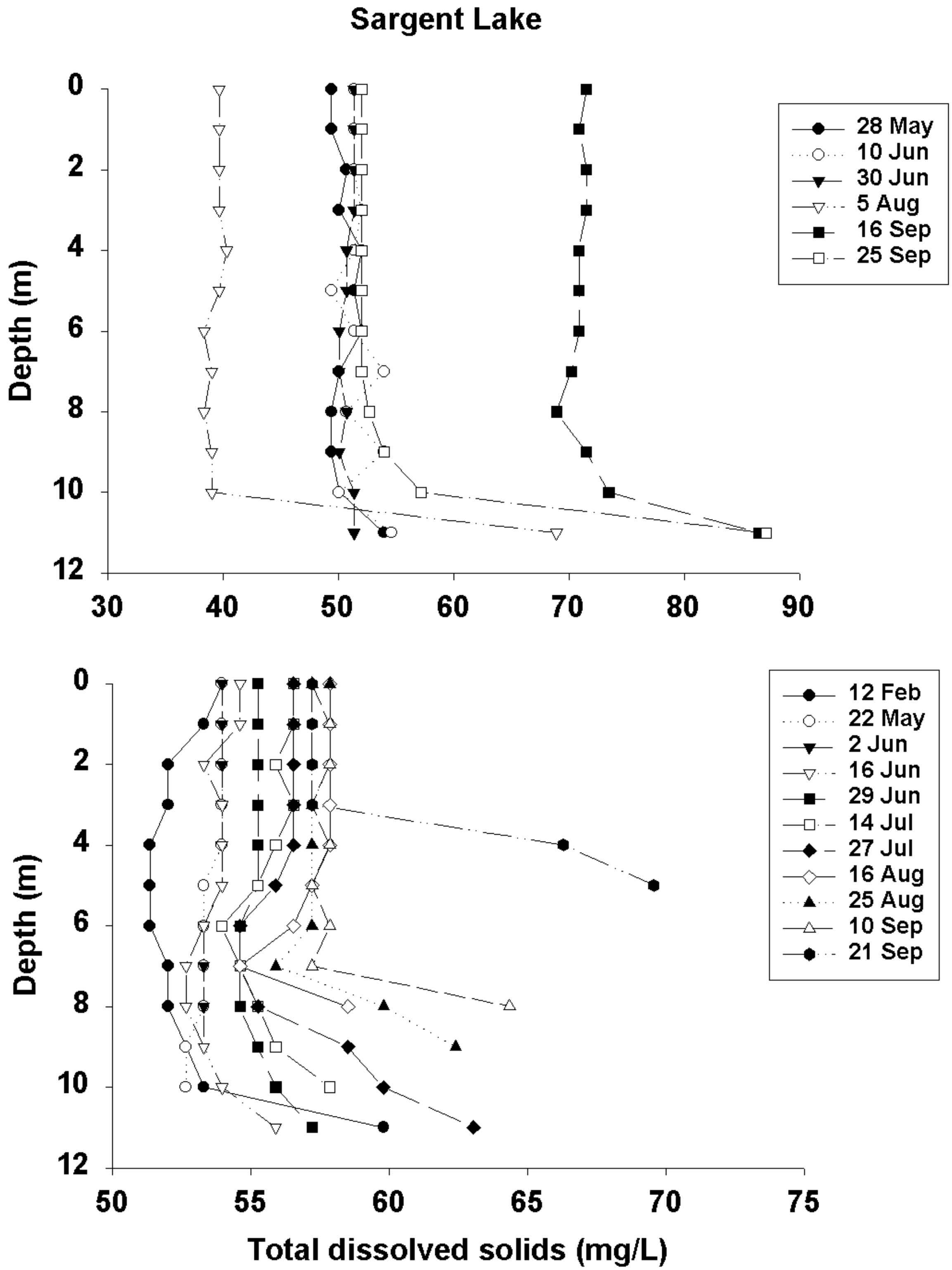


Table 6-11. Means, standard errors (SE), maximums (Max), and minimums (Min) for 1997, 1998, and for 1997-1999 in Sargent Lake, Isle Royale National Park, Michigan. NH4 = ammonium, Ca=calcium, Cl=chloride, Mg=magnesium, N-NO3+NO2=nitrogen as nitrate+nitrite, SiO2=silica, SO_4=sulfate, and P=phosphorus.

	Littoral Alkalinity Mg/L	NH_4 mg/L	Ca mg/L	Cl mg/L	Color	Hardness mg/L	Mg mg/L	$N\text{-}NO_3$+NO_2 mg/L	SiO_2 mg/L	SO_4 mg/L	Total P mg/L	Turbidity NTU
97Mean	39.05	0.12	12.73	1.23	18.75	45.47	3.33	0.04	8.90	3.50	0.03	1.17
97SE	0.31	0.06	2.79	0.38	4.73	6.59	0.30	0.01	0.91	0.44	0.00	0.33
97Max	39.90	0.29	20.90	2.33	25.00	63.72	3.90	0.05	10.00	4.80	0.03	2.10
97Min	38.50	0.05	8.30	0.73	5.00	32.26	2.80	0.01	7.10	2.90	0.02	0.65
98Mean	39.86	0.08	10.08	1.10	19.00	40.90	3.82	0.04	8.28	3.80	0.08	1.29
98SE	0.67	0.03	0.27	0.13	2.92	0.62	0.02	0.01	1.03	0.08	0.03	0.56
98Max	41.30	0.21	10.80	1.40	25.00	42.62	3.90	0.05	10.00	4.00	0.15	3.50
98Min	38.30	0.05	9.20	0.71	10.00	39.03	3.80	0.02	4.30	3.50	0.03	0.46
Mean	38.64	0.09	11.02	1.20	21.00	42.26	3.58	0.04	8.66	3.80	0.05	1.16
SE	0.93	0.03	1.13	0.16	3.06	2.61	0.14	0.00	0.63	0.22	0.02	0.30
Max	41.30	0.29	20.90	2.33	40.00	63.72	3.90	0.05	10.00	5.00	0.15	3.50
Min	30.90	0.03	8.30	0.71	5.00	32.26	2.80	0.01	4.30	2.90	0.02	0.43

	Limnetic Epilimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3$+NO_2	SiO_2	SO_4	Total P	Turbidity
97Mean	38.88	0.12	12.63	0.84	17.50	39.30	3.67	0.05	8.97	3.19	0.03	1.89
97SE	0.68	0.07	2.73	0.09	4.33	2.12	0.30	0.02	0.34	0.11	0.01	1.08
97Max	40.90	0.31	20.70	1.10	25.00	43.70	4.10	0.09	9.40	3.45	0.04	5.00
97Min	38.00	0.05	8.70	0.70	5.00	34.20	3.10	0.01	8.30	3.00	0.02	0.42
98Mean	39.96	0.05	10.42	1.08	17.00	41.59	3.78	0.04	8.92	3.68	0.07	0.49
98SE	0.54	0.00	0.26	0.12	2.00	1.94	0.06	0.01	0.30	0.10	0.03	0.05
98Max	41.30	0.07	10.90	1.30	20.00	43.28	3.90	0.05	9.60	3.80	0.17	0.63
98Min	38.50	0.05	9.70	0.70	10.00	39.46	3.60	0.02	8.00	3.30	0.03	0.34
Mean	38.85	0.08	11.20	1.01	19.00	39.30	3.72	0.05	8.96	3.53	0.05	1.04
SE	0.74	0.03	1.08	0.09	2.56	2.12	0.09	0.01	0.18	0.12	0.02	0.46
Max	41.30	0.31	20.70	1.40	35.00	43.70	4.10	0.09	9.60	4.10	0.17	5.00
Min	33.20	0.05	8.70	0.70	5.00	34.20	3.10	0.01	8.00	3.00	0.02	0.34

	Limnetic Hypolimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3$+NO_2	SiO_2	SO_4	Total P	Turbidity
97Mean	38.18	0.17	13.48	0.88	25.00	42.58	4.00	0.07	11.97	3.19	0.03	2.12
97SE	0.71	0.12	2.44	0.07	7.91	1.93	0.10	0.03	1.53	0.20	0.01	0.68
97Max	39.90	0.52	20.70	1.10	40.00	47.10	4.10	0.15	14.50	3.64	0.05	3.80
97Min	36.50	0.05	10.30	0.79	5.00	38.00	3.80	0.01	9.20	2.70	0.02	0.49
98Mean	40.02	0.07	10.24	1.09	26.00	40.89	3.72	0.04	10.24	3.48	0.03	1.12
98SE	0.58	0.02	0.29	0.12	6.20	0.95	0.06	0.01	0.92	0.13	0.00	0.40
98Max	41.40	0.13	10.90	1.30	45.00	43.28	3.90	0.05	12.80	3.80	0.03	2.50
98Min	38.40	0.05	9.40	0.70	10.00	38.30	3.60	0.02	7.90	3.20	0.03	0.30
Mean	38.68	0.11	11.41	1.07	26.00	41.16	3.79	0.05	10.78	3.49	0.03	1.48
SE	0.70	0.05	1.07	0.10	4.14	1.00	0.07	0.01	0.72	0.18	0.00	0.36
Max	41.40	0.52	20.70	1.70	45.00	47.10	4.10	0.15	14.50	4.80	0.05	3.80
Min	34.00	0.05	9.00	0.70	5.00	36.89	3.50	0.01	7.90	2.70	0.02	0.30

Table 6-12. Values from the littoral zone profiles for Sargent Lake. Means (SE) are given. Most profiles had 1-4 readings. ND=do data available, Sp. Cond.=specific conductance and Cond.=conductance.

Date	5/28/97	6/10/97	6/30/97	8/5/97	9/16/97	9/25/97
Temperature(C)	ND	22.16 (0.30)	21.84 (0.02)	22.42 (0.01)	18.69 (0.01)	15.77 (0.05)
DO(mg/L)	ND	10.20 (0.04)	9.86 (0)	7.98 (0.17)	0.1 (0.02)	9.14 (0.10)
DO(%saturation)	ND	117.1 (0.2)	112.5 (0)	92.1 (1.9)	1.1 (0.2)	92.2 (1.1)
pH	ND	7.63 (0.03)	7.97 (0)	7.82 (0.09)	7.76 (0.01)	7.76 (0.13)
TDS (mg/L)	ND	51 (0)	54 (2)	41 (0)	72 (0)	53 (0)
Sp. Cond. (μS/cm)	ND	78 (1)	83.5 (3.5)	63 (0)	112 (0)	81 (0)
Cond. (μS/cm)	ND	80 (0)	80 (0)	59 (0)	99 (0)	66 (0.)

Date	2/12/98	5/22/98	6/2/98	6/16/98	6/29/98	7/14/98
Temperature(C)	ND	17.94 (0.33)	15.82 (0.23)	22.03 (0.96)	22.22 (0.05)	25.71 (0.41)
DO(mg/L)	ND	9.31 (0.09)	9.25 (0.26)	9.96 (0.09)	12.71 (0.87)	8.50 (0.06)
DO(%saturation)	ND	98.2 (1.4)	93.4 (3.1)	114.0 (1.0)	145.9 (10.2)	104.3 (1.5)
pH	ND	8.04 (0.04)	7.97 (0.02)	8.21 (0.04)	8.21 (0.01)	8.28 (0.06)
TDS (mg/L)	ND	52 (1)	53 (0)	59.8 (1)	56 (0)	57 (0)
Sp. Cond. (μS/cm)	ND	81 (1)	82 (0)	92 (1)	86 (1)	89 (0)
Cond. (μS/cm)	ND	70 (1)	68 (1)	87 (3)	82 (0)	90 (1)

Date	7/27/98	8/16/98	8/25/98	9/10/98	9/21/98
Temperature(C)	22.2 (0.01)	23.44 (0.02)	23.27 (0.04)	20.49 (0.07)	17.75 (0.03)
DO(mg/L)	8.105 (0.01)	8.29 (0.03)	8.95 (0.01)	9.74 (0.04)	8.43 (0.02)
DO(%saturation)	93.05 (0.1)	97.5 (0.4)	104.9 (0.1)	108.1 (0.6)	88.6 (0.3)
pH	7.83 (0.01)	8.14 (0.02)	8.02 (0.01)	8.11 (0.03)	7.78 (0.03)
TDS (mg/L)	58.5 (0)	59 (0)	58 (0)	58 (0)	57 (0)
Sp. Cond. (μS/cm)	90 (0)	91 (0)	90 (0)	90 (0)	89 (0)
Cond. (μS/cm)	85 (0)	88 (0)	87 (0)	82 (0)	77 (0)

Siskiwit Lake

Both clinograde temperature (Figure 6-43) and dissolved oxygen (Figure 6-44) curves indicated that Siskiwit Lake was stratified by June in both 1997 and 1998. Stratification had already begun by late May in both years. The lake remained completely stratified on the last sampling date. Overturn did occur, likely in October or November, as evident by the February curve in 1998. Both the dissolved oxygen (Figs. 6-44) and oxygen saturation (Figs. 6-45) curves suggest high primary production throughout a large area of the water column, corresponding to the metalimnion area, during July and early August.

The water chemistry is summarized in Tables 6-13 and 6-14. No parameters in the epilimnion were significantly different between years (all t-test P-values > 0.06; P-critical = 0.007 after Bonferonni adjustment). Ammonia was consistently below detection limit except for two dates, which indicates very low concentrations. In general, within-year variation was minimal as suggested by SEM of <20% of the means for all parameters.

The water chemistry in the hypolimnion showed similar trends as the epilimnion (Table 6-13). Only marginally significant differences were observed (P = 0.4 mean alkalinity 1998 > 1997 and P = 0.05 mean nitrogen as nitrate + nitrite 1997 > 1998). Ammonia was below detection limits in all years, except for a few dates.

Water samples from the littoral zone showed similar trends as was observed in the hypolimnion. No parameters were significantly different (all P-values > 0.21). Ammonia was only detected in one sample but was consistently below detection limit in other samples.

When parameters were compared between epilimnion and hypolimnion (paired tests), silica was significantly higher in the hypolimnion than in the epilimnion (P = 0.005) and alkalinity was marginally higher in the epilimnion than the hypolimnion (P = 0.04)). All other parameters showed no significant differences. There was also no significant difference between the epilimnion and littoral water samples for all parameters. Silica was significantly higher in the hypolimnion compared to the littoral (P = 0.003), and sulfate was marginally higher (P = 0.08).

Limnology

Based on the calcium, magnesium, specific conductance, and hardness data, Siskiwit Lake is a mildly hardwater lake. The alkalinity and pH values suggest that the lake is slightly alkaline. Chloride ion concentrations were low and again should illicit no concern.

Nitrogen and phosphorus levels were indicative of an oligotrophic (possibly slightly mesotrophic) lake. Nearly all nutrients were below detection levels.

Silica is reduced in the littoral probably by increased production by diatoms. This littoral production could have caused the difference in silica concentrations between the hypolimnion and the littoral, as was observed in many other lakes in this study. Sulfates did not show significant differences with depth or among zones. Siskiwit Lake is an extremely deep lake and, although vertical temperature profiles indicated the presence of a thermocline around 8-12 meters, vertical oxygen profiles showed that the lake is well-oxygenated even to 32 meter depth. Thus, sulfates would not be expected to be different between zones of different depth.

Siskiwit Lake is in good condition based on the water chemistry data. Nutrient concentrations indicate that the lake is oligotrophic. The outstanding physical feature of the lake is its extreme depth.

Figure 6-43. Temperature profiles for Siskiwit Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

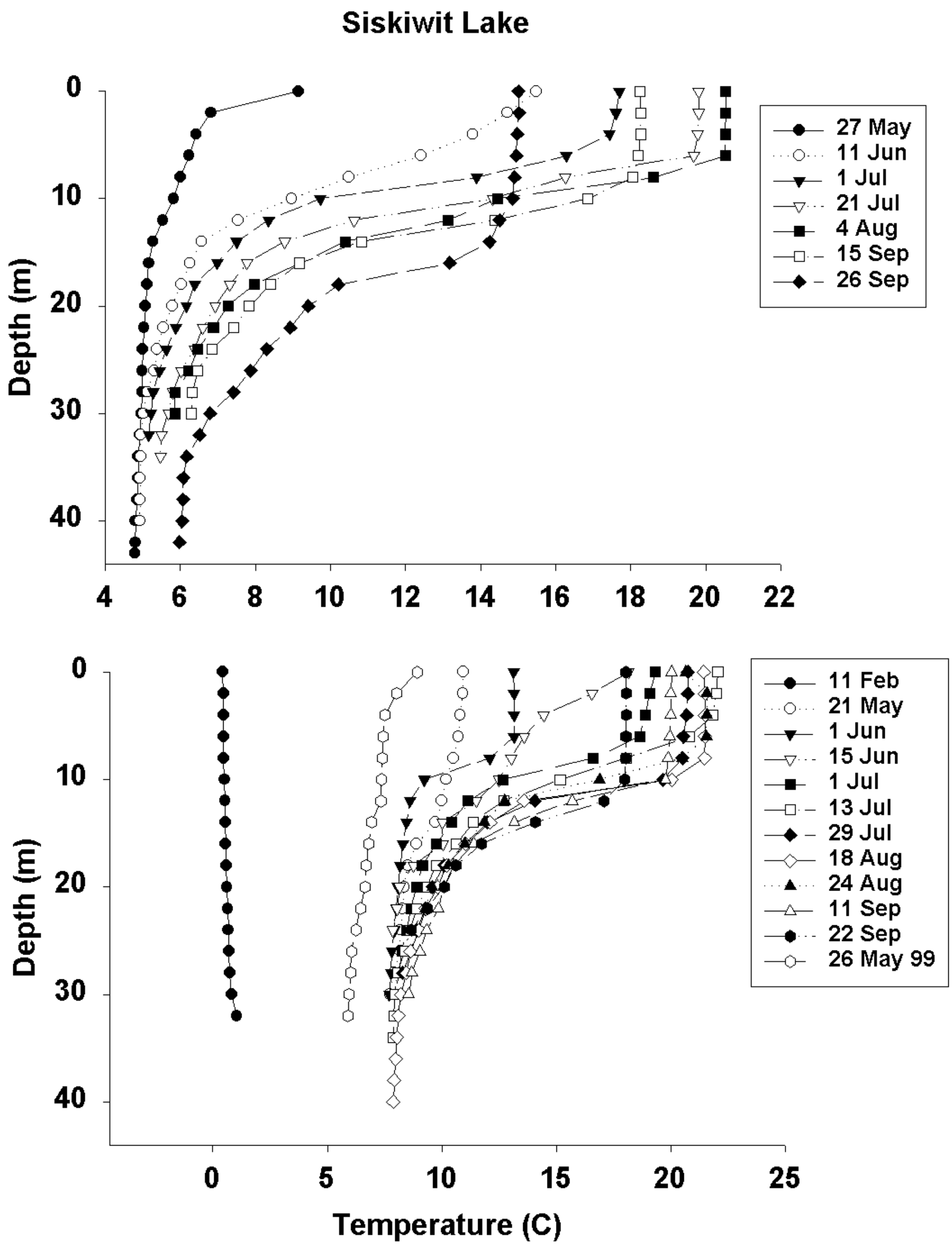


Figure 6-44. Dissolved oxygen profiles for Siskiwit Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

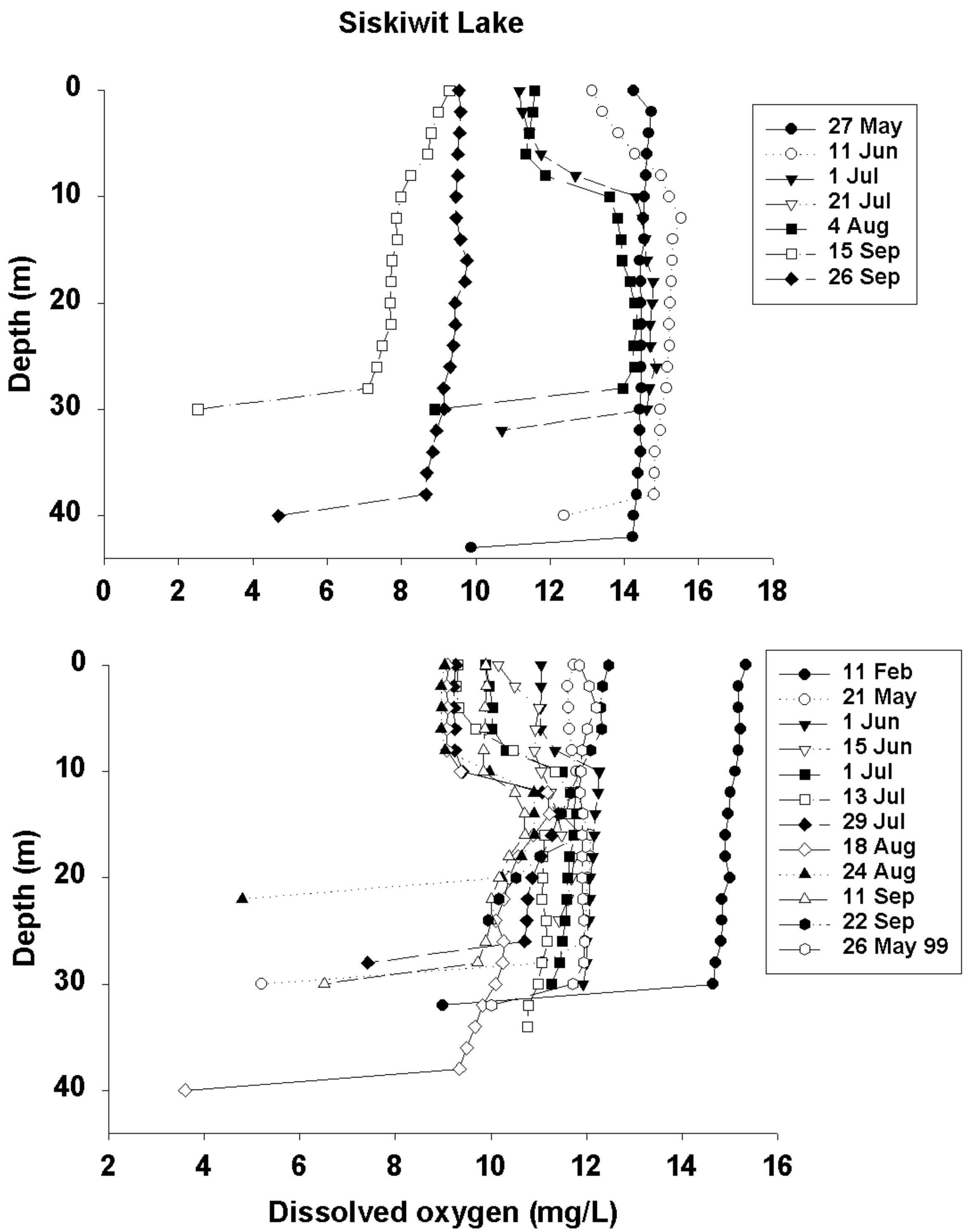


Figure 6-45. Oxygen saturation calculated from measured water temperature and dissolved oxygen in Siskiwit Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

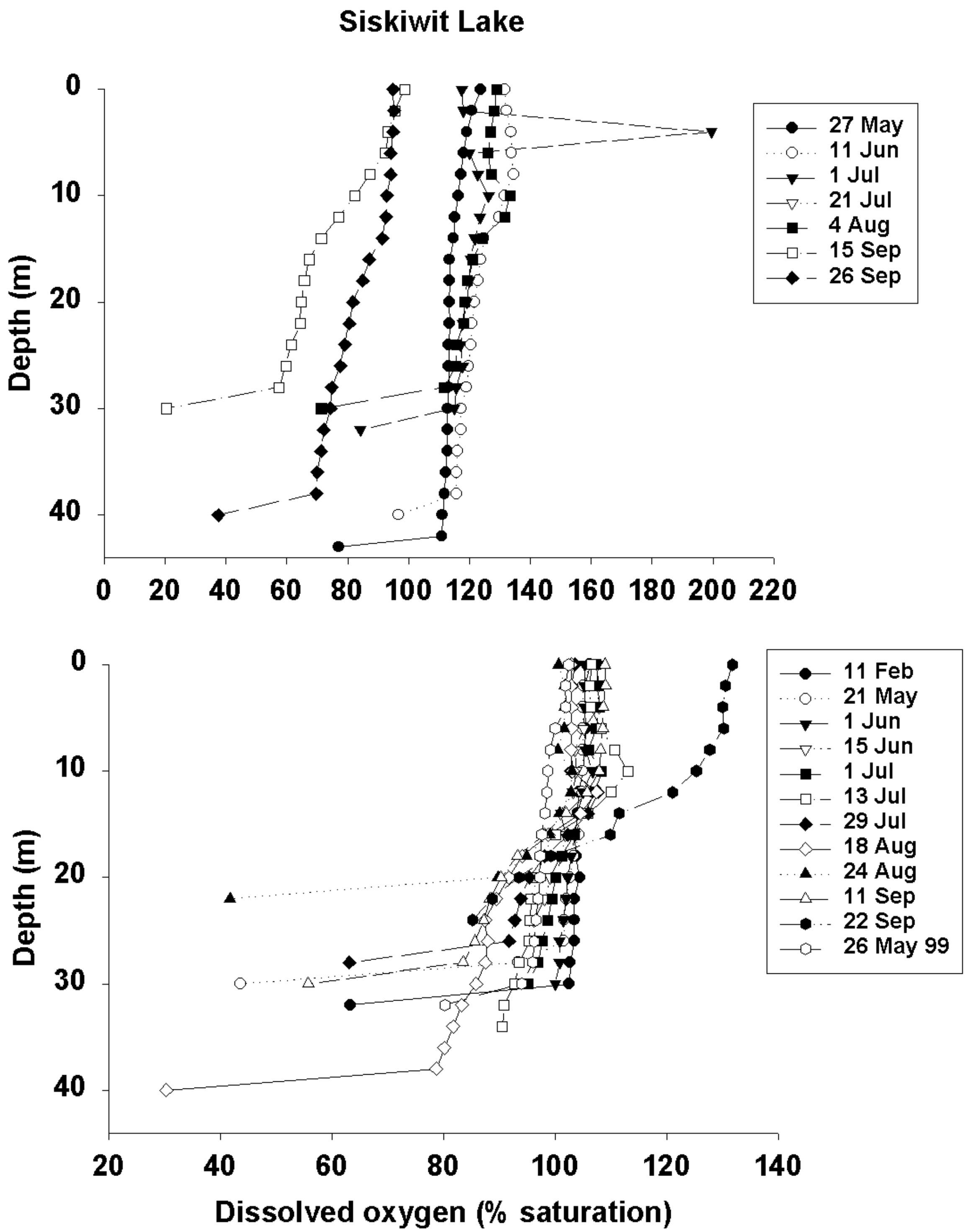
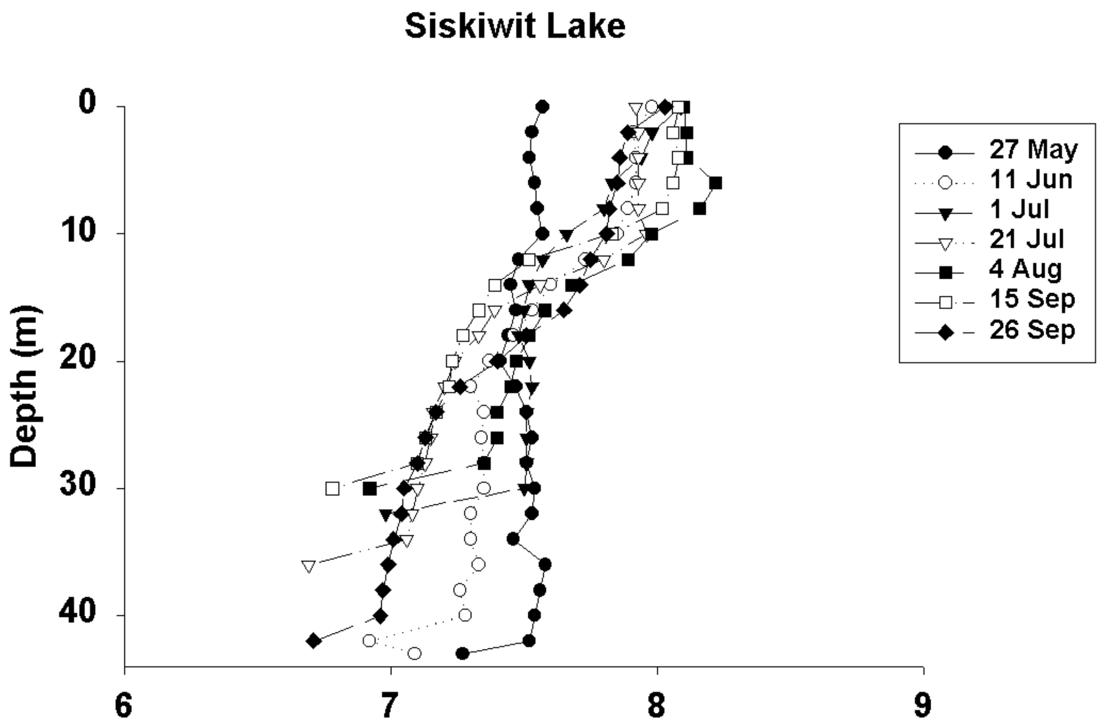


Figure 6-46. pH profiles for Siskiwit Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.



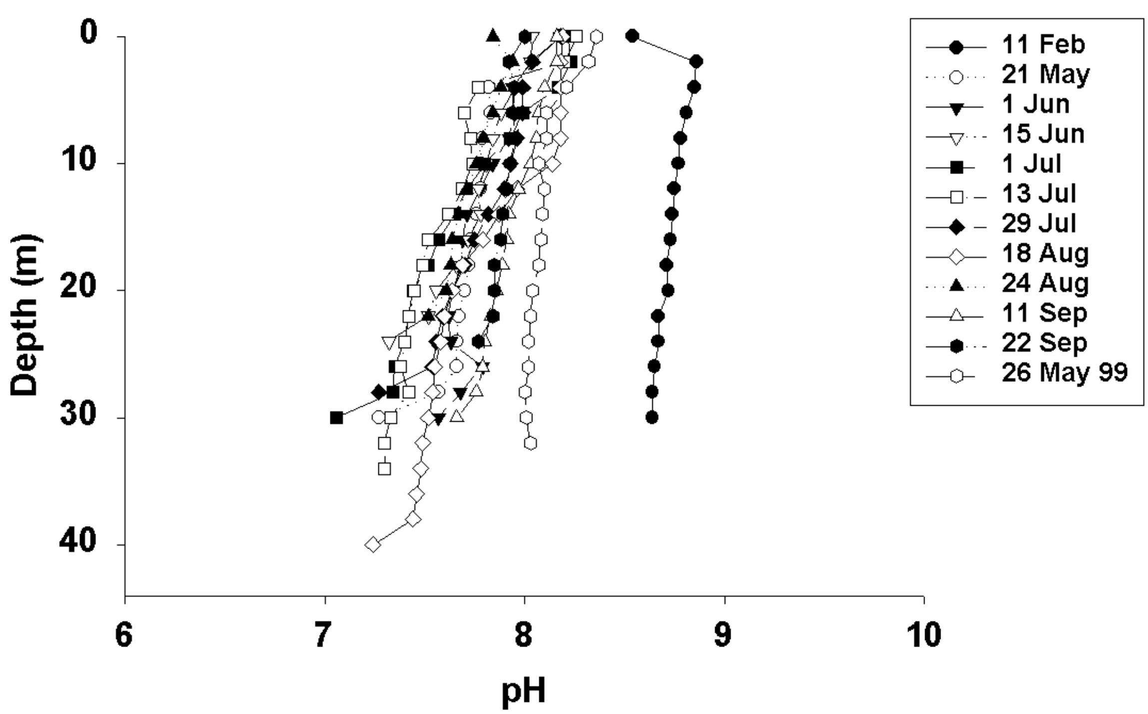


Figure 6-47. Conductivity profiles for Siskiwit Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

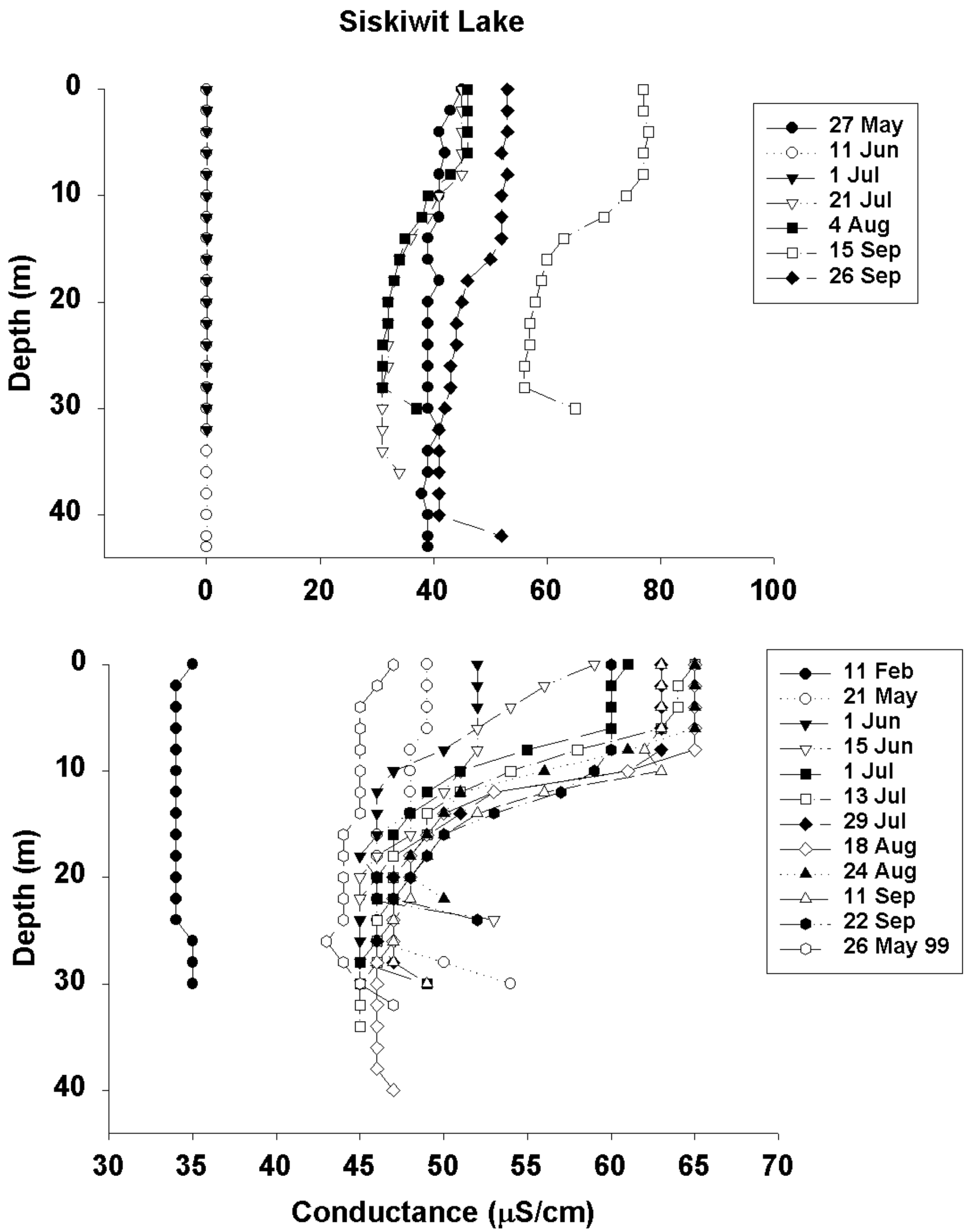


Figure 6-48. Specific conductivity profiles for Siskiwit Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

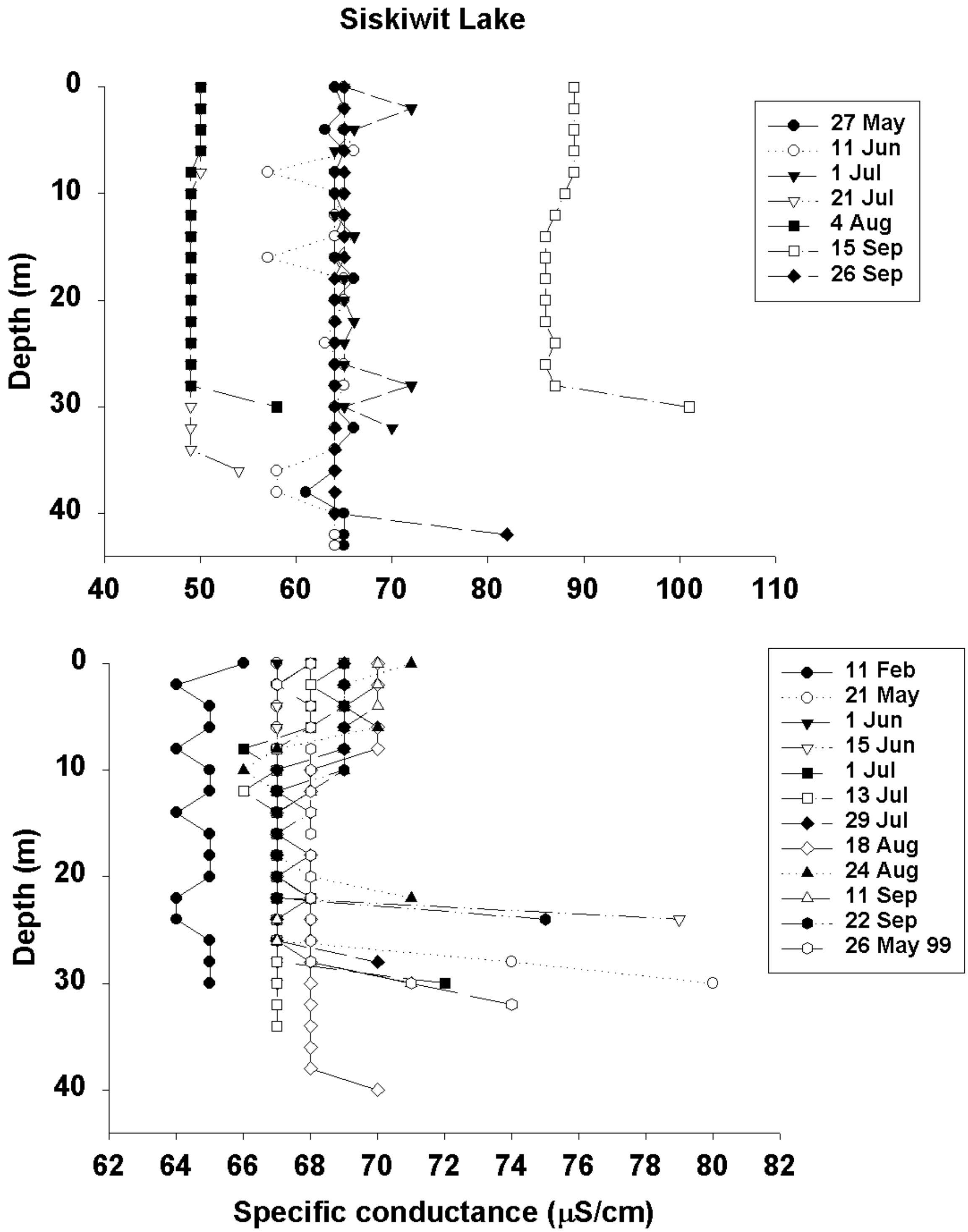


Figure 6-49. Total dissolved solids profiles for Siskiwit Lake, Isle Royale National Park, Michigan. Top panel is 1997 and bottom panel is 1998/1999.

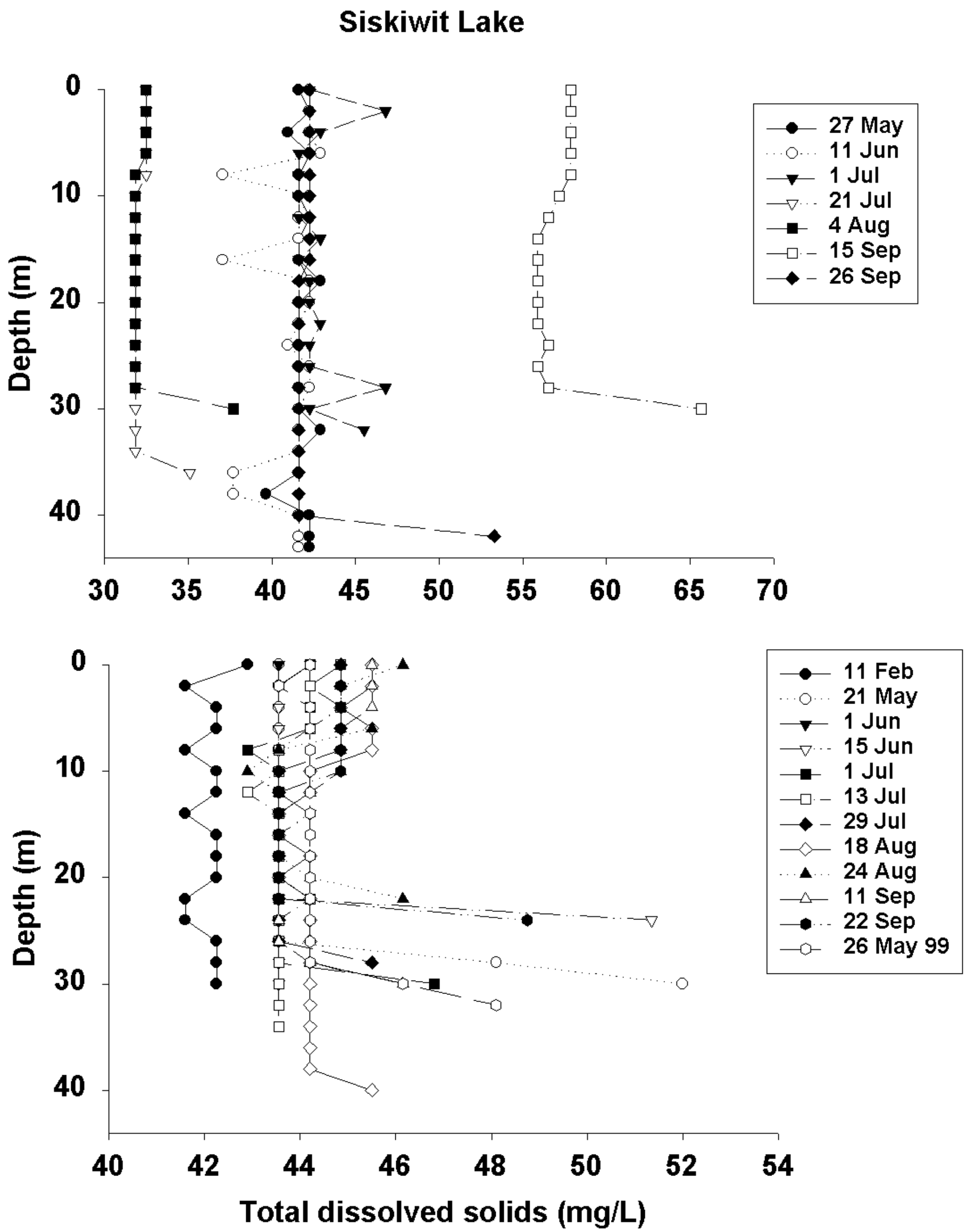


Table 6-13. Means, standard errors (SE), maximums (Max), and minimums (Min) for 1997, 1998, and for 1997-1999 in Siskiwit Lake, Isle Royale National Park, Michigan. NH4 = ammonium, Ca=calcium, Cl=chloride, Mg=magnesium, N-NO3+NO2=nitrogen as nitrate+nitrite, SiO2 =silica, SO_4=sulfate, and P=phosphorus.

	Littoral Alkalinity mg/L	NH_4 mg/L	Ca mg/L	Cl mg/L	Color	Hardness mg/L	Mg mg/L	$N\text{-}NO_3+NO_2$ mg/L	SiO_2 mg/L	SO_4 mg/L	Total P mg/L	Turbidity NTU
97Mean	28.65	0.09	11.70	0.65	8.75	38.79	2.33	0.07	3.60	4.31	0.02	0.46
97SE	0.38	0.05	2.57	0.11	1.25	6.27	0.07	0.02	0.36	0.15	0.00	0.18
97Max	29.70	0.23	19.40	0.96	10.00	57.50	2.50	0.13	4.30	4.75	0.03	1.00
97Min	27.90	0.03	8.70	0.50	5.00	30.78	2.20	0.03	3.10	4.10	0.01	0.18
98Mean	29.84	0.05	9.14	0.86	5.50	32.38	2.32	0.04	3.44	4.38	0.03	0.24
98SE	0.46	0.00	0.17	0.11	1.22	0.38	0.06	0.01	0.15	0.08	0.00	0.03
98Max	30.90	0.06	9.70	1.10	10.00	33.28	2.50	0.05	3.90	4.60	0.03	0.35
98Min	28.30	0.05	8.80	0.52	2.50	31.45	2.20	0.02	3.00	4.10	0.03	0.16
Mean	29.20	0.07	10.13	0.81	7.25	34.89	2.33	0.05	3.56	4.34	0.02	0.32
SE	0.34	0.02	1.04	0.08	0.95	2.53	0.04	0.01	0.14	0.07	0.00	0.08
Max	30.90	0.23	19.40	1.20	10.00	57.50	2.50	0.13	4.30	4.75	0.03	1.00
Min	27.90	0.03	8.70	0.50	2.50	30.78	2.20	0.01	3.00	4.10	0.01	0.16

	Limnetic Epilimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3+NO_2$	SiO_2	SO_4	Total P	Turbidity
97Mean	28.75	0.06	11.40	0.63	8.75	31.77	2.30	0.10	3.23	4.28	0.03	0.43
97SE	0.35	0.02	2.52	0.12	1.25	1.42	0.21	0.03	0.29	0.12	0.00	0.11
97Max	29.50	0.12	18.90	0.96	10.00	34.43	2.60	0.16	3.70	4.61	0.04	0.70
97Min	27.90	0.02	8.00	0.43	5.00	27.80	1.90	0.05	2.70	4.10	0.01	0.18
98Mean	30.14	0.06	9.12	0.85	6.50	32.90	2.46	0.06	3.52	4.38	0.03	0.21
98SE	0.61	0.01	0.17	0.11	1.50	0.55	0.04	0.01	0.14	0.07	0.01	0.01
98Max	31.80	0.08	9.70	1.10	10.00	34.52	2.60	0.12	4.00	4.50	0.05	0.25
98Min	28.10	0.05	8.80	0.52	2.50	31.86	2.40	0.05	3.20	4.10	0.02	0.18
Mean	29.42	0.06	9.97	0.80	7.75	32.27	2.40	0.08	3.49	4.32	0.03	0.29
SE	0.40	0.01	1.00	0.09	0.95	0.62	0.07	0.01	0.14	0.06	0.00	0.05
Max	31.80	0.12	18.90	1.20	10.00	34.52	2.60	0.16	4.10	4.61	0.05	0.70
Min	27.90	0.02	8.00	0.43	2.50	27.80	1.90	0.05	2.70	4.10	0.01	0.16

	Limnetic Hypolimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3+NO_2$	SiO_2	SO_4	Total P	Turbidity
97Mean	28.05	0.06	11.85	0.63	8.75	32.70	2.40	0.22	4.40	4.32	0.02	0.30
97SE	0.28	0.01	2.48	0.11	1.25	0.57	0.06	0.07	0.00	0.10	0.00	0.07
97Max	28.60	0.09	19.30	0.94	10.00	33.52	2.50	0.34	4.40	4.56	0.03	0.41
97Min	27.30	0.05	9.30	0.41	5.00	31.00	2.30	0.07	4.40	4.10	0.01	0.14
98Mean	30.00	0.06	8.98	0.89	6.00	32.14	2.36	0.06	3.98	4.40	0.02	0.20
98SE	0.66	0.01	0.26	0.13	1.70	0.89	0.06	0.02	0.07	0.08	0.00	0.03
98Max	31.40	0.11	9.80	1.20	10.00	34.77	2.50	0.13	4.20	4.60	0.03	0.30
98Min	27.80	0.04	8.30	0.52	2.50	29.78	2.20	0.03	3.80	4.10	0.02	0.15
Mean	29.12	0.06	10.14	0.81	7.50	32.45	2.39	0.14	4.14	4.35	0.02	0.23
SE	0.45	0.01	1.03	0.10	1.05	0.48	0.04	0.04	0.08	0.06	0.00	0.03
Max	31.40	0.11	19.30	1.20	10.00	34.77	2.50	0.34	4.40	4.60	0.03	0.41
Min	27.30	0.04	8.30	0.41	2.50	29.78	2.20	0.03	3.80	4.10	0.01	0.13

Table 6-14. Values from the littoral zone profiles Siskiwit Lake. Means (SE) are given. Most profiles had 1-4 readings. ND=no data available, Sp. Cond.=specific conductance and Cond.=conductance.

Date	5/27/97	6/11/97	7/1/97	7/21/97	8/4/97	9/15/97	9/26/97
Temperature(C)	ND	15.27 (0.48)	18.52 (0.48)	19.93 (0.01)	20.63 (0.01)	18.34 (0.03)	15.03 (0.04)
DO(mg/L)	ND	13.28 (0.13)	11.17 (0.06)	ND	11.55 (0.11)	2.05 (0.13)	9.61 (0.06)
DO(%saturation)	ND	132.4 (0.4)	119.3 (0.4)	ND	128.7 (1.2)	21.8 (1.4)	95.4 (0.6)
pH	ND	7.83 (0.02)	7.99 (0.02)	8.00 (0.03)	8.12 (0.01)	8.05 (0.00)	7.82 (0.01)
TDS (mg/L)	ND	45 (1)	42 (0)	32 (0)	32 (0)	57 (0)	42 (0)
Sp. Cond. (μS/cm)	ND	69 (2)	65 (1)	50 (0)	50 (0)	88 (0)	65 (0)
Cond. (μS/cm)	ND	0 (0)	0 (0)	45 (0)	46 (0)	77 (0)	52 (0)

Date	2/11/98	5/21/98	6/1/98	6/15/98	7/1/98	7/13/98	7/29/98
Temperature(C)	ND	11.44 (0.35)	13.05 (0.01)	15.03 (0)	19.29 (0.08)	22.15 (0.28)	20.92 (0.01)
DO(mg/L)	ND	11.875 (0.04)	11.15 (0.01)	11.03 (0)	107.05 (0.25)	15.58 (1.91)	9.37 (0.06)
DO(%saturation)	ND	108.8 (0.5)	106.0 (0.1)	109.4 (0)	9.87 (0.0)	178.6 (20.9)	105.0 (0.6)
pH	ND	8.32 (0.04)	8.09 (0.03)	8.22 (0)	8.17 (0.01)	8.45 (0.02)	8.19 (0.03)
TDS (mg/L)	ND	43.55 (0)	43 (0)	44.2 (0)	44 (0)	44 (0)	44 (0)
Sp. Cond. (μS/cm)	ND	67 (0)	67 (0)	68 (0)	68 (0)	68 (1)	69 (0)
Cond. (μS/cm)	ND	49.5 (0.5)	52 (0)	55 (0)	60 (1)	64 (1)	64 (0)

Date	8/18/98	8/24/98	9/11/98	9/22/98	5/26/99
Temperature(C)	21.69 (0.04)	ND	20.25 (0.09)	17.90 (0.03)	9.22 (0.63)
DO(mg/L)	105.15 (0.14)	ND	9.99 (0.02)	14.49 (0.13)	12.06 (0.08)
DO(%saturation)	9.2 (0)	ND	110.5 (0.4)	152.8 (1.2)	104.92 (2.2)
pH	8.29 (0.01)	ND	8.22 (0.01)	7.84 (0.02)	8.54 (0.01)
TDS (mg/L)	45 (0)	ND	45 (0)	44 (0)	43 (0)
Sp. Cond. (μS/cm)	69 (1)	ND	69 (0)	69 (0)	67 (0)
Cond. (μS/cm)	65 (0)	ND	63 (0)	60 (0)	47 (1)

Locator Lake

Locator Lake was stratified during the entire sampling period in both 1997 and 1998. Temperature profiles exhibited typical clinograde curves of a stratified lake (Figure 6-50). Dissolved oxygen profiles were slightly positive orthograde curves (Figure 6-51), which suggests that algae were productive at the epilimnion-metalimnion interface during the day. The oxygen saturation curves were clinograde (Figure 6-52) and did not reflect primary production at this interface, although some algal activity was obvious in the dissolved oxygen profiles.

The water chemistry in the three zones is summarized in Table 6-15 and 6-16. No parameters were significantly different between years (all t-test P-values > 0.01; P-critical = 0.007 after Bonferonni adjustment), although total phosphorus (P = 0.032) and turbidity (P = 0.011) were slightly different with both parameters being higher in 1997 than in 1998. Silica could not be compared because it was added to the analysis later in the season. In general, within-year variation was minimal as suggested by SEM of <20% of the means for all parameters.

The water chemistry in the hypolimnetic zone did not show much between-year variation (Table 6-15). Only chloride was slightly higher in 1998 compared to 1997 (P = 0.024), whereas all other parameters were not significantly different (all P-values > 0.05). Between-year testing could not be done on magnesium or silica. Ammonia was consistently below detection limit, which indicates very low concentrations.

Water samples from the littoral zone showed similar trends as was observed in the hypolimnion. Only chloride was slightly higher in 1998 compared to 1997 (P = 0.013), whereas all other parameters were not significantly different (all P-values > 0.05). Ammonia was consistently below detection limit, which indicates very low concentrations.

When parameters were compared between epilimnion and hypolimnion (paired tests), no parameter showed significant differences. There was also no significant difference between the epilimnion and littoral water samples for all parameters. However, alkalinity (P = 0.002) and silica (P < 0.001) were significantly higher in the hypolimnion compared to the littoral.

Limnology

Locator Lake is a soft water lake characterized by low concentrations of calcium and magnesium. The alkalinity in the lake was also low and indicative of rather soft water. The specific conductance, which can indicate the presence of cations, was also low.

Nutrient levels in all sections of the lake were generally low. Total phosphorus and nitrate-nitrite nitrogen concentrations were usually below detection levels. Ammonia was always below detection levels. The lack of ammonia accumulating in the hypolimnion would suggest that productivity in the lake is low, which agrees with the other nutrient data.

Silica concentrations were also low in all sections of the lake, especially the littoral zone. Low silica is indicative of high productivity by the diatoms, but the other chemistry results suggest that this is not the case. Sulfates were at reasonable levels throughout the lake.

Based on the water chemistry data alone, Locator Lake is an oligotrophic lake. The low nutrient concentrations indicate that there are no immediate concerns about lake eutrophication.

Figure 6-50. Temperature profile for Locator Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

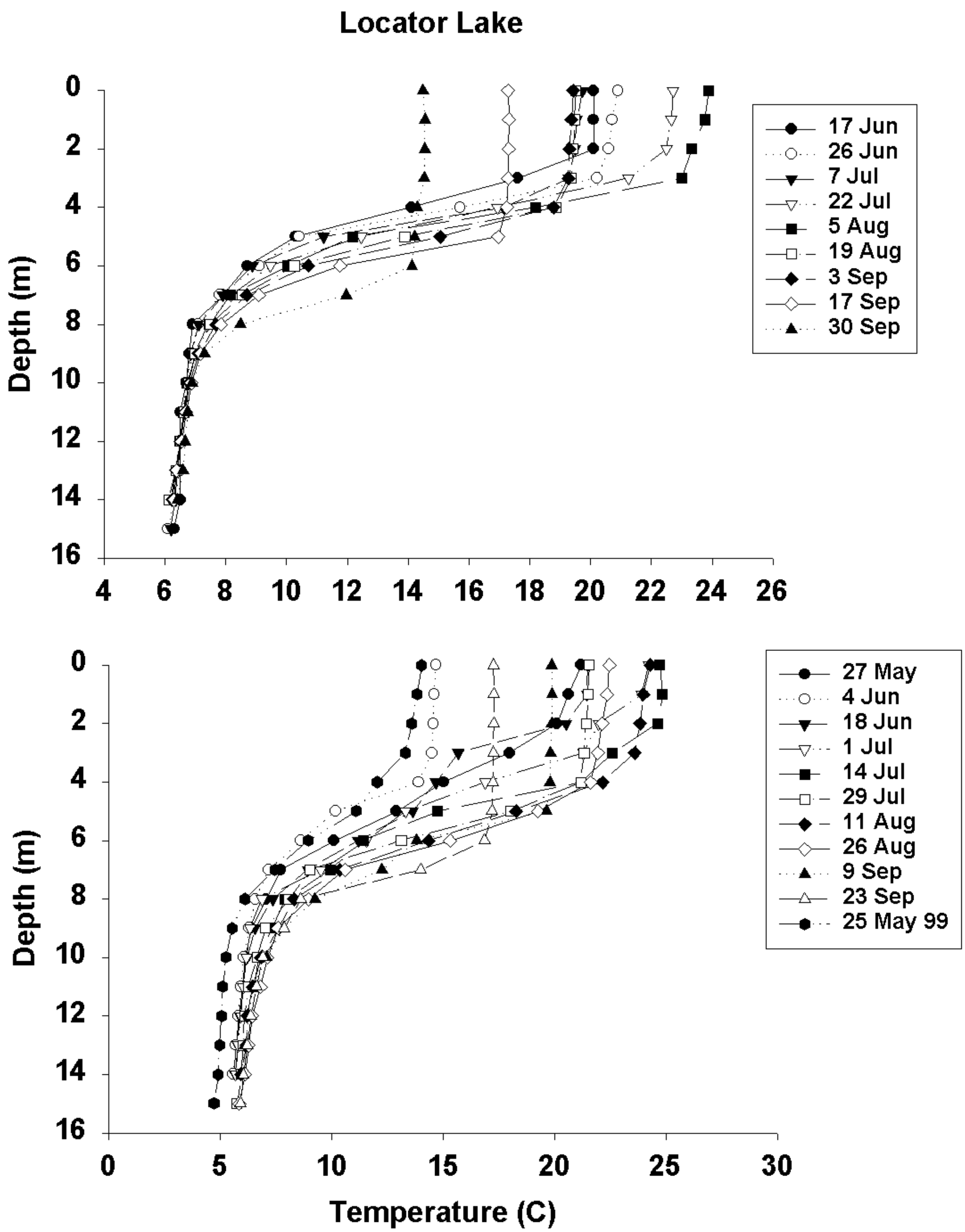


Figure 6-51. Dissolved oxygen profile for Locator Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

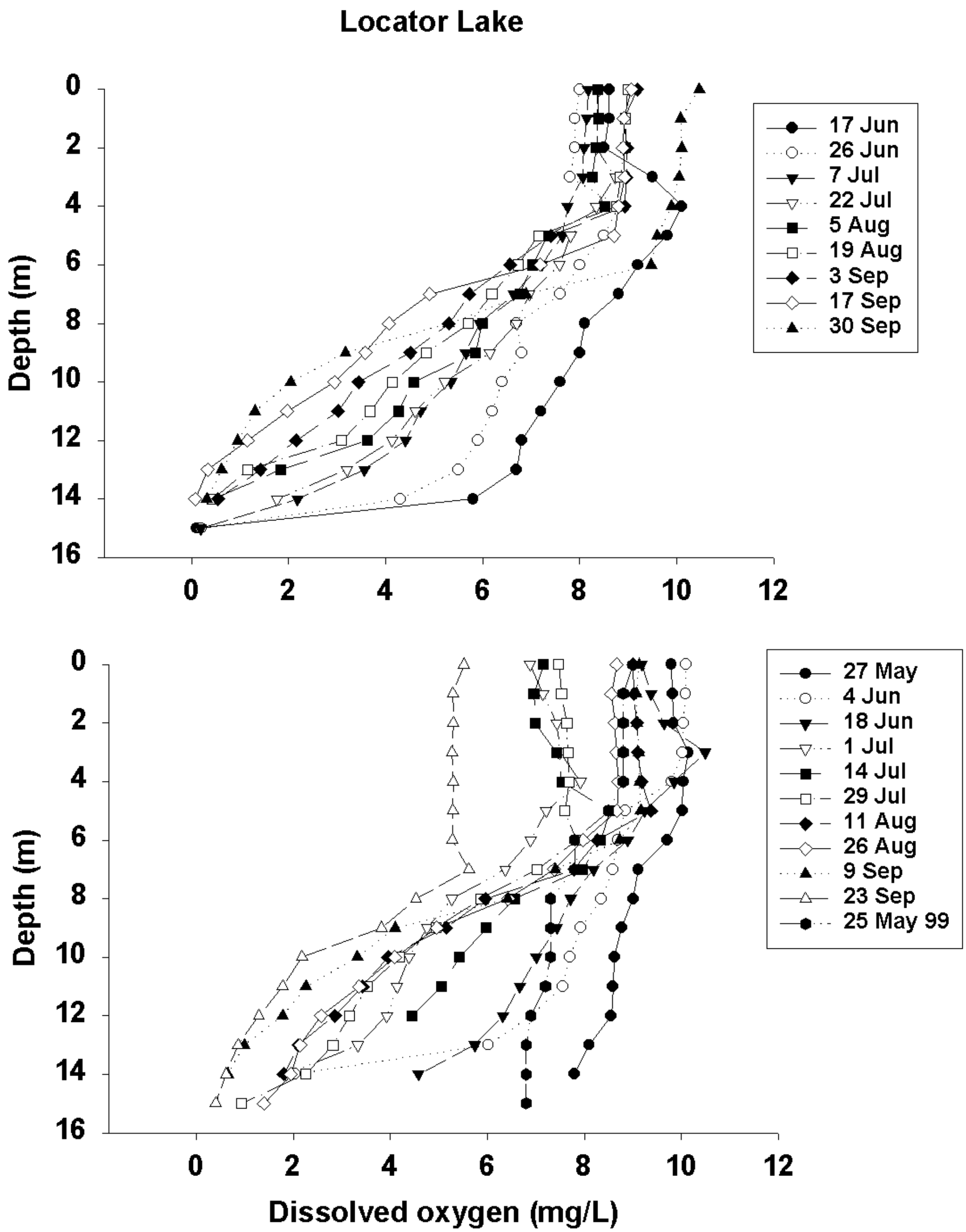


Figure 6-52. Oxygen saturation calculated from measured water temperature and dissolved oxygen in Locator Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

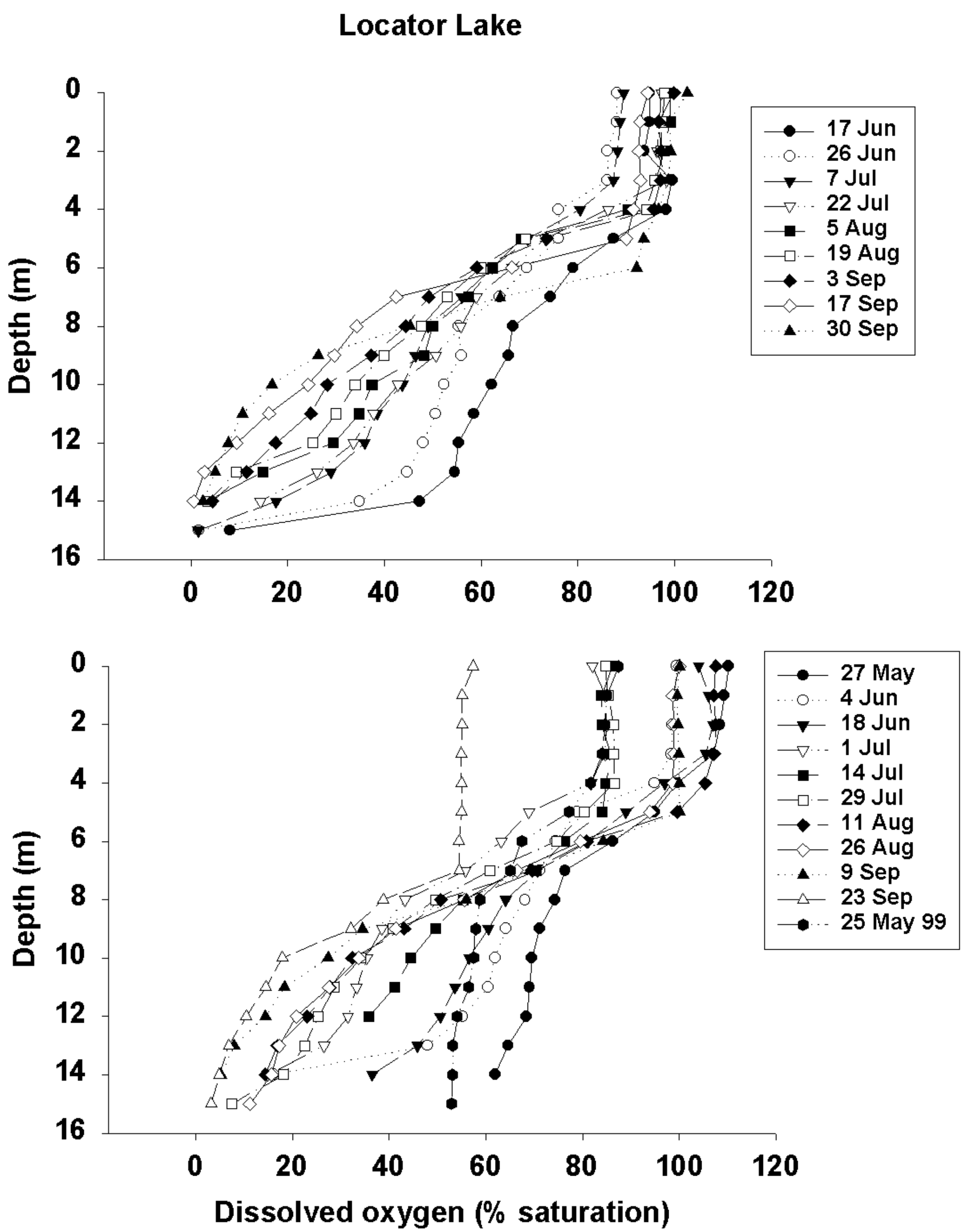


Figure 6-53. pH profiles for Locator Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

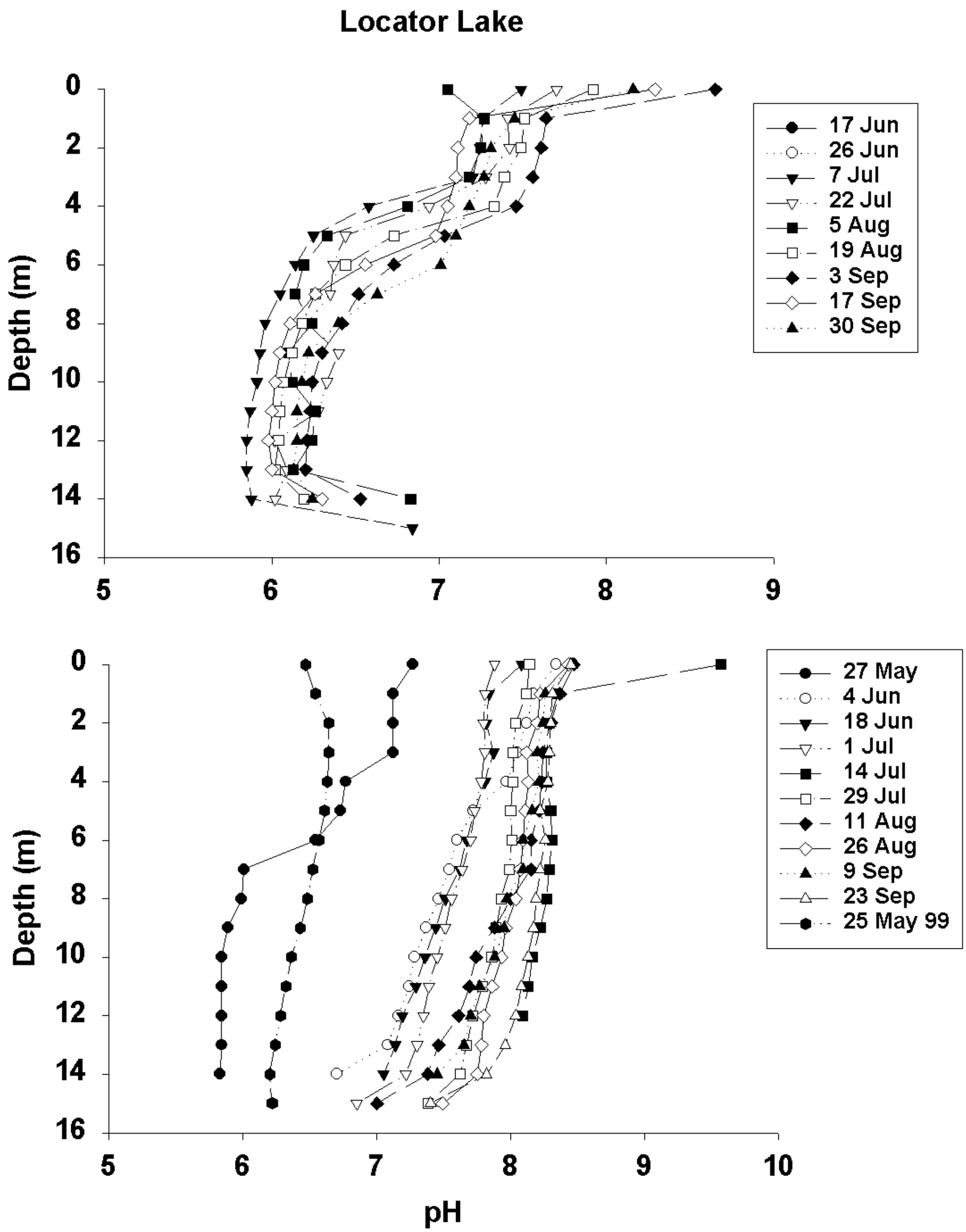


Figure 6-54. Conductivity profiles for Locator Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

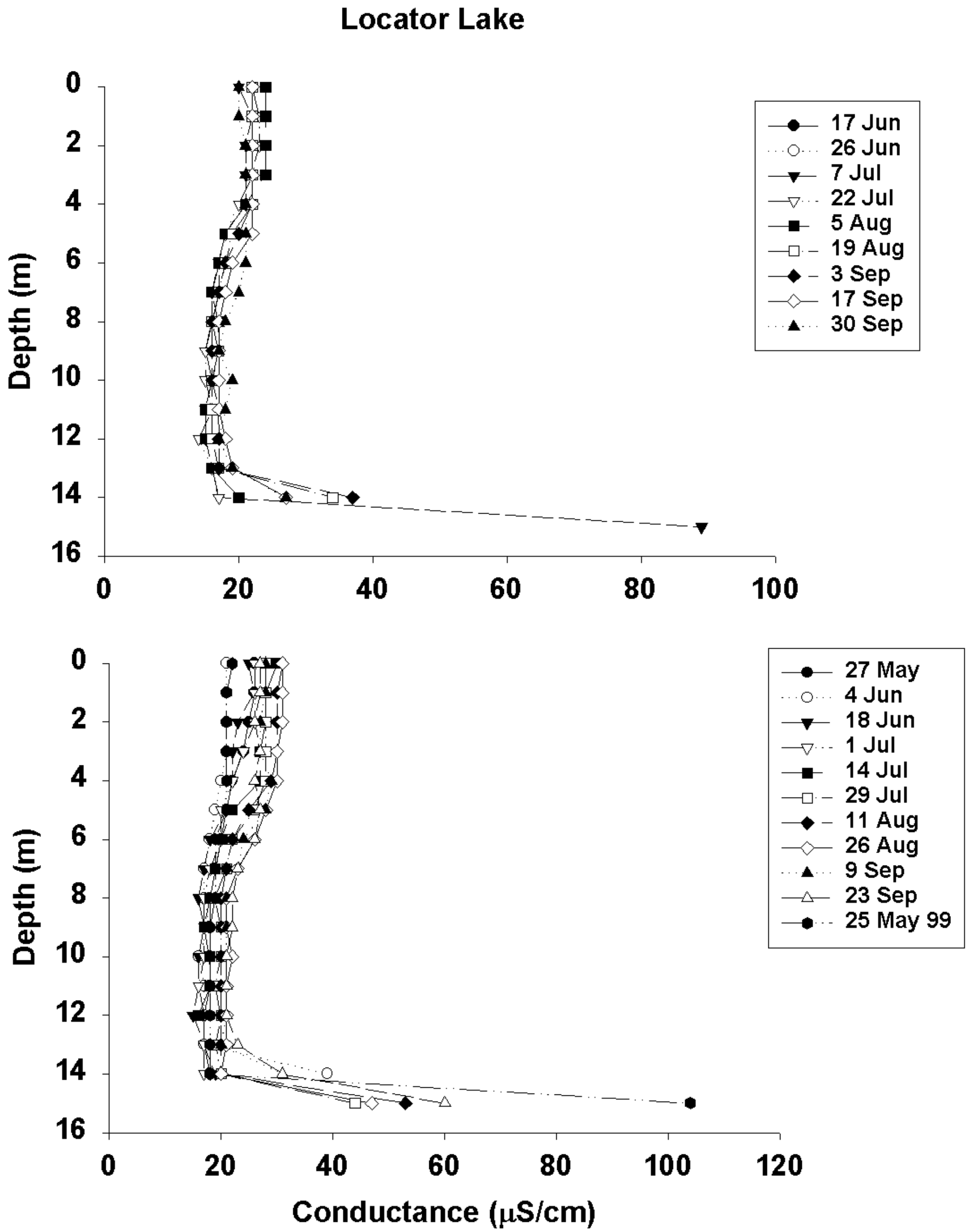


Figure 6-55. Specific conductivity profiles for Locator Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

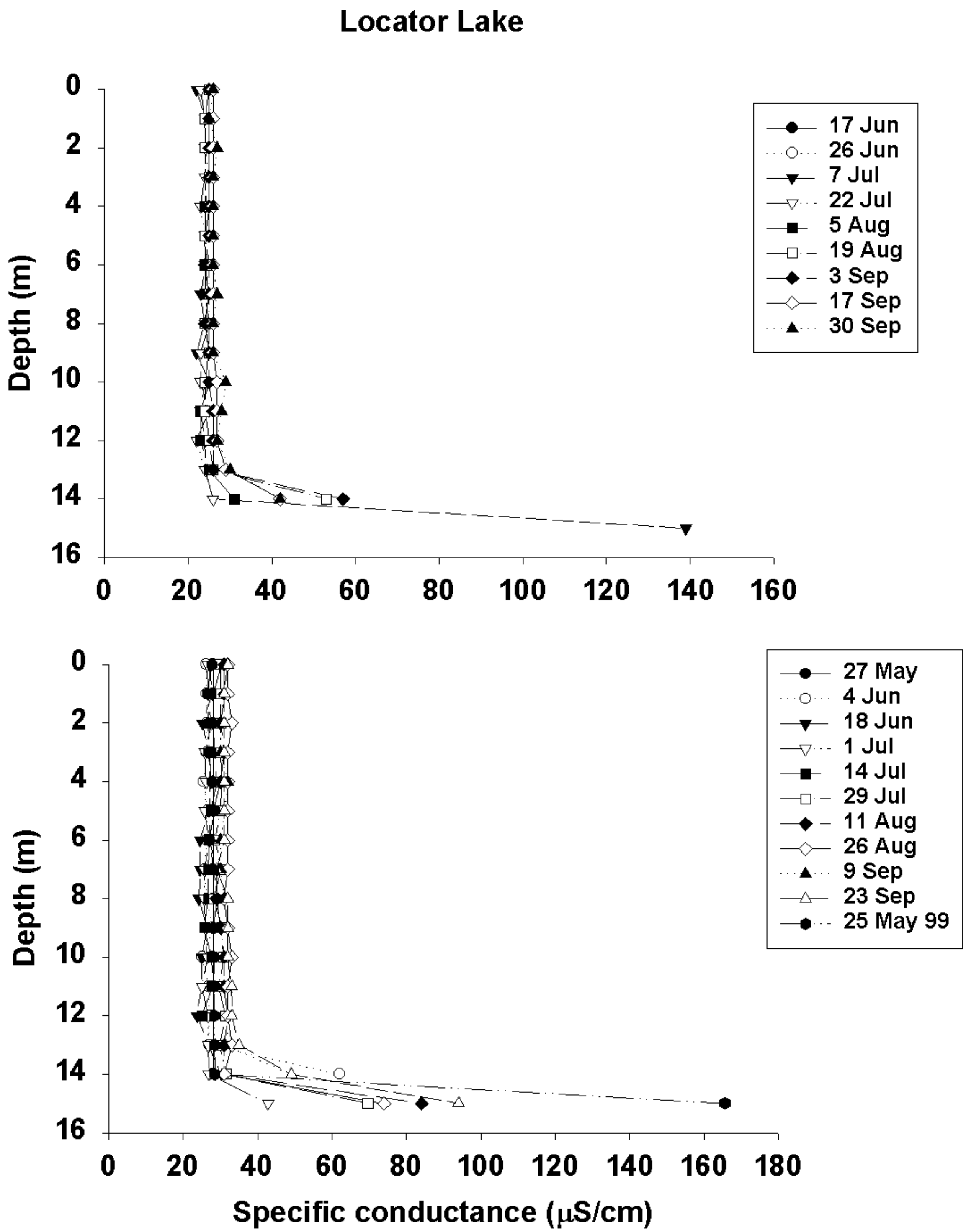


Figure 6-56. Total dissolved solids profiles for Locator Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

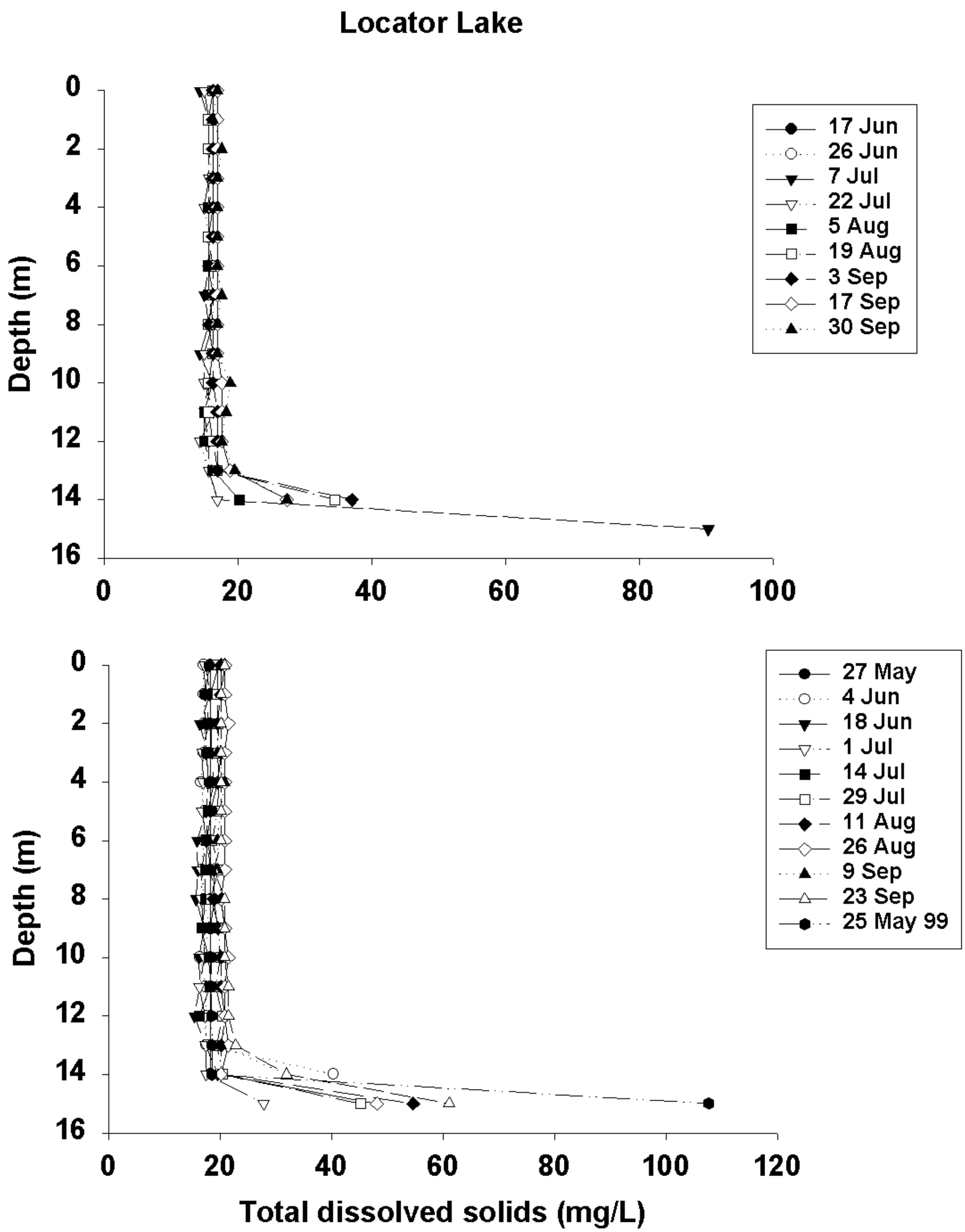


Table 6-15. Means, standard errors (SE), maximums (Max), and minimums (Min) for 1997, 1998, and for 1997-1999 in Locator Lake, Voyageurs National Park, Minnesota. NH4 = ammonium, Ca=calcium, Cl=chloride, Mg=magnesium, N-NO3+NO2=nitrogen as nitrate+nitrite, SiO2= silica, SO_4=sulfate, and P=phosphorus.

Littoral	Alkalinity mg/L	NH_4 mg/L	Ca mg/L	Cl mg/L	Color	Hardness mg/L	Mg mg/L	$N\text{-}NO_3+NO_2$ mg/L	SiO_2 mg/L	SO_4 mg/L	Total P mg/L	Turbidity NTU
97Mean	6.98	0.05	3.30	0.57	25.00	10.80	0.90	0.03	1.80	3.08	0.03	1.31
97SE	0.50	0.00	0.61	0.04	8.22	0.43	0.14	0.01	0.32	0.31	0.00	0.53
97Max	8.50	0.05	4.89	0.67	40.00	12.00	1.10	0.05	2.30	4.30	0.05	2.70
97Min	5.60	0.04	1.90	0.42	5.00	10.00	0.63	0.01	1.20	2.70	0.03	0.44
98Mean	8.93	0.05	2.53	1.03	32.50	10.01	1.06	0.04	1.54	3.18	0.03	0.36
98SE	0.46	0.00	0.03	0.07	1.44	0.68	0.04	0.01	0.29	0.09	0.00	0.03
98Max	10.00	0.05	2.60	1.10	35.00	11.02	1.10	0.05	2.30	3.40	0.03	0.45
98Min	8.10	0.05	2.50	0.83	30.00	7.34	0.95	0.02	0.95	3.00	0.02	0.32
Mean	8.00	0.05	2.95	0.84	29.50	10.67	1.06	0.04	1.79	3.15	0.03	0.85
SE	0.45	0.00	0.31	0.10	4.25	0.49	0.09	0.01	0.23	0.15	0.00	0.29
Max	10.00	0.05	4.89	1.40	40.00	13.42	1.50	0.05	2.80	4.30	0.05	2.70
Min	5.60	0.04	1.90	0.42	5.00	7.34	0.63	0.01	0.95	2.70	0.02	0.32

Limnetic Epilimnion	Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3+NO_2$	SiO_2	SO_4	Total P	Turbidity
97Mean	6.90	0.05	3.54	0.62	36.00	12.56	0.98	0.08	2.73	3.80	0.04	1.08
97SE	0.41	0.00	0.71	0.04	15.68	2.88	0.07	0.03	0.98	0.84	0.01	0.27
97Max	8.00	0.05	5.62	0.70	90.00	21.00	1.10	0.14	4.60	7.00	0.05	1.80
97Min	5.60	0.05	2.10	0.46	5.00	8.00	0.87	0.01	1.30	2.50	0.02	0.44
98Mean	9.68	0.05	2.53	1.04	31.25	10.30	1.06	0.04	1.47	3.18	0.02	0.37
98SE	0.64	0.00	0.05	0.06	1.25	0.43	0.04	0.01	0.27	0.09	0.00	0.02
98Max	11.30	0.05	2.60	1.10	35.00	11.02	1.10	0.05	2.20	3.40	0.04	0.41
98Min	8.20	0.05	2.40	0.87	30.00	8.83	0.94	0.02	0.89	3.00	0.02	0.33
Mean	8.28	0.05	3.04	0.87	34.50	11.28	1.04	0.06	2.10	3.51	0.03	0.74
SE	0.55	0.00	0.37	0.09	7.47	1.13	0.03	0.01	0.42	0.41	0.00	0.17
Max	11.30	0.05	5.62	1.40	90.00	21.00	1.10	0.14	4.60	7.00	0.05	1.80
Min	5.60	0.05	2.10	0.46	5.00	8.00	0.87	0.01	0.89	2.50	0.02	0.33

Limnetic Hypolimnion	Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3+NO_2$	SiO_2	SO_4	Total P	Turbidity
97Mean	6.46	0.05	3.22	0.58	40.00	21.18	1.00	0.11	3.40	3.79	0.03	1.97
97SE	0.36	0.00	0.55	0.05	16.36	6.71	0.05	0.04	0.76	0.97	0.00	0.71
97Max	7.50	0.05	4.62	0.67	90.00	38.00	1.10	0.23	4.40	7.60	0.04	4.00
97Min	5.30	0.04	2.10	0.38	5.00	9.95	0.93	0.01	1.90	2.40	0.02	0.42
98Mean	9.23	0.05	2.50	1.00	68.75	10.43	1.10	0.05	3.93	2.98	0.03	2.52
98SE	0.65	0.00	0.04	0.07	8.26	0.35	0.00	0.00	0.19	0.07	0.00	0.71
98Max	10.90	0.05	2.60	1.10	80.00	11.02	1.10	0.05	4.30	3.10	0.03	3.60
98Min	7.90	0.05	2.40	0.81	45.00	9.07	1.10	0.04	3.40	2.80	0.03	0.46
Mean	7.83	0.05	2.88	0.83	51.50	14.86	1.07	0.08	3.65	3.43	0.03	2.05
SE	0.54	0.00	0.28	0.10	9.52	3.00	0.03	0.02	0.29	0.48	0.00	0.46
Max	10.90	0.05	4.62	1.40	90.00	38.00	1.20	0.23	4.40	7.60	0.04	4.00
Min	5.30	0.04	2.10	0.38	5.00	9.07	0.93	0.01	1.90	2.40	0.02	0.42

Table 6-16. Values from the littoral zone profiles Locator Lake. Means (SE) are given. Most profiles had 1-4 readings. ND=no data available, Sp. Cond.=specific conductance and Cond.=conductance.

Date	6/17/97	6/26/97	7/7/97	7/22/97	8/5/97	8/19/97	9/3/97
Temperature(C)	20.10 (0)	21.00 (0)	20.50 (0.2)	22.44 (0.11)	24.19 (0.12)	19.41 (0.18)	19.50 (0.09)
DO(mg/L)	8.40 (0)	0.20 (0)	6.54 (0.20)	8.39 (0.02)	8.41 (0.06)	8.83 (0.13)	8.95 (0.08)
DO(%saturation)	92.6 (0)	2.2 (0)	72.7 (2.5)	96.8 (0.5)	100.3 (1.0)	96.0 (1.8)	97.4 (1.0)
pH	6.60 (0)	6.65 (0)	7.11 (0.06)	7.28 (0.03)	7.18 (0.03)	7.28 (0.06)	7.28 (0.01)
TDS (mg/L)	17 (0)	14 (0)	15 (0)	15 (0)	16 (0)	16 (0)	16 (0)
Sp. Cond. (μS/cm)	27 (0)	21 (0)	24 (0)	24 (0)	25 (0)	24 (0)	24 (0)
Cond. (μS/cm)	25 (0)	19 (0)	22 (0)	23 (0)	24 (0)	22 (0)	22 (0)

Date	9/17/97	9/30/97	5/27/98	6/4/98	6/18/98	7/1/98	7/14/98
Temperature(C)	17.53 (0.01)	14.67 (0.00)	22.52 (0.12)	14.98 (0.19)	17.19 (0.62)	24.60 (0.47)	ND
DO(mg/L)	9.22 (0.20)	10.65 (0.11)	9.51 (0.03)	10.34 (0.03)	10.26 (0.18)	6.64 (0.20)	ND
DO(%saturation)	96.4 (2.2)	104.9 (1.1)	109.9 (0.6)	102.4 (0.3)	106.5 (0.5)	79.8 (1.8)	ND
pH	7.12 (0.09)	7.00 (0.03)	7.13 (0.05)	7.83 (0.06)	8.37 (0.21)	7.44 (0.03)	ND
TDS (mg/L)	17 (0)	16 (0)	18 (0)	16 (0)	17 (0)	16 (0)	ND
Sp. Cond. (μS/cm)	26 (0)	26 (1)	28 (0)	25 (0)	26 (0)	25 (1)	ND
Cond. (μS/cm)	22 (0)	20 (1)	27 (0)	21 (0)	22 (0)	25 (1)	ND

Date	7/29/98	8/11/98	8/26/98	9/9/98	9/23/98	5/25/99
Temperature(C)	22.02 (0.01)	24.48 (0.45)	22.85 (0.09)	19.87 (0.07)	17.23 (0.01)	14.51 (0.34)
DO(mg/L)	9.41 (1.43)	8.98 (0.18)	8.58 (0.07)	9.53 (0.03)	5.27 (0.01)	8.93 (0.06)
DO(%saturation)	107.7 (16.3)	107.7 (2.9)	99.8 (0.7)	104.6 (0.4)	54.8 (0.1)	87.6 (1.1)
pH	7.66 (0.03)	7.89 (0.13)	7.74 (0.01)	7.82 (0.04)	7.86 (0.02)	6.92 (0.01)
TDS (mg/L)	19 (0)	26 (2)	20 (0)	20 (1)	20 (0)	17 (0)
Sp. Cond. (μS/cm)	30 (1)	40 (3)	31 (0)	32 (1)	31 (0)	26 (0)
Cond. (μS/cm)	28 (1)	39.75 (2)	30 (0)	29 (1)	27 (0)	21 (0)

Mukooda Lake

Mukooda Lake was stratified throughout the sampling in both 1997 and 1998. Temperature profiles exhibited typical clinograde curves (Figure 6-57). Dissolved oxygen profiles were positive orthograde curves (Figure 6-58), which suggests that algae were extremely productive at the epilimnion-metalimnion interface during the day. The oxygen saturation curves also suggest high primary production in this area (Figure 6-59).

The water chemistry is summarized by year in Tables 6-17 and 6-18. Chloride was significantly higher in 1998 than in 1997 ($P = 0.002$). No other parameters were significantly different between years (all t-test P-values > 0.12; P-critical = 0.007 after Bonferonni adjustment), although turbidity was slightly higher in 1997 than in 1998 ($P = 0.047$). In general, within-year variation was minimal as suggested by SEM of <20% of the means for all parameters.

The water chemistry in the hypolimnion showed similar trends as the epilimnion (Table 6-17). Chloride was significantly higher in 1998 than in 1997 ($P = 0.001$), whereas all other parameters were not significantly different (all P-values > 0.13). Nitrogen as nitrate+nitrite was slightly higher in 1997 than in 1998 ($P = 0.039$). Ammonia was below detection limits all years.

Water samples from the littoral zone showed similar trends as was observed in the hypolimnion. Only chloride was significantly different ($P = 0.001$, mean 1998 > mean 1997), whereas all other parameters were not significantly different (all P-values > 0.15). Silica was marginally higher in 1997 than in 1998 ($P = 0.44$). Ammonia was only detected in one sample, but was consistently below detection limit in other samples, which indicates very low concentrations.

When parameters were compared between epilimnion and hypolimnion (paired tests), silica and chloride were significantly higher in the hypolimnion than in the epilimnion ($P < 0.001$ for silica and $P = 0.001$ for chloride). All other parameters showed no significant differences, although turbidity was marginally higher in 1998 than in 1997 ($P = 0.02$). Silica was significantly higher in the hypolimnion compared to the littoral ($P < 0.001$) and turbidity was marginally higher ($P = 0.008$).

Limnology

Mukooda Lake was comparable to Locator Lake in many aspects. Mukooda Lake was low in calcium and magnesium and would be considered a soft water lake. Although alkalinity was higher than Locator Lake, it is still in the low range. Specific conductance values were consistent with alkalinity, hardness, and other cation data. The extremely high values measured at the very bottom of the lake were caused by the probe contacting the bottom sediments and are of no concern.

Ammonia was below detection level throughout the lake, and as before this low level in deeper water during stratification suggests low productivity. Total phosphorus and other nitrogenous compounds also were low, again suggesting low productivity.

Silica was especially low in the littoral zone, which would indicate high diatom production in this zone. The hypolimnion was higher in silica as expected. Sulfate levels were normal for a lake during stratification. Based on the water chemistry data, Mukooda Lake is an oligotrophic to slightly mesotrophic lake. The low nutrients in the water column suggest that the lake has no eutrophication problems at present.

Figure 6-57. Temperature profile for Mukooda Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

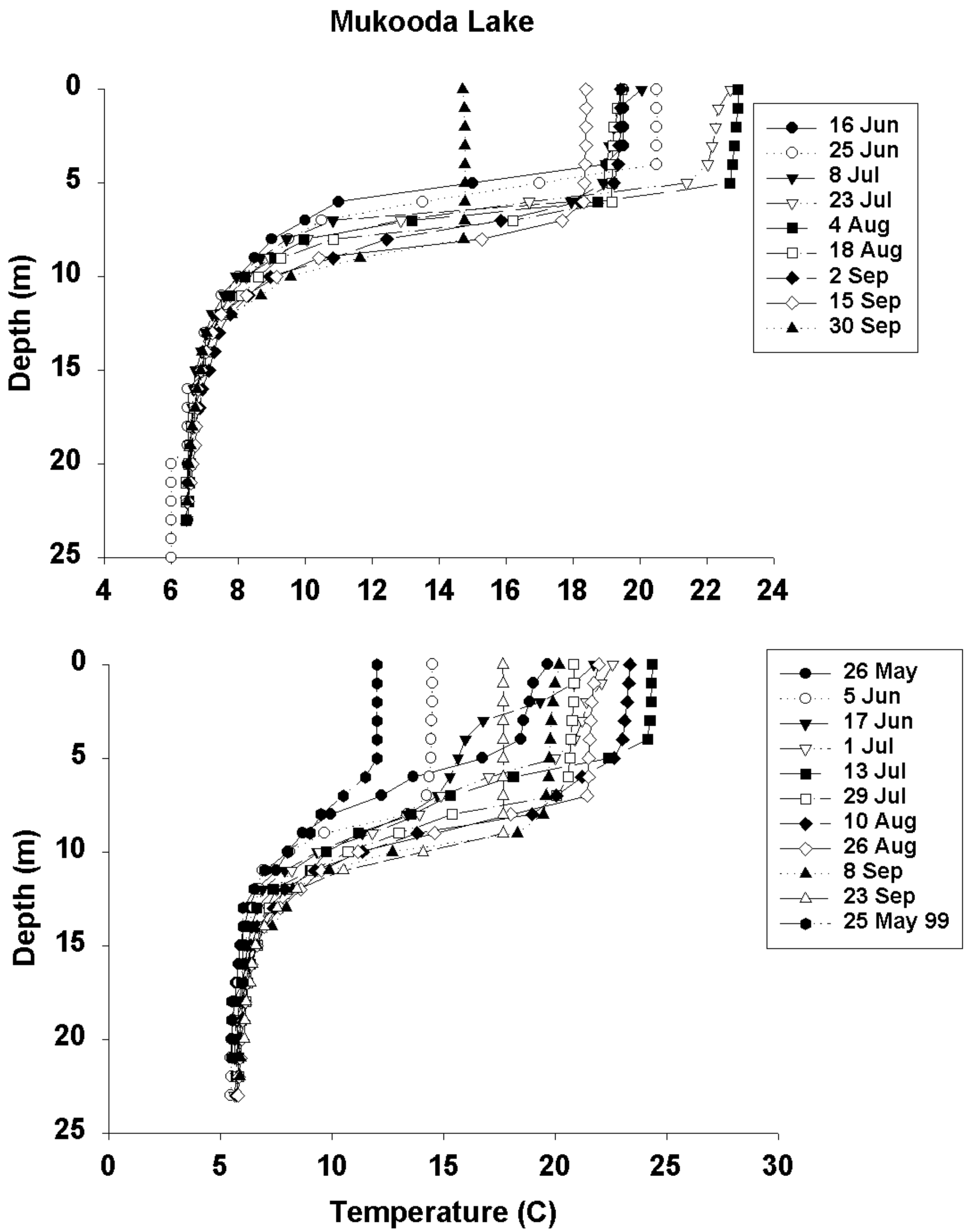


Figure 6-58. Dissolved oxygen profiles for Mukooda Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

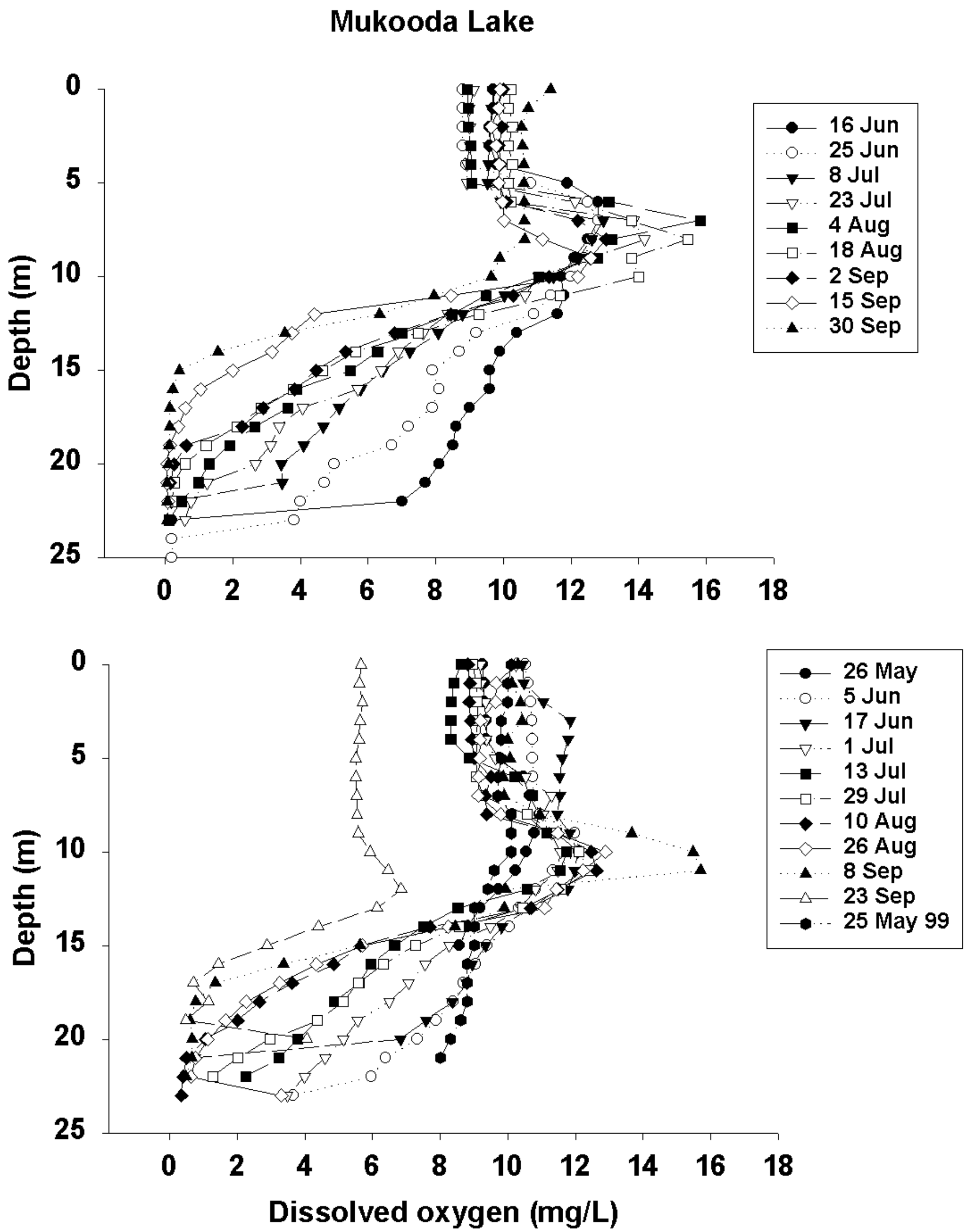


Figure 6-59. Oxygen saturation calculated from measured water temperature and dissolved oxygen in Mukooda Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

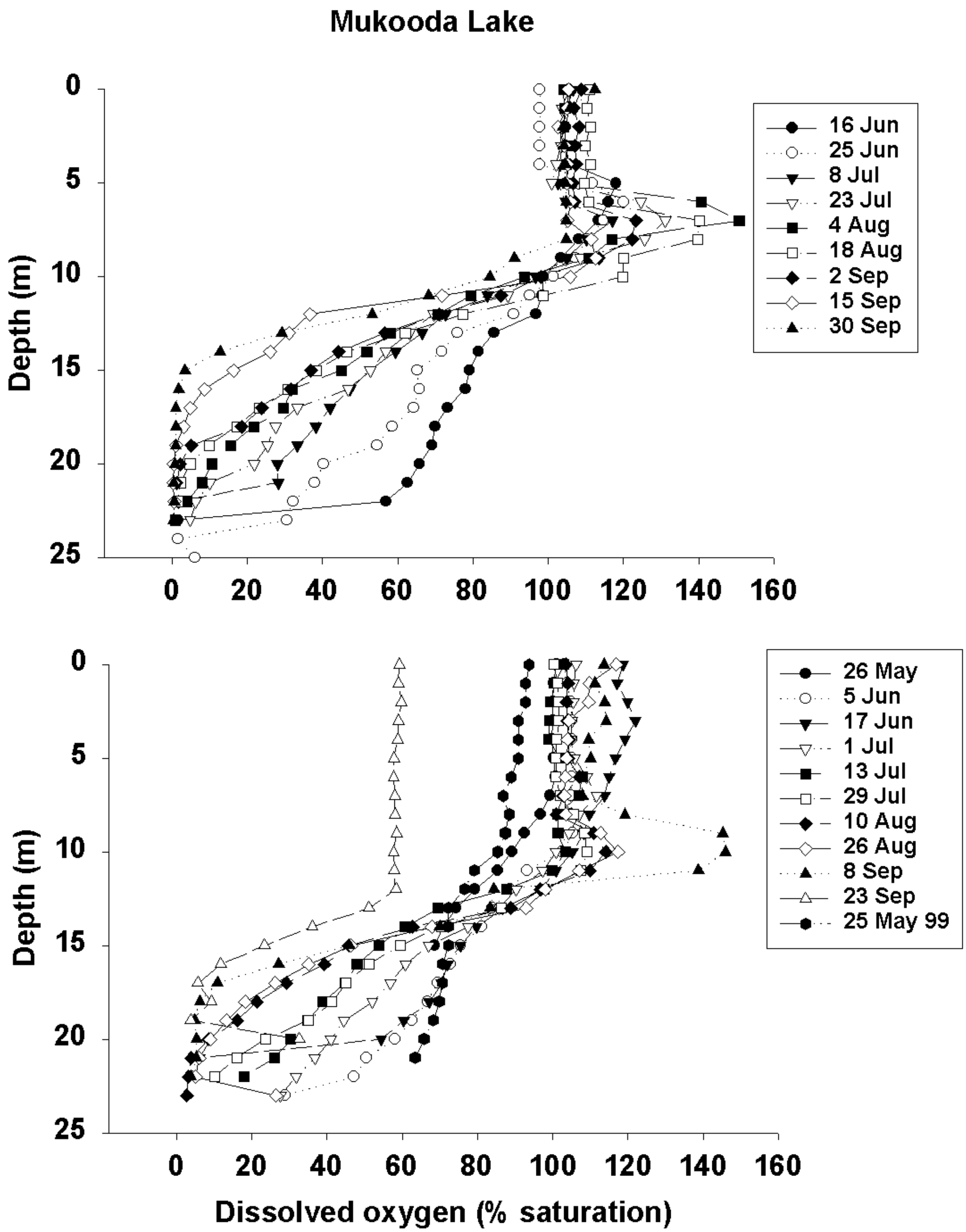


Figure 6-60. pH profiles for Mukooda Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

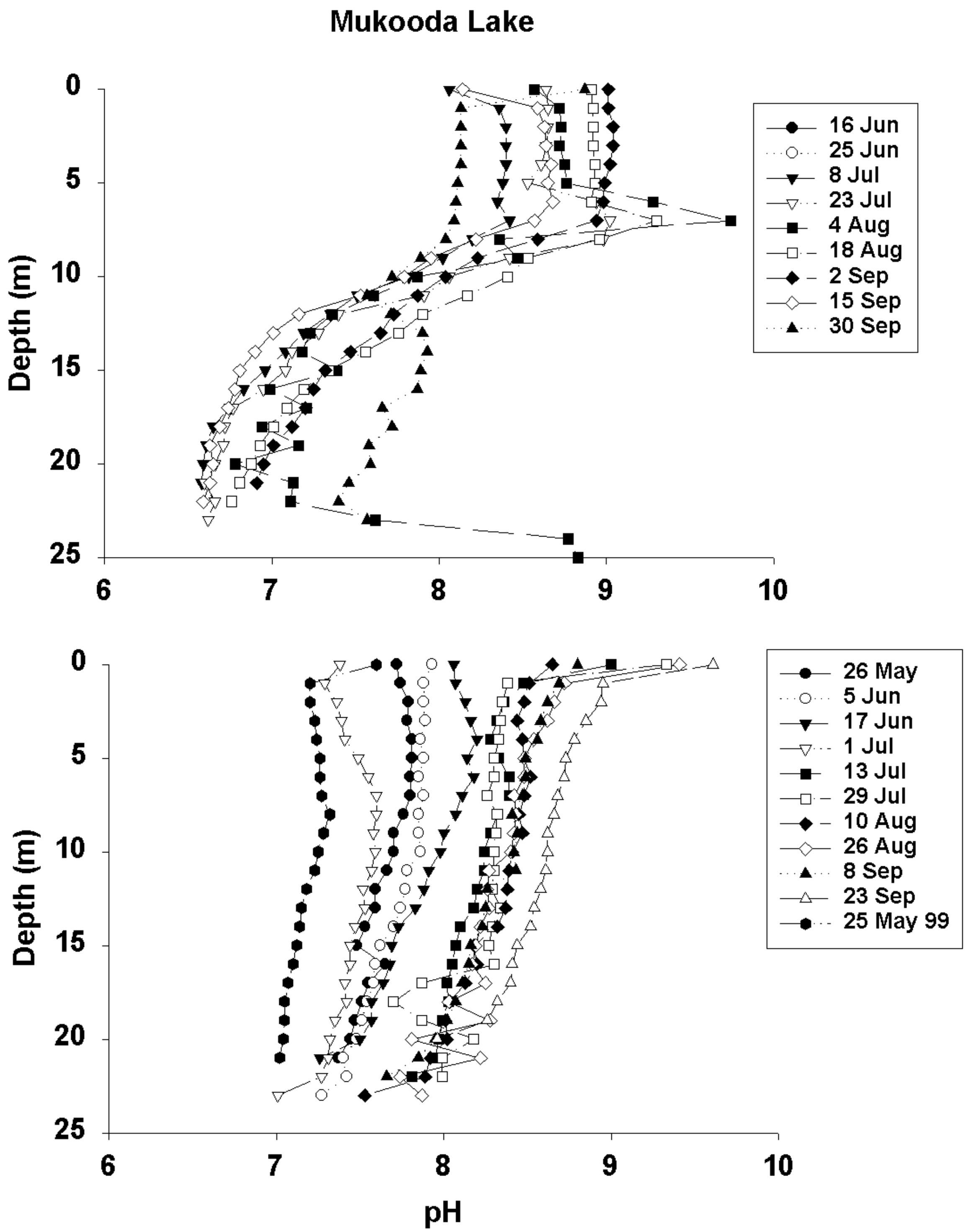


Figure 6-61. Conductivity profiles for Mukooda Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

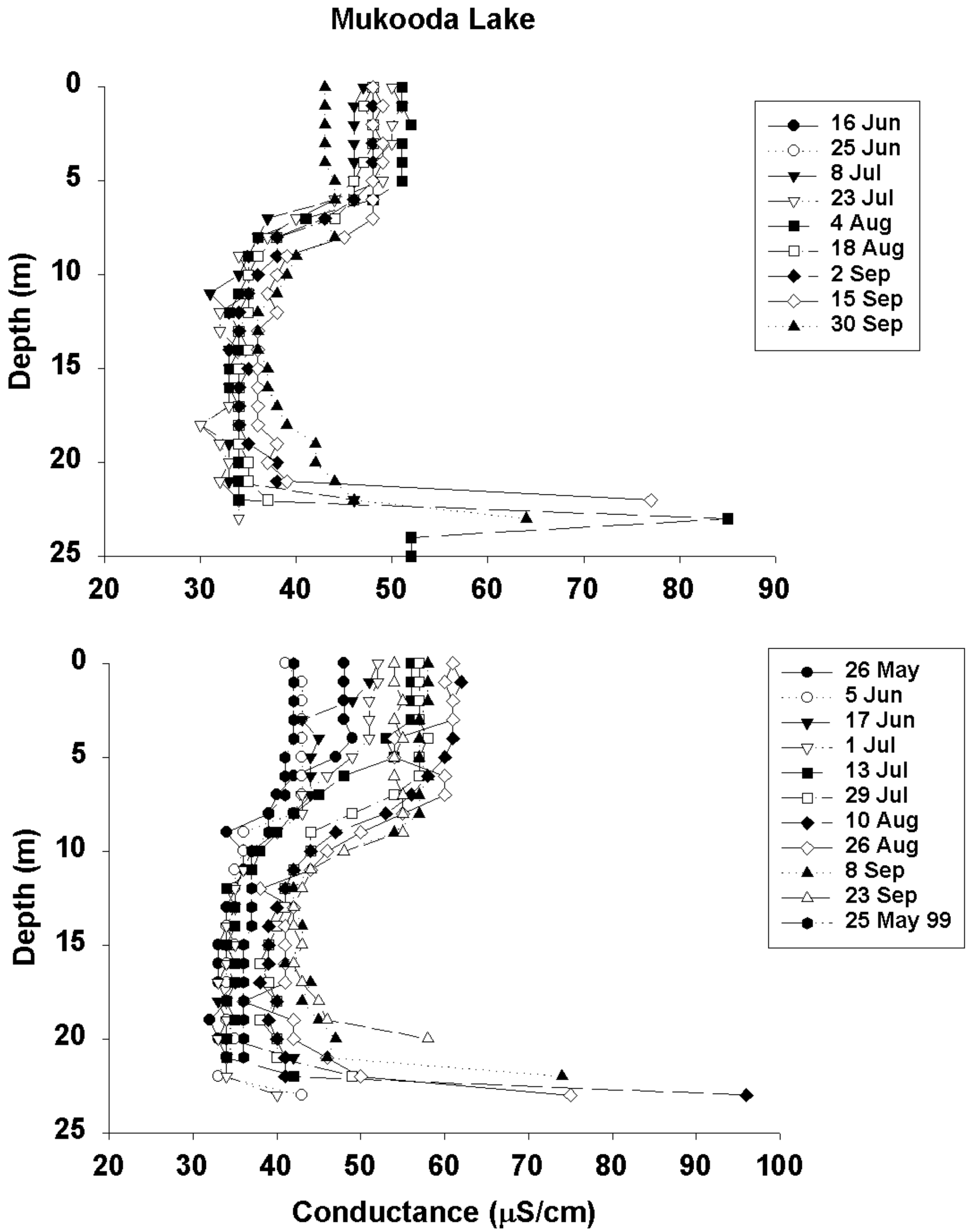


Figure 6-62. Specific conductivity profiles for Mukooda Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

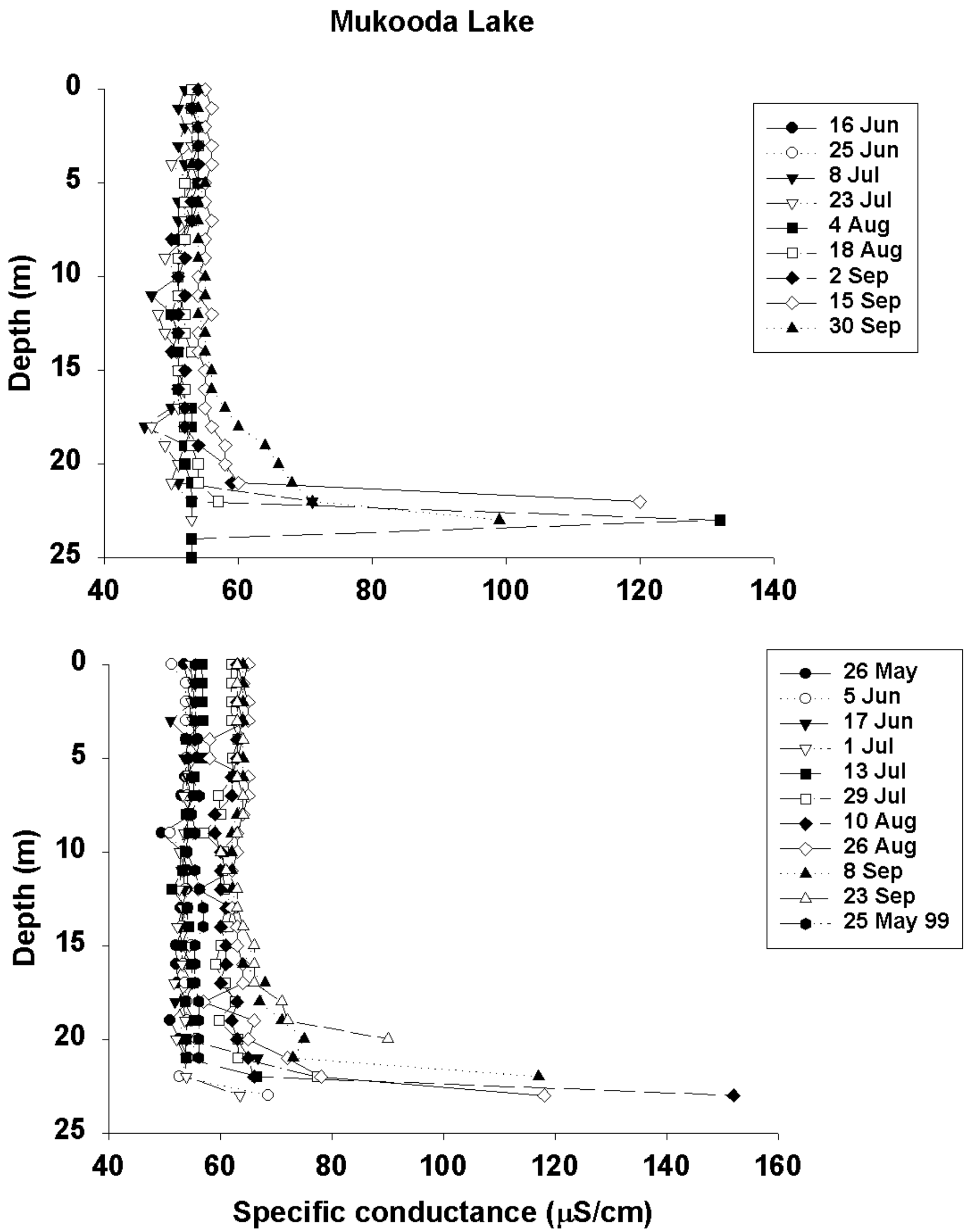


Figure 6-63. Total dissolved solids profiles for Mukooda Lake, Voyageurs National Park, Minnesota. Top panel is 1997 and bottom panel is 1998/1999.

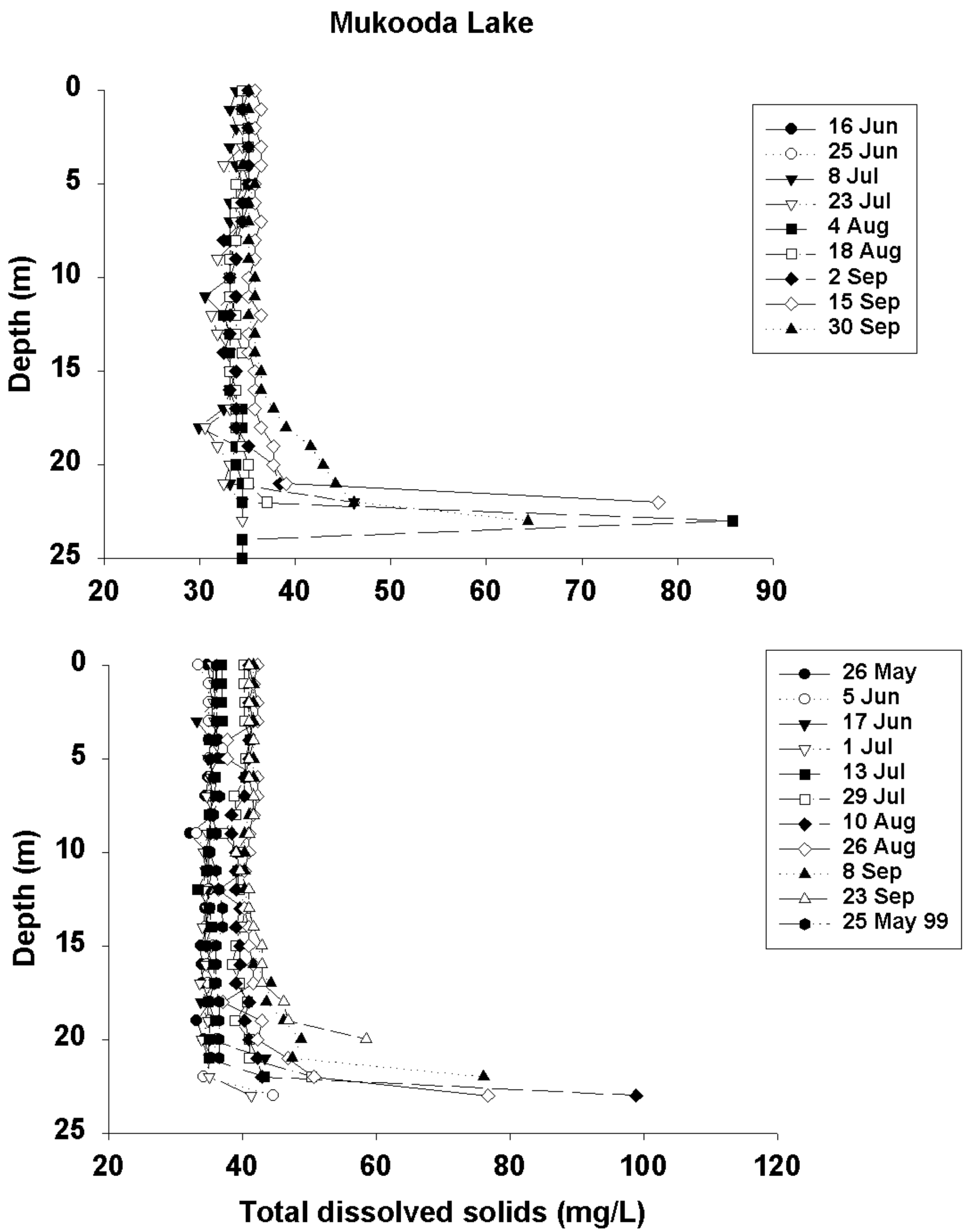


Table 6-17. Means, standard errors (SE), maximums (Max), and minimums (Min) for 1997, 1998, and for 1997-1999 in Mukooda Lake, Voyageurs National Park, Minnesota. NH4 = ammonium, Ca=calcium, Cl=chloride, Mg=magnesium, N-NO3+NO2=nitrogen as nitrate+nitrite, SiO2= silica, SO_4=sulfate, and P=phosphorus.

	Littoral Alkalinity mg/L	NH_4 mg/L	Ca mg/L	Cl mg/L	Color	Hardness mg/L	Mg mg/L	$N\text{-}NO_3+NO_2$ mg/L	SiO_2 mg/L	SO_4 mg/L	Total P mg/L	Turbidity NTU
97Mean	26.12	0.07	7.43	0.49	8.00	30.98	2.75	0.06	0.93	2.11	0.03	0.81
97SE	0.45	0.02	1.20	0.03	1.22	2.86	0.18	0.02	0.11	0.04	0.00	0.23
97Max	27.60	0.15	10.70	0.60	10.00	38.66	3.00	0.14	1.10	2.27	0.03	1.70
97Min	25.00	0.04	4.90	0.42	5.00	26.59	2.20	0.01	0.65	2.00	0.02	0.51
98Mean	26.78	0.04	5.85	0.91	6.25	25.50	2.90	0.09	0.65	2.20	0.03	0.40
98SE	0.37	0.01	0.17	0.07	1.25	1.20	0.08	0.05	0.03	0.04	0.00	0.02
98Max	27.60	0.05	6.30	0.99	10.00	28.50	3.10	0.24	0.72	2.30	0.03	0.43
98Min	25.80	0.02	5.50	0.69	5.00	21.29	2.70	0.02	0.57	2.10	0.03	0.34
Mean	26.46	0.06	6.66	0.74	7.50	27.90	2.84	0.07	0.86	2.19	0.03	0.63
SE	0.27	0.01	0.62	0.09	0.83	1.47	0.09	0.02	0.09	0.05	0.00	0.13
Max	27.60	0.15	10.70	1.30	10.00	38.66	3.10	0.24	1.40	2.50	0.03	1.70
Min	25.00	0.02	4.90	0.42	5.00	21.29	2.20	0.01	0.57	2.00	0.02	0.34

	Limnetic Epilimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3+NO_2$	SiO_2	SO_4	Total P	Turbidity
97Mean	25.76	0.04	7.59	0.49	8.00	27.19	2.87	0.11	0.83	2.29	0.03	1.09
97SE	0.54	0.01	1.26	0.05	1.22	1.44	0.09	0.05	0.07	0.14	0.00	0.25
97Max	27.00	0.05	11.00	0.59	10.00	31.00	3.00	0.26	0.93	2.71	0.03	1.80
97Min	24.00	0.02	4.90	0.32	5.00	24.00	2.70	0.01	0.69	2.00	0.01	0.61
98Mean	26.85	0.05	5.78	0.89	5.63	25.68	2.88	0.04	0.63	2.20	0.03	0.40
98SE	0.16	0.00	0.10	0.07	1.57	0.71	0.06	0.01	0.04	0.04	0.00	0.04
98Max	27.30	0.05	6.00	0.99	10.00	27.34	3.00	0.05	0.71	2.30	0.04	0.50
98Min	26.60	0.04	5.50	0.67	2.50	23.35	2.70	0.02	0.53	2.10	0.02	0.34
Mean	26.29	0.05	6.80	0.73	7.25	27.17	3.04	0.07	0.81	2.28	0.03	0.76
SE	0.31	0.00	0.66	0.10	0.95	1.05	0.17	0.03	0.11	0.08	0.00	0.16
Max	27.30	0.05	11.00	1.30	10.00	34.52	4.20	0.26	1.50	2.71	0.04	1.80
Min	24.00	0.02	4.90	0.32	2.50	23.35	2.70	0.01	0.53	2.00	0.01	0.34

	Limnetic Hypolimnion Alkalinity	NH_4	Ca	Cl	Color	Hardness	Mg	$N\text{-}NO_3+NO_2$	SiO_2	SO_4	Total P	Turbidity
97Mean	25.84	0.05	7.55	0.51	15.00	26.98	2.73	0.15	2.47	2.30	0.04	2.31
97SE	0.40	0.00	1.26	0.04	4.74	1.23	0.27	0.04	0.20	0.20	0.01	0.89
97Max	26.80	0.05	11.10	0.61	30.00	30.00	3.00	0.22	2.80	3.01	0.06	4.90
97Min	24.80	0.03	4.90	0.39	5.00	24.00	2.20	0.01	2.10	1.90	0.03	0.70
98Mean	27.03	0.05	5.85	0.91	17.50	25.58	2.93	0.04	2.10	2.23	0.03	0.76
98SE	0.61	0.00	0.18	0.07	3.23	1.26	0.10	0.01	0.22	0.02	0.01	0.12
98Max	28.00	0.06	6.40	1.00	25.00	29.16	3.20	0.05	2.60	2.30	0.05	1.10
98Min	25.50	0.05	5.60	0.69	10.00	21.29	2.70	0.02	1.60	2.20	0.01	0.56
Mean	26.51	0.05	6.72	0.75	16.00	26.32	2.86	0.09	2.23	2.30	0.04	1.66
SE	0.37	0.00	0.66	0.09	2.56	0.78	0.11	0.03	0.14	0.10	0.00	0.49
Max	28.00	0.06	11.10	1.30	30.00	30.00	3.20	0.22	2.80	3.01	0.06	4.90
Min	24.80	0.03	4.90	0.39	5.00	21.29	2.20	0.01	1.60	1.90	0.01	0.56

Table 6-18. Values from the littoral zone profiles Mukooda Lake. Means (SE) are given. Most profiles had 1-4 readings. ND=no data available, Sp. Cond.=specific conductance and Cond.=conductance.

Date	6/16/97	6/25/97	7/8/97	7/23/97	8/4/97	8/18/97	9/2/97
Temperature(C)	20.00 (0)	20.50 (0)	19.73 (0.32)	22.86 (0.33)	ND	19.80 (0.08)	19.38 (0.03)
DO(mg/L)	8.40 (0)	8.80 (0)	9.56 (0.08)	9.12 (0.16)	ND	10.32 (0.09)	9.99 (0.02)
DO(%saturation)	92.4 (0)	97.8 (0)	104.6 (0.9)	106.0 (1.2)	ND	113.0 (0.9)	108.5 (0.3)
pH	7.3 (0)	7.55 (0)	8.26 (0.15)	8.72 (0.06)	ND	8.93 (0.05)	9.03 (0.01)
TDS (mg/L)	35 (0)	35 (0)	32 (1)	33 (0)	ND	34 (0)	34 (0)
Sp. Cond. (μS/cm)	55 (0)	55 (0)	49 (1)	51 (0)	ND	53 (0)	53 (0)
Cond. (μS/cm)	49 (0)	50 (0)	44 (1)	49 (1)	ND	48 (0)	48 (0)

Date	9/15/97	9/30/97	5/26/98	6/5/98	6/17/98	7/1/98	7/13/98
Temperature(C)	18.57 (0.03)	14.80 (0.02)	20.35 (0.17)	14.51 (0.00)	23.03 (0.40)	21.01 (0.45)	24.86 (0.03)
DO(mg/L)	9.96 (0.07)	11.07 (0.16)	5.80 (0.11)	10.84 (0.06)	10.72 (0.06)	3.84 (0.16)	7.91 (0.12)
DO(%saturation)	106.4 (0.6)	109.3 (1.5)	64.3 (1.0)	106.4 (0.6)	125.1 (1.6)	43.0 (1.4)	95.6 (1.5)
pH	8.54 (0.12)	7.96 (0.04)	7.75 (0.01)	7.96 (0.03)	7.98 (0.04)	7.64 (0.03)	8.24 (0.01)
TDS (mg/L)	36.4 (0)	35 (0)	35 (0)	34 (0)	42 (0)	35 (0)	36 (0)
Sp. Cond. (μS/cm)	56 (0)	54 (0)	54 (0)	53 (0)	65 (0)	54 (0)	55 (0)
Cond. (μS/cm)	49 (0)	44 (0)	50 (0)	43 (0)	63 (0)	50 (0)	55 (0)

Date	7/29/98	8/10/98	8/26/98	9/8/98	9/23/98	5/25/99
Temperature(C)	20.89 (0.01)	23.09 (0.08)	22.00 (0.10)	19.97 (0.09)	17.47 (0.12)	12.16 (0.16)
DO(mg/L)	8.15 (0.06)	7.74 (0.14)	9.32 (0.04)	10.10 (0.25)	5.86 (0.12)	10.26 (0.03)
DO(%saturation)	91.2 (0.74)	90.4 (1.5)	106.6 (0.3)	111.0 (2.9)	61.2 (1.3)	95.5 (0.0)
pH	8.11 (0.02)	8.27 (0.02)	8.16 (0.01)	8.25 (0.01)	8.34 (0.01)	7.55 (0)
TDS (mg/L)	40 (0)	41 (0)	41 (0)	41 (0)	40 (0)	35 (0)
Sp. Cond.(μS/cm)	62 (0)	63 (0)	64 (0)	63 (0)	63 (0)	54 (0)
Cond. (μS/cm)	57 (0)	61 (0)	60 (0)	57 (0)	53 (0)	41 (0)

Comparisons among lakes and parks

Discriminant analyses showed that lakes could be separated based on a few water chemistry parameters. Alkalinity and either chloride or silica were the best parameters for separating lakes. In the epilimnion, alkalinity and chloride were different enough among lakes to provide good separation, with the exception of some overlap between North Bar and Loon lakes, and almost complete overlap between Siskiwit and Mukooda lakes (Figure 6-64). Round Lake had such high alkalinity that it was far removed from the other lakes. When lakes were combined into parks, SLBE showed clear separation from the other parks, which in turn separated well among each other based on alkalinity combined with silica (Figure 6-65). Inter-park variation among lakes is observable by the lack of data points fitting inside the 95% confidence ellipses. Similar patterns of separation were observed using the water chemistry data collected from the hypolimnion of the lakes, with the exception that silica fell out of the analysis at the among-park comparisons and was replaced with chloride. In the littoral zones, alkalinity and chloride combined to offer the best separation. Only Mukooda and Siskiwit lakes showed meaningful overlap, whereas Loon and North Bar lakes showed only slight overlap (Figure 6-66). When the data were grouped by park and alkalinity and chloride were considered, slight overlap occurred only between Voyageurs and Isle Royale, whereas the other parks separated well among each other (Figure 6-67). These differences in alkalinity can best be explained by differences in bedrock geology. SLBE has a limestone-shale dominated bedrock, which would tend to increase alkalinity. PIRO is mostly dolomite and sandstone. ISRO is composed of a conglomerate. VOYA is predominantly sedimentary and igneous rock.

Figure 6-64. Canonical scores plot from a discriminant analysis. Parameters were alkalinity and chloride in the epilimnion of lakes from 1997 and 1998. Ellipses represent 95% confidence around the centroid for each lake.

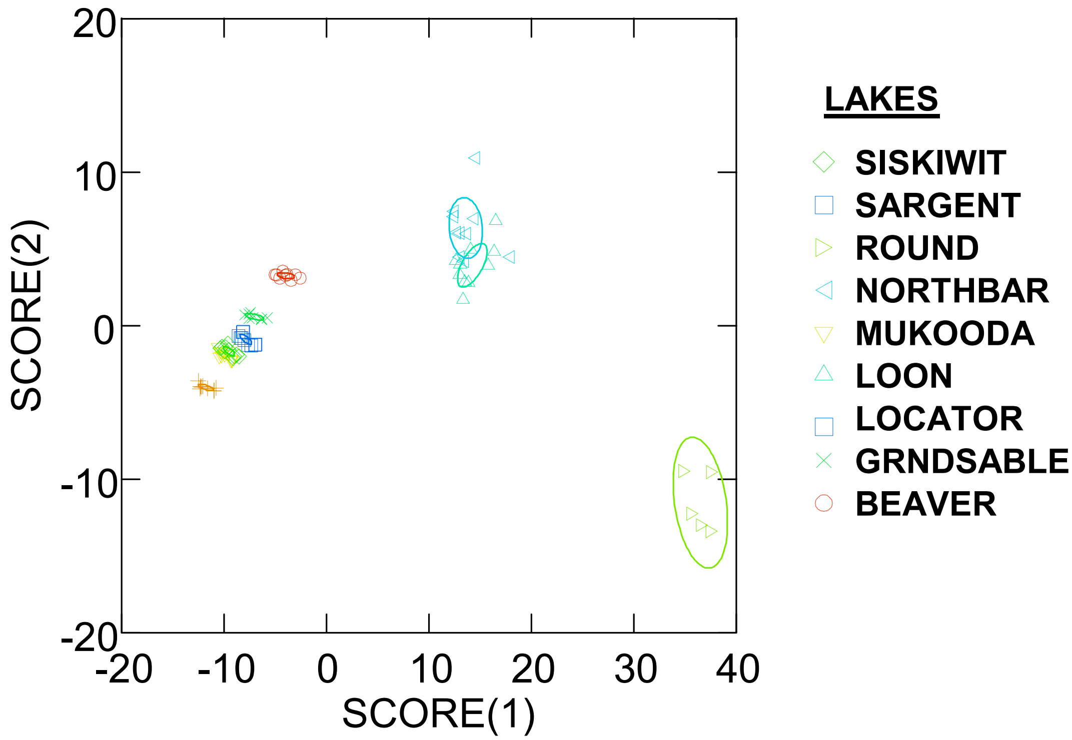


Figure 6-65. Canonical scores plot from a discriminant analysis. Parameters were alkalinity and silica in the epilimnion of lakes from 1997 and 1998. Ellipses represent 95% confidence around the centroid for each park.

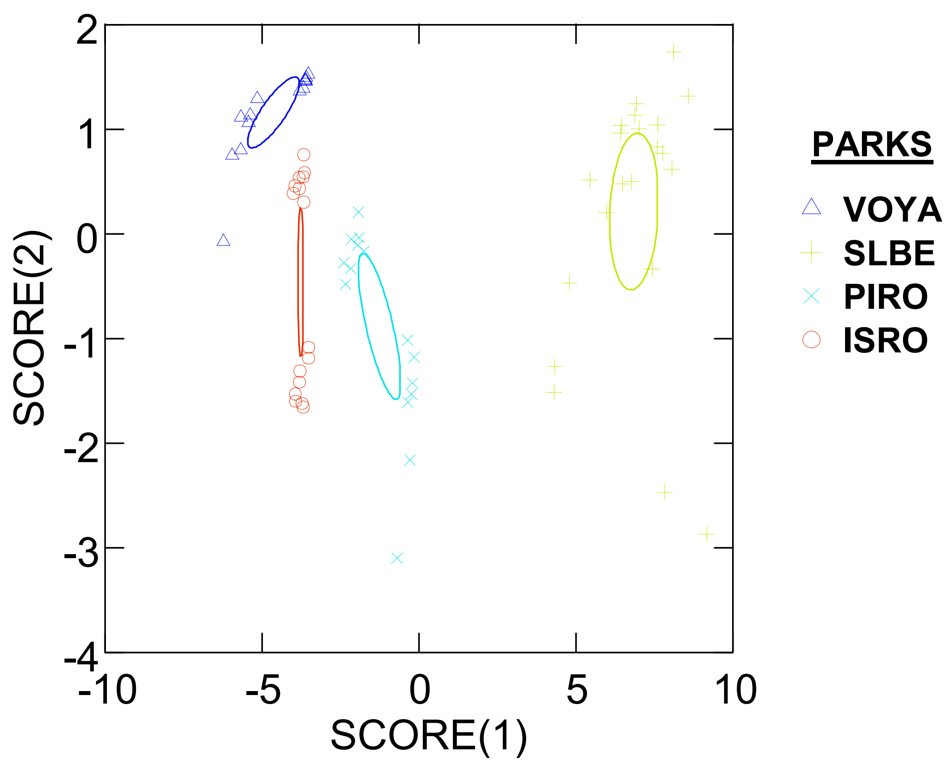


Figure 6-66. Canonical scores plot from a discriminant analysis. Parameters were alkalinity and chloride in the littoral zone of lakes from 1997 and 1998. Ellipses represent 95% confidence around the centroid for each lake.

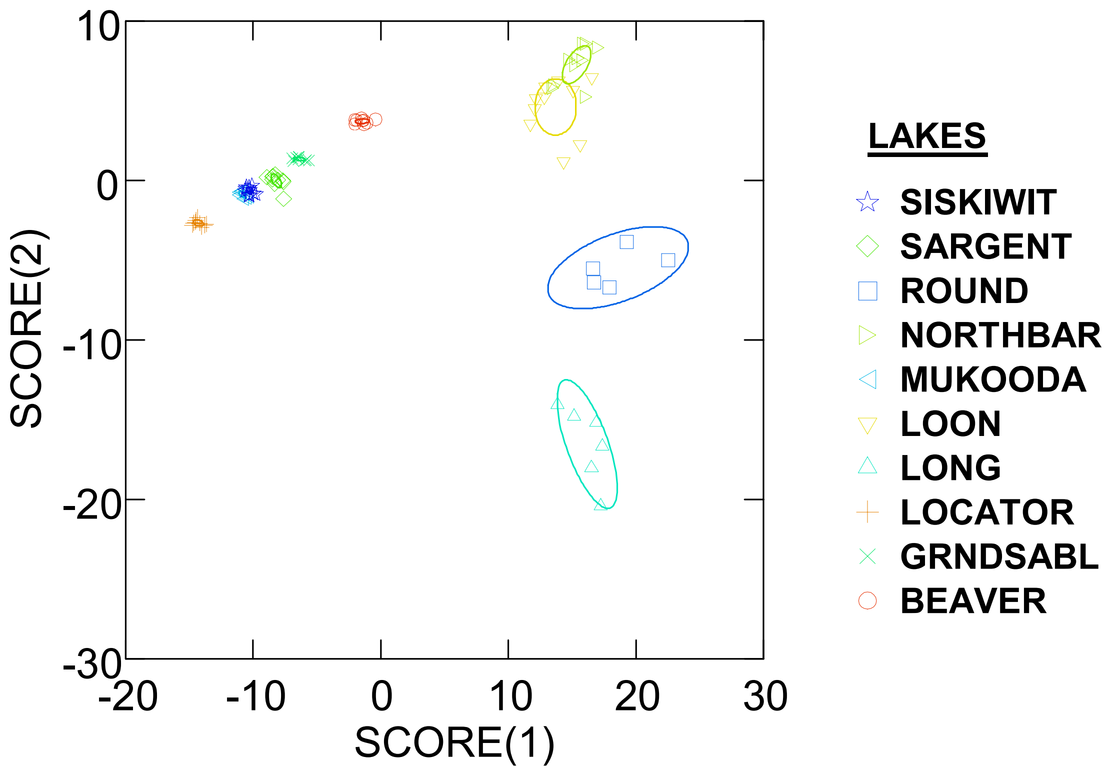
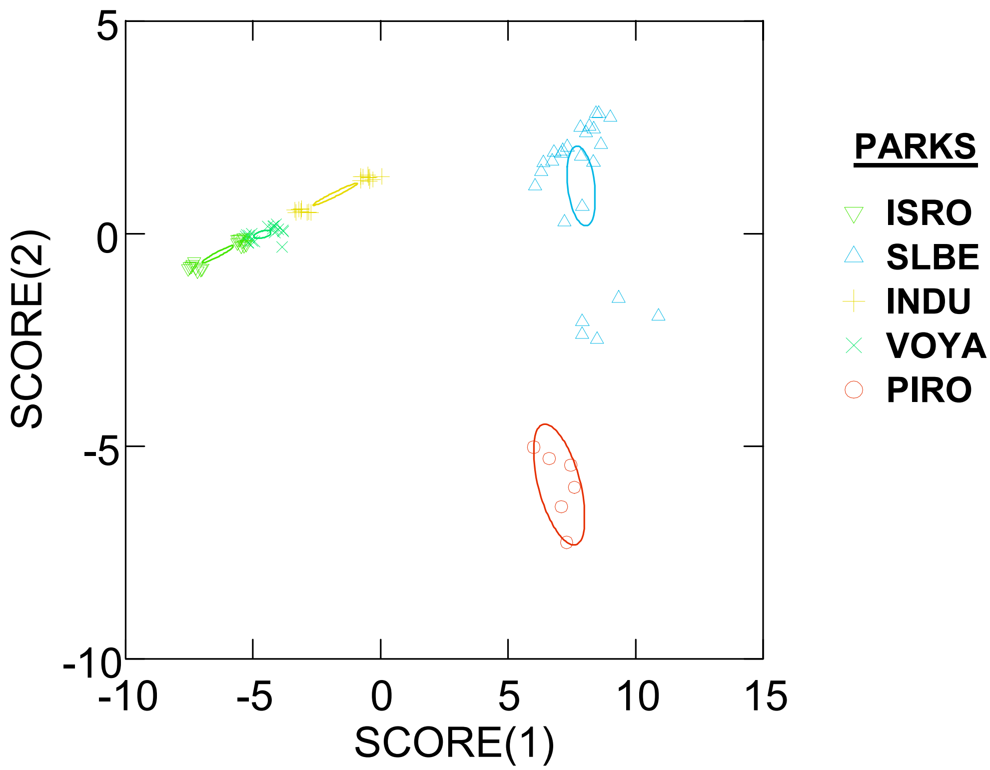


Figure 6-67. Canonical scores plot from a discriminant analysis. Parameters were alkalinity and chloride in the littoral zone of lakes from 1997 and 1998. Ellipses represent 95% confidence around the centroid for each park.



Chapter 7 Sediment Chemistry Results

Nutrients

During the first season of sampling, 1997, sediment samples were collected between June and September at all parks. One park, Pictured Rocks, did not collect sediment samples from Beaver and Grand Sable lakes in June and instead collected early July samples. For clarification and comparison purposes, the early July sample will be referred to as a June sample and the late July sample will be referred to as simply July. Similarly, the sample taken for Siskiwit Lake on July 1 will be referred to as the June sample. These samples were taken within a week of the other parks' June samples and were the first samples of the season.

Total phosphorus, total Kjeldahl nitrogen, and nitrate plus nitrite as N were measured monthly in each lake. Nitrate plus nitrite as N was below detection limits on almost all samples analyzed. After June, total organic carbon was added to the group of nutrients analyzed. Because of the lack of change over the course of the season, results were averaged, and means were recorded for the limnetic zone (Table 7-1) and the littoral zone (Table 7-2).

Loon Lake Total organic carbon, total Kjeldahl nitrogen (TKN), and total phosphorus were all constant through the season in both the littoral and limnetic zones, but concentrations in the limnetic zone were generally higher. Total organic carbon in the littoral zone was around 1%, and in the limnetic zone it was consistently around 5%. TKN in the limnetic zone was more variable, with a concentration of 1062 mg/kg in May increasing to a high of 6390 mg/kg in June. Littoral concentrations ranged from 600-1700 mg/kg. Total phosphorus in the limnetic zone was low in May (80 mg/kg) but increased to a high of 478 mg/kg in June. Total phosphorus was also highest in the littoral zone in June, although the concentration was 60 mg/kg. Nitrate plus nitrite as N was below detection limits on all months sampled.

North Bar Lake Total organic carbon remained around 4% in the limnetic zone throughout the season, and around 1% in the littoral zone. TKN was variable, with concentrations ranging from 1062-6300 mg/kg in the limnetic zone and 114-1490 mg/kg in the littoral zone. Concentrations of total phosphorus were also somewhat variable, ranging from 110-634 mg/kg in the limnetic zone and 24-64 mg/kg in the littoral zone. Patterns of concentrations were different between the two sampling locations and among nutrient variables. Nitrate plus nitrite as N was below detection limits on all months sampled.

Beaver Lake Total organic carbon was higher in the limnetic zone of Beaver Lake than most lakes included in the study. It ranged from 15-17% during the sampling season. These high percentages did not extend to the littoral zone, where total organic carbon was only 0.2-0.3%--among the lowest for any of the lakes in this study. Total phosphorus concentrations in the limnetic and littoral zones followed a similar pattern with increases in June and September. Concentrations ranged from 737 to 1100 mg/kg in the limnetic zone and from 19 to 35 mg/kg in the littoral zone. TKN was very high in the limnetic zone of Beaver Lake; it remained almost an order of magnitude higher than in the other lakes, but for two. Concentrations remained constant throughout the season, albeit at a high level, and ranged from 18000 to 21000 mg/kg with a high concentration in the first sample. In the limnetic zone, nitrate plus nitrite as N was around 6 mg/kg in July and September but was below detection limits in August. Concentrations were lower in the littoral zone where nitrate plus nitrite as N was 0.14 mg/kg in July, 0.31 mg/kg in September, and below detection limits in August.

Grand Sable Lake Low percentages of total organic carbon were present in Grand Sable Lake. TOC was about 4% in the limnetic zone and 0.4-2.6% in the littoral zone. Total phosphorus was also stable through the sampling season with higher concentrations in the limnetic zone. Limnetic concentrations ranged from 426 to 675 mg/kg, and littoral concentrations were 92-124 mg/kg. TKN was constant with concentrations of 2440-3730 mg/kg in the limnetic zone and 364-930 mg/kg in the littoral zone. Nitrate plus nitrite as N was only detected in July in the limnetic zone at a concentration of 1.4 mg/kg. It was not detected in any other limnetic samples or littoral samples.

Sargent Lake In the limnetic zone of Sargent Lake, total organic carbon was between 10-12% through the sampling season. Concentrations were much lower in the littoral zone, but they remained stable around 1-2%. Total phosphorus had more variation in both the limnetic and littoral zones. In June, the limnetic concentration was 65 mg/kg, but by July, the concentration had increased to a high of 868 mg/kg. Concentrations then decreased but remained above 500 mg/kg. Similarly, TKN in the limnetic zone was low in the June sample, 864 mg/kg but then increased to a high of 13800 mg/kg by August. Patterns were not as noticeable in the littoral zone where total phosphorus and TKN were stable throughout the season. Littoral phosphorus ranged between 171 and 292 mg/kg, and littoral TKN ranged from 499-1140 mg/kg. Nitrate plus nitrite as N was below detection limits in all samples.

Siskiwit Lake Total organic content in Siskiwit Lake remained constant through the sampling season, ranging from 2-5% in the limnetic zone and 0.3-0.4% in the littoral zone. Both total phosphorus and TKN were relatively low in the first sampling and increased substantially in July. Total phosphorus in June was 384 mg/kg and increased to 2170 mg/kg in August. TKN was 539 mg/kg in June and increased to a high in August of 5030 mg/kg. Both total phosphorus and TKN remained at steady concentrations in the littoral zone: 447-602 mg/kg of phosphorus and 230-467 mg/kg of TKN. Nitrate plus nitrite as N was below detection limits in all samples.

Locator Lake Total organic carbon of the limnetic sediments of Locator Lake was higher than almost all of the lakes studied. It ranged from 15-19% while the littoral total organic carbon was between 1 and 6%. In June, total phosphorus (111 mg/kg) and TKN (477 mg/kg) in the limnetic zone were relatively low, but concentrations spiked in July to 2120 mg/kg total phosphorus and 20300 mg/kg TKN. Concentrations of both nutrients then decreased for the remaining two months of the sampling season. In the littoral zone, TKN was much lower with concentrations ranging between 130 and 327 mg/kg. Total phosphorus was also lower in the littoral zone with concentrations ranging from 700 to 3320 mg/kg. Nitrate plus nitrite as N was only detected in the July littoral sample at a concentration of 0.033 mg/kg. It was below detection limits in all other samples analyzed.

Mukooda Lake In the limnetic zone of Mukooda Lake, total organic carbon ranged from 7-10%. It was much lower in the littoral zone, with composition between 0.4 and 1%. Very unusual nutrient results were recorded for the limnetic zone of Mukooda Lake in the June samples: total phosphorus was 0.2 mg/kg and TKN was 1.8 mg/kg. These low concentrations were distinct outliers in any lake comparisons. Concentrations were more typical during the rest of the season, with concentrations ranging from 622-6100 mg/kg total phosphorus and 7510 to 15900 mg/kg TKN in the limnetic zone. The highest concentrations of both nutrients were recorded for July. Littoral nutrient concentrations were generally lower. TKN concentrations throughout the season ranged from 296-932 mg/kg, and total phosphorus ranged from 102 to 221 mg/kg. Nitrate plus nitrite as N was below detection limits in all samples analyzed.

Long Lake Sediment samples from Long Lake only included littoral samples because the lake is too shallow to have a limnetic zone. Total organic carbon over the course of the sampling season was stable between 0.5 and 2.7%. Total phosphorus was also very constant with a range of 62-77 mg/kg. There was some variation in sediment TKN, but overall it was also stable. The highest concentration was recorded in June at 684 mg/kg, with a decrease to a low of 189 mg/kg by August. Nitrate plus nitrite as N was only detected in the August sample at Long Lake when the concentration was 2.1 mg/kg.

N:P Ratios

The ratio of nitrogen and phosphorus is used for distinguishing the limiting nutrient in a lake. Because nutrients tend to move from the water column to the sediment, there is often a buildup of nutrients in the sediment that may become available to the system during lake turnover, which occurs in the fall and spring for most of the lakes included in this study. Beaver Lake is a polymictic lake, and therefore turnover occurs periodically through the season.

When N:P was compared among all lakes, both Siskiwit Lake and Beaver Lake tended to separate from the other lakes in the study. Beaver Lake was typically in its own group after an analysis of variance on limnetic zone results followed by either a Student-Newman-Keuls or a Duncan post-hoc test. Beaver Lake had the highest ratios in the limnetic zone through most of the season (Figure 7-1). The mean N:P ratio for Beaver Lake was 21.8. Siskiwit Lake had the lowest mean ratio, with 2.0. In the littoral zone, Loon Lake clearly separated from all of the other lakes in and analysis of variance, with a mean N:P ratio of 31.5. Most of the lakes clustered within a certain range, but Loon Lake was clearly well above that range (Figure 7-2). At the other extreme, Siskiwit Lake had a mean N:P of 0.73.

Overall, individual lakes had stable concentrations of nutrients in the sediment over the entire season. Sediment nutrients do not fluctuate nearly as much as nutrients in the water because they are only mixed periodically during the season. Even so, nutrients are often lost to the lake system when they settle to the lake bottom and are buried. The sediment concentrations are therefore stable because nutrients are continuously settling to the lake bottom.

Figure 7-1. Ratios of nitrogen to phosphorus in sediment samples collected from the limnetic zone of each lake.

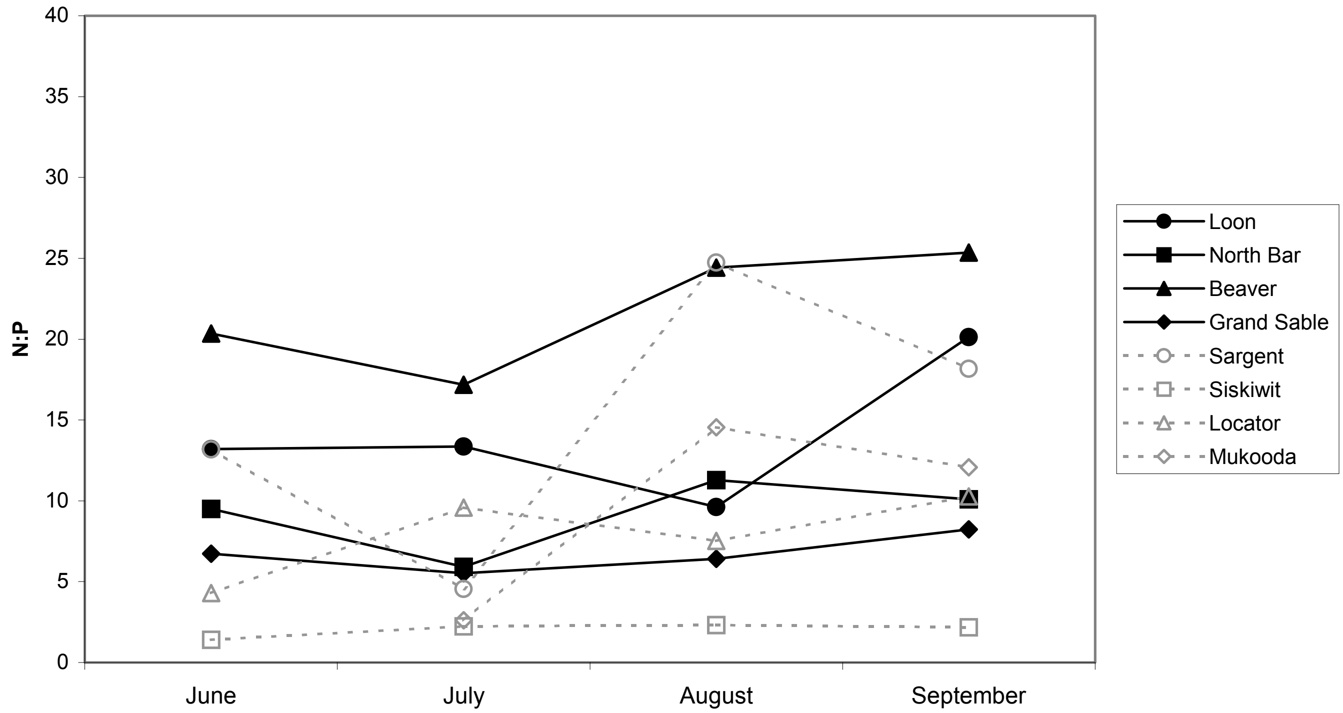


Figure 7-2. Ratios of nitrogen to phosphorus in sediment samples collected from the littoral zone of each lake.

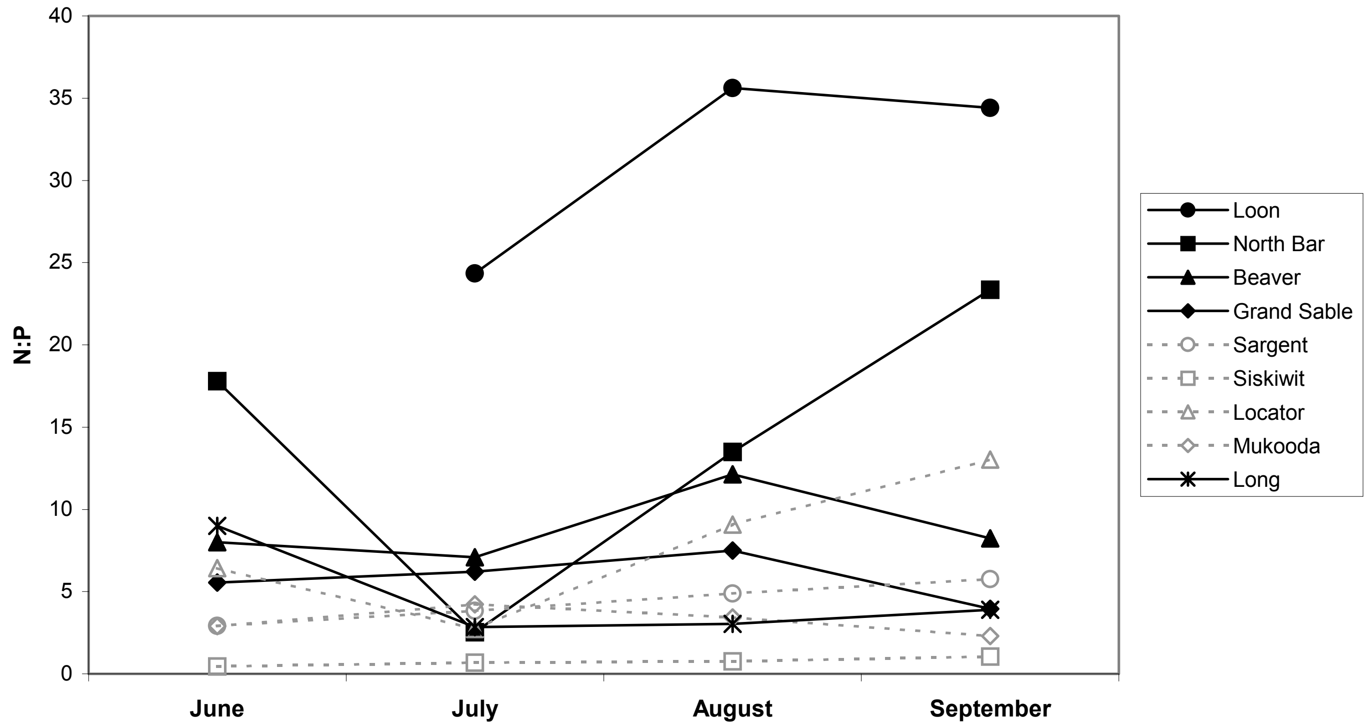


Table 7-1. Means and standard errors for total organic carbon (TOC), total phosphorus (P), and total Kjeldahl nitrogen (TKN) in the limnetic zones of study lakes. Sediments were collected between June and September, 1997.

		TOC	TP	TKN
Loon	Mean	5.17	303.35	4213.0
	N	3	4	4
	Std. Error	0.12	83.41	1232.13
North Bar	Mean	4.03	452.75	4053.50
	N	3	4	4
	Std. Error	0.30	116.85	1121.68
Beaver	Mean	18.7	913.6	19451.23
	N	3	4	4
	Std. Error	0.80	89.71	647.12
Grand Sable	Mean	4.03	508.78	3352.9
	N	3	4	4
	Std. Error	0.15	66.53	307.72
Sargent	Mean	11.87	500.88	6978.5
	N	3	4	4
	Std. Error	0.75	165.26	2864.74
Siskiwit	Mean	4.23	1341.0	2937.25
	N	3	4	4
	Std. Error	1.05	398.72	977.15
Locator	Mean	17.13	1410.25	12619.25
	N	3	4	4
	Std. Error	0.98	454.24	4247.60
Mukooda	Mean	8.57	2462.0	11023.33
	N	3	3	3
	Std. Error	0.65	1819.04	2516.08

Table 7-2. Means and standard errors for total organic carbon (TOC), total phosphorus (P), and total Kjeldahl nitrogen (TKN) in the littoral zones of study lakes. Sediments were collected between June and September, 1997.

		TOC	P	TKN
Loon	Mean	0.97	52	1358.25
	N	3	3	4
	Std. Error	0.17	4.08	253.9
North Bar	Mean	0.59	40.25	603.0
	N	3	4	4
	Std. Error	0.26	9.06	303.55
Beaver	Mean	0.25	29.43	249.35
	N	3	4	4
	Std. Error	0.027	3.6	10.35
Grand Sable	Mean	1.25	104.68	622
	N	3	4	4
	Std. Error	0.68	7.32	120.19
Sargent	Mean	1.37	217.0	942.25
	N	3	4	4
	Std. Error	0.27	26.14	150.49
Siskiwit	Mean	0.36	518.75	373.75
	N	3	4	4
	Std. Error	0.029	37.91	50.73
Locator	Mean	3.47	242.0	1956.5
	N	3	4	4
	Std. Error	1.36	40.96	690.45
Mukooda	Mean	0.65	184.5	608.0
	N	3	4	4
	Std. Error	0.18	28.15	144.11
Long	Mean	1.83	70.23	336.85
	N	3	4	4
	Std. Error	0.68	3.56	116.69

Metals and Contaminants

Data were received on chemicals measured or detected during 1997 in sediments from the following lakes in National Parks:

Chemicals	Lake	Park
Metals	Long	Indiana Dunes
Organic pesticides	Loon	Sleeping Bear Dunes
Volatile organics	North Bar	Sleeping Bear Dunes
Semi-volatile organics	Beaver	Pictured Rocks
	Grand Sable	Pictured Rocks
	Sargent	Isle Royale
	Siskiwit	Isle Royale
	Locator	Voyageurs
	Mukooda	Voyageurs

The concentrations in sediments were screened for toxicity by comparison with sediment benchmark values for toxicity to freshwater biota. These benchmark values were assembled by scientists at the Great Lakes Science Center in two tables, used for screening residues in Lake Erie/Lake St. Clair NAWQA and the Illinois River basin NAWQA. The former table relied on benchmarks from the Oak Ridge National Laboratories (URL http://www.hsrd.ornl.gov/ecorisk/tm95r4.pdf) and NOAA (URL http//www.orca.nos.noaa.gov/projects/nsandt/sedimentquality.html). The latter table incorporated consensus-based sediment quality guidelines for freshwater ecosystems from MacDonald et al. (MacDonald et al. 2000). For chemicals without consensus values, available values from the first table were used. In evaluating the residues in the lakes, we did not normalize by organic content of the sediment, although some benchmark values are normalized to 1% organic carbon. The raw data from which these compiled benchmarks were derived are available upon request from Dora Reader, USGS, Ann Arbor, Michigan (Dora_Reader@usgs.gov).

The benchmark values used in the table below are for comparison with site-specific measured values. The benchmark for most chemicals was a Consensus-Based Sediment Quality Guideline (MacDonald et al. 2000). We used the Threshold Effect Concentration (TEC) value, which is the concentration below which no adverse effects would be expected. As seen in MacDonald's paper, the TEC value is a geometric mean of several prior criteria from the literature. For some of these criteria, the range is over an order of magnitude (e.g., pyrene), whereas for others (e.g., copper) the range is about a factor of two. Hence, when comparing a measured value from the National Parks, with these benchmarks, a relatively small deviation from the benchmark is not meaningful. Consensus values are only established for 28 chemicals. For other chemicals, we used our table (CRITERIA2), which is a compilation of reference values from several agencies, both USA and Canadian. These values are mainly from the Oak Ridge National Laboratory, Oak Ridge, TN, website: http://www.hsrd.ornl.gov/ecorisk/ecorisk.html

Organic pesticides and PCBs (measured as Aroclors) were generally non-detected. Where detected, they were below the consensus-based sediment quality guidelines or other benchmarks.

Metals and one PAH exceeded the benchmark values in the limnetic zone at three of the lakes: Beaver, Sargent, and Siskiwit Lakes and in the littoral zone at Siskiwit Lake (Table 7-3).

Table 7-3. Benchmark values for metals and contaminants and lakes that exceeded the values.

Chemical	Benchmark-mg/kg	Beaver	Sargent	Siskiwit-Limnetic	Siskiwit-Littoral
Cu	31.6	37.9	80.2	89.9	
Fe	20,000	20,800	25,500	38,900	
Hg	0.18	0.35	0.21		
Ni	22.7		35.1	35.7	24.9
Pb	35.8		53.6		
V	NA		72.6	67.1	
Zn	121	145			
Benzo(e)pyrene	Ca. 0.1		0.590		

None of these measured values exceed the Probable Effects Concentration (PEC) (Table 7-4). For those chemicals that exceed the Threshold Effects Concentration (TEC), I recommend that additional samples be collected at the site and analyses be conducted to substantiate the measured value. Since most of the exceedences of TEC are for metals, a measurement of acid volatile sulfides of the sediment would be useful to evaluate the bioavailability of the metals to organisms, and hence their likely toxicity (Hansen et al. 1996, Leonard et al. 1996, Leonard et al. 1999).

Table 7-4. Probable Effects Concentration for metals and contaminants that exceeded benchmark values.

Chemical	PEC Benchmark-mg/kg	Beaver	Sargent	Siskiwit-Limnetic	Siskiwit-Littoral
Cu	149	37.9	80.2	89.9	
Fe	40,000	20,800	25,500	38,900	
Hg	1.06	0.35	0.21		
Ni	48.6		35.1	35.7	24.9
Pb	128		53.6		
V	NA		72.6	67.1	
Zn	459	145			
Benzo(e)pyrene	About 1.0		0.590		

Where PEC = Probable Effects Concentration

The analytical laboratory reported many chemicals in the sediments. Most were very low, just above detection levels. For the organics, the lab qualitative analysis was restricted to library matches of similar retention times on the GS-MS. It was obvious to all organic chemists consulted that most of these results were routine analytical artifacts and not of environmental significance. If sediment analysis had demonstrated a suspicious presence for a particular contaminant, the laboratory would have been directed to rerun the test against a standard reference for that chemical. A large number of the organic chemicals listed are most likely naturally occurring biological compounds and not contaminants (e.g., esters, steroids, fatty acids, and cholesterols). Internal standards used by the laboratory are listed separately.

The vast majority of organic and inorganic chemicals retrieved from the lake sediments were below acute or chronic toxicity thresholds or average benchmarks. Most chemicals were within anticipated background levels. Thus, the discussion will be restricted to potential contaminants that exceed benchmark levels.

We discuss individual chemicals below much of which is derived from Irwin et al. (Irwin et al. 1997). The reader is referred to this document for a far more thorough discussion of biological, toxicological and environmental impacts of water and sediment contamination. It is available on the Internet via the NatureNet portion of the National Park Service Home Page (www.nps.gov).

Copper

Copper most commonly occurs in the aqueous state in the +2 oxidation state. It commonly forms stable salts with oxides, chlorides, sulfides, and carbonates. It is found in sediments in the oxidized form, and in copper-rich areas such as Isle Royale it can be found as native copper. It is one of nearly 130 priority pollutants listed by U.S. EPA. As such, its toxicity has been well demonstrated for vertebrates, invertebrates, algae, and plants. It is also an effective toxicant: copper has been routinely used as an aquatic biocide for weeds and algae. Copper sulfides (often associated with acid mind drainage) have been of special concern. It is possible for fish to exhibit toxicity effects from contact to contaminated sediments by direct ingestion or mere dermal contact. Toxicosis and cell damage have been reported. Freshwater organisms seem more sensitive to lethal and growth effects of copper. Copper concentrations tend to be highest in sediments because the metal is generally tightly bound to sulfides or organics and sometimes deeply buried, far from bioactivity. When dissolved, copper is particularly toxic.

According to EPA, Great Lakes Harbors having sediments with copper concentrations in excess of 50 mg/kg dry weight may be considered "heavily polluted" and concentrations between 25 and 50 mg/kg dry weight were classified as “moderately polluted” (Ingersoll and Nelson 1989, Beyer 1990). The Illinois EPA classified sediments having concentrations of copper higher than 60 mg/kg dry weight as elevated (Ingersoll and Nelson 1989), and the International Joint Commission stated that levels under 21.1 mg/kg dry weight were normal (Ingersoll and Nelson 1989). Other limits determined include a geometric mean of 35 mg/kg dry weight adjusted for sediment texture (National Oceanic and Atmospheric Administration 1991) and a threshold of 35.7 mg/kg dry weight (Batts and Cubbage 1995).

Thus, it appears that Beaver Lake is moderately high in copper and the Isle Royale lakes, Sargent and Siskiwit, clearly exceed recommended benchmarks. These sediments may contain enough copper to have toxicological effects. Copper is an abundant natural component of Isle Royale sediments but apparently not Pictured Rocks (Jerry Belant, Resource Mangament Specialist Pictured Rocks National Lakeshore, *personal communication* April 20, 2001). Much of the copper in Isle Royale sediments may be due to background levels of the metal introduced by sedimentation or historical mining. Copper is occasionally introduced to a lake when it is added to control nuisance plants, and historical records would determine if such measures were ever used in these lakes.

The toxicity and environmental implications of these elevated copper levels depend on a number of factors. The nature of the ecosystem, the community that inhabit the sediments, the life history and behavior of those organisms, the distribution of the chemicals in the sediment and most importantly, the bioavailability of copper. Copper may be tightly sequestered by sulfides or organics and not freely available to the biota. On the other hand, filter feeding amphipods, mollusks, midges and fish may ingest copper-laden sediment and convert the chemical to toxic states that could affect growth, reproduction, and ultimately survival of selected organisms. To assess fully the impact of copper on the infauna of these lakes, bioassays may be attempted on collected sediments, using organisms representing all trophic levels: primary producers, primary and secondary consumers (e.g. *Selenastrum Lumbriculus or Tubifex,* midges, amphipods, and fat head minnows). These tests should include chronic and acute testing. Examining the effects of elevated copper on embryonic development of selected organisms might also be considered. Anthropogenic and natural sources determination of high copper levels should be further investigated.

<u>Iron</u>

After silica, iron is the most common element in the earth's outer crust. Nonetheless, the concentration of iron dissolved in lakes is relatively small. This is largely because iron readily oxidizes into an insoluble state in oxygenated water, forming a precipitate by itself and with other minerals such as iron phosphate. Under anaerobic conditions, sediments can release iron and other minerals that are bound to it. The most common form of iron is iron oxide, ferrous (+2) or ferric states (+3). Natural sources of iron include igneous rock and re-precipitated sedimentary species such as ferrous sulfide. Magnetite is a very common form of iron sediment and is highly resistant to dissolution. Iron toxicity for aquatic organisms is unclear but mostly depends on iron bioavailability. The levels of iron found at Beaver, Siskiwit, and Sargent are of marginal concern and are likely within levels expected for deep lakes. Iron naturally precipitates and accumulates in lakes over time and as such, it is an important component of the lakes nutrient and mineral cycling. Especially, important is iron's ability to bind with phosphorus in aerobic conditions. This reduces algal production in the epilimnion and during mixing. During stratification anaerobic conditions allow for iron and phosphorus to be released, which helps account for spring and autumn algal blooms. The formation of iron sulfide, especially in the hypolimnion, is essentially non-reversible. This and the formation of other forms of insoluble iron means that in deep older lakes iron content in the profundal sediments maybe relatively high. This does not necessarily mean that the sediments are toxic. Organisms living within the profundal zone likely adapt well to these iron-rich conditions. Unless there is evidence of anthropogenic input from historical mining or other human perturbations, the results of iron analysis provide no reason for major concern. Regardless, toxicological evaluation of these sediments for iron would be difficult since duplication of the specific conditions such as high redox, cold water, and anaerobic condition would be difficult.

<u>Mercury</u>

Mercury is one of U.S EPA's 129 priority pollutants and is a major concern when allowed to enter lake water or sediment (Keith and Telliard 1979). Of special concern are its toxicity and its ability to bioaccumulate in the tissues of plants and animals. Although it is most stable in its elemental stage, mercury easily forms chlorides and hydroxide complexes. Methane-producing bacteria generate organic complexes such as methyl mercury, and these and other organic forms of mercury are toxic. Methyl mercury bioaccumulates in the food web, and this affects reproduction, growth, and survival of top consumers such as predatory vertebrates.

Dissolved mercury is highly toxic to all aquatic organisms even in small quantities. The implications of absolute mercury content in sediments are more problematic. Mercury can form extremely insoluble complexes with ions such as sulfides and is easily incorporated into detritus. Nonetheless, the an excess of mercury is a matter of environmental concern since it enters and moves through the ecosystem with relative ease and may be ingested directly by primary consumers. Terrestrial species (including humans) feeding on contaminated aquatic species can become ultimate reservoirs of mercury through bioaccumulation.

Some major sources of mercury are coal and oil fired power generation facilities, metal smelting, landfills, waste incineration, and atmospheric deposition from remote locations. Of these, atmospheric deposition is a more common means of accumulation of mercury in lakes (US Geological Survey 1996). Once in the system, mercury can be converted to methyl or alkyl mercury, the most biologically and ecologically damaging form of the metal.

Beyer (Beyer 1990) considered sediment mercury concentration in excess of 1 mg/kg as heavily polluted. NOAA found that 38 of 175 sites examined exceeded 0.41 mg/kg. This was the mercury level they believed had the potential for toxicity (Agency for Toxic Substances and Disease Registry 1992). Wisconsin's interim criterion for disposal of sediment in water is 0.1 mg/kg. The lowest effect level for the St. Lawrence River Interim Freshwater Sediment Criteria is set at 0.2 mg/kg dry weight, and Environment Canada Interim Sediment Quality Assessment Values, 1994 threshold effect is 0.174 mg/kg dry weight with a probable effect level of 0.486 mg/kg dry weight (Irwin et al. 1997).

Only Beaver and Sargent Lakes exceeded the average benchmark for mercury. Sediment concentration of Beaver (0.35 mg/kg) was nearly twice the benchmark (0.18 mg/kg), and Sargent Lake was only slightly above the benchmark (0.21 mg/kg), which is similar to results reported by Gorski et al. (Gorski et al. 2001) who found concentrations between 0.079 and 0.218 mg/kg with highest concentrations in the shallow sediments. Lower concentrations corresponded with pre-industrial background concentrations.

Increased sediment mercury may cause limited toxicity of lake biota and influence ecosystem functioning at Beaver Lake. The concentrations in Sargent are more marginal, but a recent publication by Kallemeyn (Kallemeyn 2000) demonstrates a strong potential for bioaccumulation of mercury in local fish. Thus, concerns about mercury problems in both Isle Royale lakes cannot be discounted.

We recommend that sediments be re-analyzed for mercury at Beaver Lake to confirm the results found during this study. Confirmation of Sargent Lake are likely unnecessary since Gorski et al. (2001) found values for mercury for Sargent essentially identical to our findings. Gorski et al. (2001) assessed possible sources of mercury within the Richie and Sargent Lake watersheds but not Siskiwit. They doubted direct human impact from mining and smelting on the island because these activities ended in the 19th Century. Siskiwit and Beaver Lake watersheds should be inspected to be certain that direct human-induced inputs that could be mitigated have not been overlooked. The biological effects of these sediments should also be evaluated. This can be most readily be accomplished by standard sediment toxicity tests on several trophic levels but should focus on benthic organisms present in these lakes (e.g. amphipods, segmented worms, midge larvae). Finally, fish tissue should be surveyed to see whether mercury is concentrating in higher trophic levels. This exercise would be especially important in Beaver Lake where levels are higher and intensive fishery contaminant research is limited.

<u>Nickel</u>

Nickel is a common constituent in the earth's crust. This metal, like cobalt, easily substitutes for iron-magnesium complexes in igneous rocks. It tends to precipitate out of solution with iron, especially in the presence of manganese. Nickel is generally bivalent when dissolved in water. Most contamination in water bodies is derived from water disposal and mining of the metal. The metal is very commonly used in industry to substitute for iron to provide corrosive resistance. The concentration of nickel in surface water is usually quite low. In North American rivers, for instance, the median concentration is 10 ug/l (Durum and Haffty 1963).

Nickel is moderately toxic to invertebrates when dissolved. For instance, the acute LD50 for *Daphnia magna* was reported by EPA to be 0.85 mg/l, and the 96-hour LC50 for the midge, *Chironomus* was 8.6 mg/l 998 (US EPA 1997). The 24 and 96-hour LC50 of the caddisflies were 48.4 and 30.2 mg/l. Thus, it appears that these benthic macroinvertebrates are moderately tolerant to intermediate levels of nickel ions. Acute bioassays performed by EPA resulted in LC50s ranging from 2.916 to 17.678 mg/L for 96-hour exposures, but most LC50 were about 9 mg/l.

NOAA suggested a level of 69 ppm dry weight as a benchmark for sediments to be classified as high in nickel (National Oceanic and Atmospheric Administration 1991). NOAA reported a mean concentration of 35 ppm of nickel in fine-grained sediment. Sediments with concentrations of 34 ppm were considered typical. In a 1977 review, EPA suggested that Great Lakes Harbors having nickel concentrations in excess of 50 mg/kg dry weight be classified as "heavily polluted" (Ingersoll and Nelson 1989).

Sargent and Siskiwit Lakes had nickel concentrations moderately above the average benchmark calculated by Reader, Hickey, and Begnoche (see table). The nickel residue in these lakes is within the typical range, as classified by an EPA 1977 report (Beyer 1990), and below NOAA's benchmark (National Oceanic and Atmospheric Administration 1991). The average amount of nickel in the earth's crust is 60-90 mg/kg (National Research Council Canada 1981), and the average igneous rock concentration is 75 mg/kg dry. U.S. soil has an average nickel content of 20 mg/kg dry weight (Irwin et al. 1997). Thus, the range of nickel found in Sargent and Siskiwit Lakes is well within expected background concentrations, and our calculated benchmark is likely overly conservative.

<u>Lead</u>

Lead occurs commonly in sedimentary rocks. It is not especially mobile in the environment due to its tendency to form insoluble salts. Lead co-precipitates with other metals, especially manganese and is readily absorbed to organic and inorganic surfaces. Because of this, high levels of lead do not usually occur in open natural waters. Historically, leaded gasoline has been an important source of lead in lake sediments through atmospheric deposition and direct runoff, which may remain buried in the sediment of many lakes. Lead shot from hunting and target practice is another important source of lead. This contribution can be directly toxic to wildlife when ingested (a few pellets can kill a goose or duck) or indirectly toxic to aquatic animals when it is dissolved in water. The toxicity of lead is largely due to its cumulative neurological effects, but kidney disease, enzymatic malfunction, and anemia are common side effects as well.

Lead is toxic in all forms to most aquatic organisms. The level of lead necessary for toxicity is complicated by the chemistry of the metal within its media matrix and the modality of biological effects. For instance, lake water hardness, alkalinity, and salinity may have a strong influence on the toxicity and bioavailability of lead. The means of exposure is important since ingestion, absorption, and adsorption would affect organisms differently. Finally, individual species are affected differently. For instance the LC50 for *Ceriodaphnia reticulata* (water flea) was 0.53 mg/L for a 48-hr exposure, 5.1 mg/L for a 48-hr exposure LC50 for *Dugesia tigrina* (flatworm) and 6.2 mg/l for *Perna viridis* (green mussel) for a 24-hr exposure. LC50s for *Micropterus dolomieui* (smallmouth bass) ranged from 2 to 29 mg/L for 96-hr exposures (US EPA 1997).

NOAA National Status and Trends Program (1984-1990) considered 89 ug/g dry weight at 4.6% TOC dry weight to be their benchmark for high lead sediment content (Irwin et al. 1997). In a 1977 study by EPA, Great Lakes Harbors sediments with lead concentrations higher than 60 mg/kg dry weight were classified as "heavily polluted, and Illinois EPA classifies sediments in excess of 38 mg/kg as "elevated" (Beyer 1990).

Only Sargent Lake sediment exceeded the average benchmark for lead. The average sediment benchmark for lead is 35.8 mg/kg, and Sargent Lake had a concentration of 53.6 mg/kg. It is likely that much of this lead is naturally occurring in the sediments of Isle Royale, although atmospheric inputs cannot be discounted. The influence of historic smelting and mining activity is unknown. Further, it is unknown what toxicological influences this lead might have on local biota. A sediment bioassay would be useful in this regard. Like other heavy metals occurring in Isle Royale, it is important to partition background native sediment content from contaminants. This will be difficult in an area so naturally rich in metals and mining activity.

Vanadium

As a transition metal, vanadium is especially complex in its chemistry. It has common three common oxidation states that are stable in water, but the dominant state is pentavalent. In low oxygen conditions, vanadium is rather insoluble but it is otherwise easily oxidized. One of the important anthropogenic sources of vanadium is coal-fired combustion. Surface water rarely contains concentrations in excess of 10 ug/l.

The mean of vanadium in soils and sediments for the U.S. is 76 ppm with a range of less than 7-500 ppm. The mean for earth's crust is about 150 mg/l and for igneous rock, 135 mg/l (Irwin et al. 1997). EPA guidelines for protecting of aquatic life from vanadium are not available. Sargent and Siskiwit Lakes were well within these ranges.

Zinc

Zinc is a common constituent in earth's crust, occurring at the same concentration as copper and nickel. It is an essential micronutrient for animals and plants, but at high concentrations it is toxic. It is widely used in industry, especially to coat steel (galvanization), and to reduce oxidation in paint and rubber. Mine drainage often contains zinc in excess of 0.1 mg/l and can accumulate to much higher concentrations in the sediment. While human health effects are unclear, it is easily detected by taste even at low levels. The toxicity of zinc to aquatic organism occurs at relatively low levels. Zinc is toxic to plants, algae, invertebrates and fish. Toxicity varies but the EPA's acute ambient water quality a criterion is about 0.12 mg/l for one hour of exposure in water with a hardness of 100 mg/l. LC50s for *Ceriodaphnia reticulata* (water flea) have varied from 0.076 to 0.264 mg/L during a 48-hr exposure. LC50s for

Chironomus sp. (midge) were 21.5 and 18.2 mg/L for 24- and 96-hr exposures, respectively. LC50 for Pimephales promelas (fathead minnow) is most frequently 2.0 mg/l (Irwin et al. 1997).

Great Lakes Harbors sediments were classified as polluted when the zinc content exceeded 200 mg/kg dry weight. EPA considered sediments with zinc content of 90 - 200 mg/kg dry weight as moderately polluted. Illinois EPA, 1984 used 100 mg/kg of zinc as their minimal criteria for a classification of elevated levels (Irwin et al. 1997).

Only Beaver Lake was above the benchmark zinc level of 121 mg/kg with a concentration of 145 mg/kg. As with other elevated levels of metals found in Beaver Lake (i.e. Cu, Fe, Hg) it is not known to what extent these chemicals are natural background or anthropogenic. Further, we do not know the ecological implications of the levels found. It is conceivable that these profundal sediments would act as a sink for deposition and precipitation of many of these metal complexes whether introduced naturally or from human-induced contaminant runoff. As with other metals discussed, it is important to identify naturally occurring metal content in lake sediments from anthropogenic sources. Further, we recommend testing the sediment for toxicity using organisms normally expected in this zone of the lake, perhaps Oligochaetes and midges.

Benzo(e)pyrene

Benzo(e)pyrene was presumably detected in one sample from Sargent Lake sediments. If present, the residue would exceed the average benchmark for this PAH, but it is likely that these results represent a false positive laboratory error. False positive analytical results are common, especially when referencing chemicals using database libraries of chemicals with similar analytical characteristics. Further, it is unlikely that only this PAH would occur exclusive of other isomers and homologues. This particular PAH is one of the more light-weight, volatile PAHs, and it is unlikely to occur in the absence of other volatile or semi-volatile, long-lived counterparts.

Chapter 8 Phytoplankton Results

Loon Lake

Loon Lake was sampled for littoral and limnetic phytoplankton between June and September 1997 and for limnetic phytoplankton from May to September 1998.
1997 Limnetic: Diatoms dominated the phytoplankton community in Loon Lake in June 1997: 95% of the community was diatom species (Figure 8-1). Otherwise only yellow-green algae made up more than 1% of the community. In July, this discrepancy was even more pronounced with a full 97.7% of the community comprised of diatoms and no other planktonic group composing even 1% of the community. Diatom relative abundance dropped off slightly in August as yellow-green numbers rose dramatically to 15%, but by September, yellow-greens decreased with an increase in diatoms. Green algae and crytpomonads composed a small percentage of the community through the season, but greens exhibited a steady increase from June to September.

Total abundance of phytoplankton was highest in July and decreased each month for the rest of the season, with the biggest drop-off between July and August (approximately 1.3 X 10^7 cells or colonies/liter in July; 3 X 10^6 cells or colonies/liter in August) (Figure 8-4). The decrease throughout the season was 1.3 X 10^7 cells/colonies per liter in July to 2.4 X 10^5 in September. A total of 36 different taxa were counted from the 1997 samples.

1997 Littoral: Diatoms also dominated the phytoplankton community in the littoral zone in 1997 (Figure 8-2). In general, community composition in the littoral zone was almost identical to the limnetic zone in relative abundance of phytoplankton groups. The community composition in June was very similar to the limnetic zone, with 93% diatom species and 3% yellow-green species. Diatoms were more abundant in July, composing almost 96% of the community. By August, yellow-greens increased dramatically to almost 25% of the community, but they decreased again in September to 8% of the community. With the decrease in diatoms in September, there was a slight increase in green algae and cryptomonads. Abundant species included *Cyclotella* sp. and *Dinobryon* sp.

Diatoms again dominated in 1998 throughout the season (Figure 8-1). Constituting 68% of the community in May, diatoms were as much as 91% of the community by August. In May, 21% of the community was yellow-green algae, but the pattern throughout the season was opposite that of the diatoms: numbers decreased steadily from May to September. Green algae were low at the start of the 1998 season but increased to 14% of the community by July before the numbers decreased again. Cryptomonads were a constant presence in Loon Lake, composing approximately 2-7% of the community. Blue-greens, dinoflagellates, and euglenoids were present but in very low concentrations.

Total abundance increased throughout the season to a high of 1.3 X 10^6 cells or colonies/liter. The phytoplankton abundance increased steadily from May to September. A total of 48 different taxa were counted from the 1998 samples.

Diatoms dominated the samples collected in May of 1999, as in the previous two years. Diatoms composed 97% of the community, and yellow-greens made up 2.2% of the community. Other algal groups were present in very low numbers or completely absent. Total abundance of organisms (3.6 X 10^6 cells or colonies/liter) was higher than in 1998 but fell within the range found in the previous two sampling seasons.

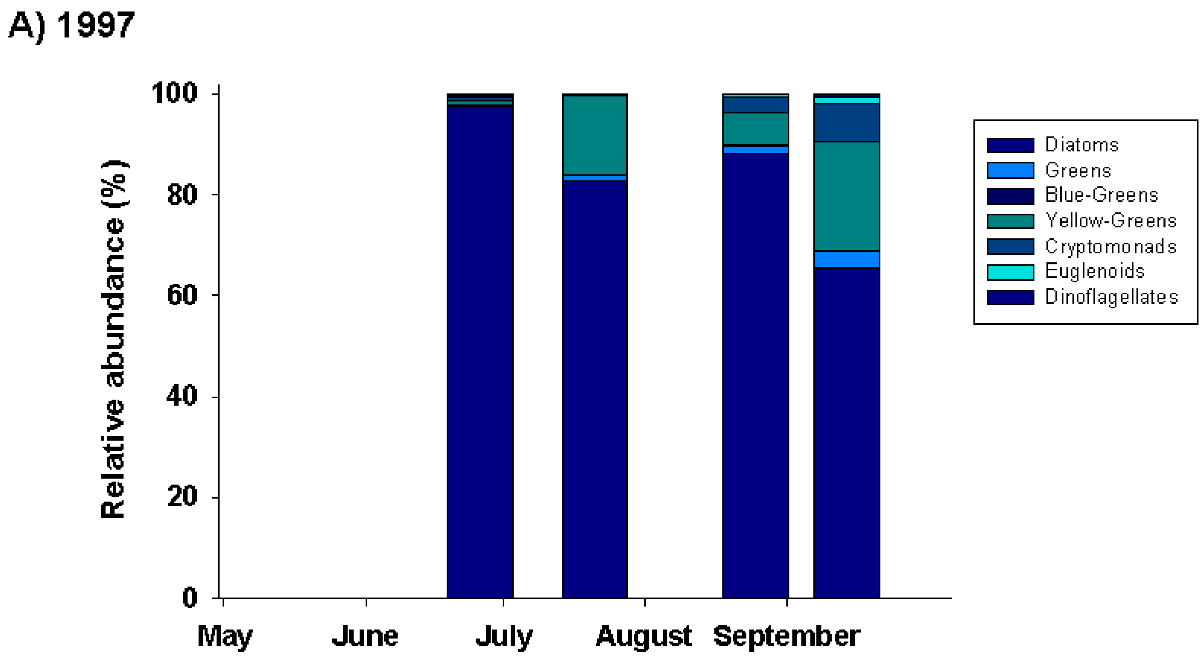
In both 1997 and 1998, *Cyclotella* sp. was the most dominant species encountered (Table 8-1). It was the only species that comprised more than 10% of the community every month sampled, and usually, it comprised well over 50% of the community. Other species abundantly present included *Chlorochromonas* spp. and *Synedra* spp.

Limnetic chlorophyll *a* concentrations were relatively stable in 1997 until a dramatic increase in concentration in September (Figure 8-3). Contrarily, in 1998, chlorophyll *a* was highest in May, decreased through July and then rose again in September.

The predominance of diatoms in both years is a good indication for the nutrient balance in Loon Lake. The 1997 diatom counts follow a pattern consistent with a spring diatom bloom followed by a decrease in numbers and the subsequent increase of other groups (i.e. yellow-green algae). The gradual increase in both diatoms and total phytoplankton abundance over the 1998 season indicates slow growth. This may be attributed to a number of factors including later turnover, lower nutrient availability, or other growth deterrents.

The dominance of *Cyclotella* sp. follows a typical seasonal succession pattern where one diatom species typically dominates early in the season followed by a dominance of several coexisting species. Again, in 1998, the pattern was atypical with delayed single species dominance.

Figure 8-1. Relative abundance of algal groups (common names given for Phyla) collected from the limnetic zone of Loon Lake in A) 1997 and B) 1998. Triplicate samples were collected from 1 meter below the surface above the deepest point in the lake.



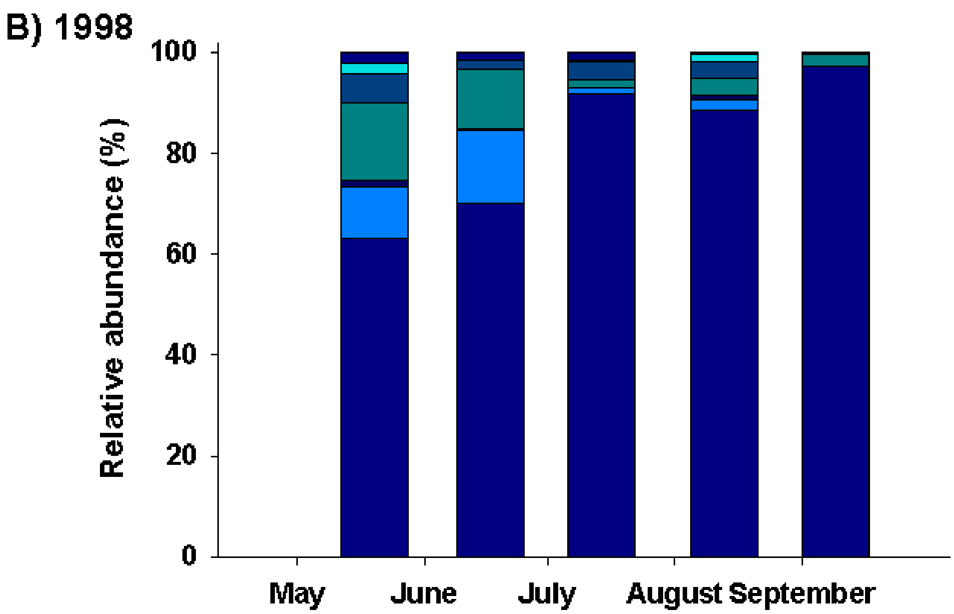


Figure 8-2. Relative abundance of algal groups (common names given for Phyla) collected from the littoral zone of Loon Lake in 1997. Triplicate samples were collected from 1 meter below the surface.

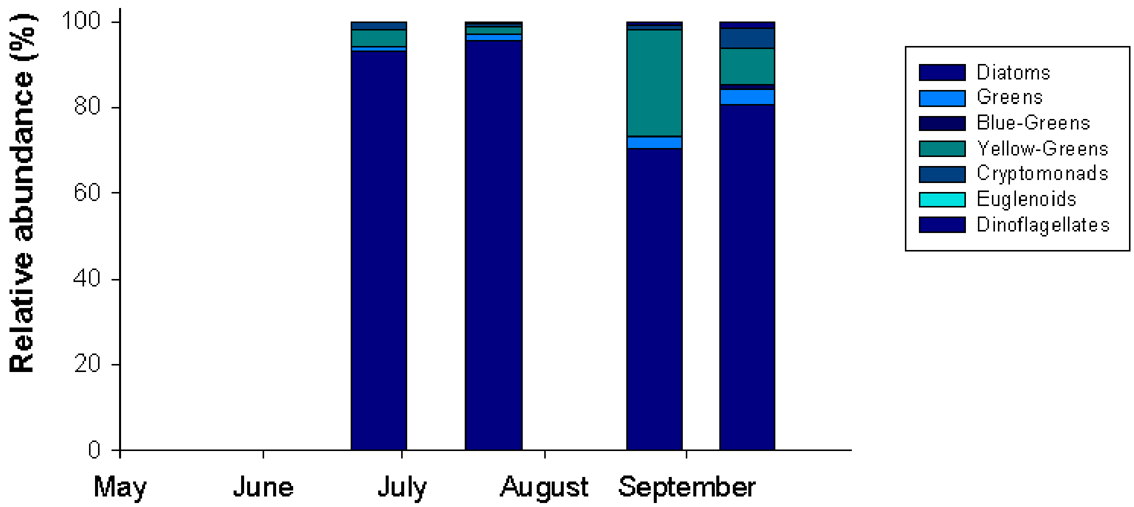


Figure 8-3. Chlorophyll *a* concentration in Loon Lake. Data points represent sample taken from 1 meter below the surface at the deepest point in the lake.

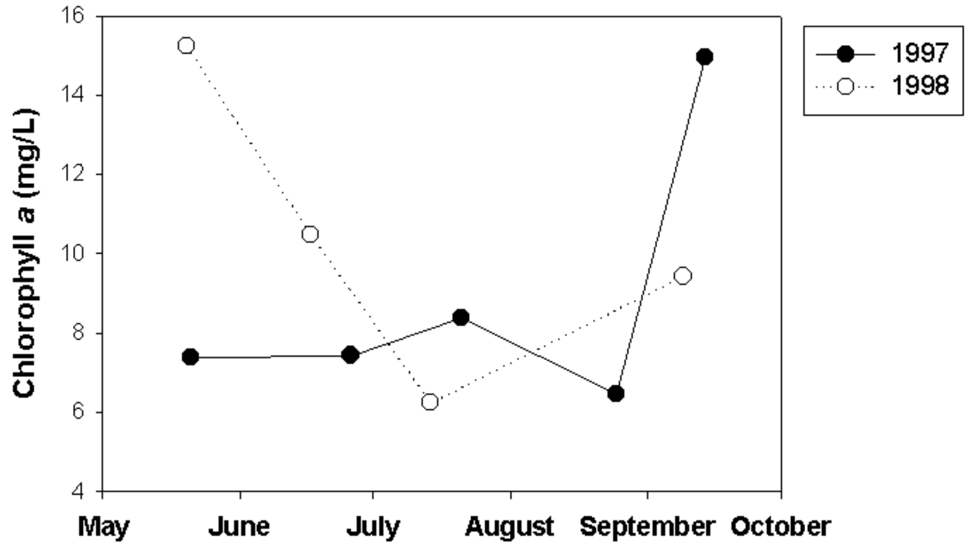


Figure 8-4.Total abundance of limnetic phytoplankton each month in Loon Lake. Counts calculated from three composited samples.

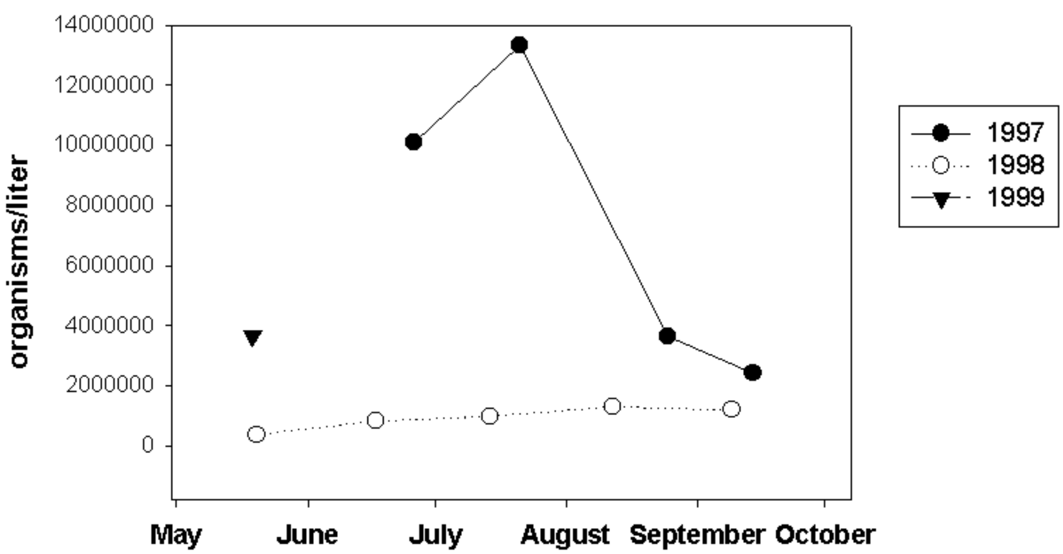


Table 8-1. Phytoplankton taxa that amounted to ≥ 10% of the relative abundance in the limnetic zone of Loon Lake, with percent relative abundance.

		Cyclotella spp.	*Synedra* spp.	*Sphaerocystis schroeteri*	*Chlorochromonas* spp.	*Dinobryon* spp.
Loon	06/26/97	93				
Loon	07/21/97	95				
Loon	08/25/97	80				14
Loon	09/14/97	86				
Loon	05/20/98	25			21	
Loon	06/17/98	60				
Loon	07/14/98	65		12		
Loon	08/12/98	57	29			
Loon	09/09/98	52	23			
Loon	05/19/99	82				

North Bar Lake

North Bar Lake was sampled in the littoral and limnetic zone between June and September 1997 and in the limentic zone from May to September in 1998. Additionally, samples were collected in the limnetic zone in May of 1999.

1997 Limnetic: In 1997, North Bar Lake was clearly dominated by diatom species (Figure 8-5). Diatoms composed 98.6% of the phytoplankton community in June, and although its percentage of community composition decreased gradually through the season, by September, diatoms still accounted for 85% of the community. The only other group to make up a noticeable portion of the community was the yellow-green algae, which increased from 0.4% to 10% over the course of the season. Dinoflagellates, cryptomonads, and green algae were a very small portion of the community but generally increased through the season.

Total abundance was highest early in the season and decreased consistently until a slight reversal in September (Figure 8-8). The sharpest decline was between July and August when counts decreased from 1.1 X 10^7 cells or colonies/liter to 3 X 10^6 cells or colonies/liter. A total of 46 different taxa were collected over the course of the season.

1997 Littoral: The phytoplankton community in the littoral zone was almost identical to that in the limnetic zone (Figure 8-6). Diatoms followed the same pattern, composing almost 99% of the community in June, with percent community abundance decreasing throughout the season. As in the limnetic zone, yellow-greens increased steadily through the season from 0.3% to 10% of the relative community abundance. Green algae were the only other group to compose greater than 2% of the community, and at most, greens were at 4% of community composition. Some of the dominant algal species present included *Cyclotella* sp. and *Chlorochromonas* sp.

Patterns of phytoplankton abundance were very different in the first two months of 1998 (Figure 8-5). Diatoms composed 60% of the community in May, and by June, community composition dropped to 25%. Cryptomonads actually made up 15% of the community in May when yellow-greens made up 7%. By June, however, yellow-green algae were dominant, making up 35% of the phytoplankton community, and cryptomonads had also increased to 26%. Strangely, the pattern of phytoplankton succession took on characteristics of the 1997 season by July of 1998. Diatoms were dominant and remained so for the rest of the season. Diatom community composition was above 90% for those three months, and all other algal groups decreased to 3% or less. A total of 43 different taxa were collected over the course of the season.

The unusual patterns may be explained in part by the overall abundance of phytoplankton (Figure 8-8). There were generally low numbers in May and June (3.7 X 10^5 and 2.7X10^5 cells or colonies/liter) followed by a sharp increase in July (6.2X10^6 cells or colonies/liter). If there was an influx of nutrients in July, it may have led to a late diatom bloom. North Bar Lake is unique in that it is often connected directly to Lake Michigan, depending on the position of a sand bar. Any extended periods of connectedness may have impacted North Bar Lake in a number of ways, including nutrient influx or reduction.

Limnetic chlorophyll *a* concentrations were comparable between 1997 and 1998 (Figure 8-7). In 1997, chlorophyll *a* peaked in September at 13 mg/l after ranging between 3-8 mg/l for the first part of the season. In 1998, concentrations were similar, except the peak (16 mg/l) occurred in August. North Bar Lake could likely be classified as a mesotrophic lake based on the mean chlorophyll *a* concentrations through the season.

In May of 1999, diatoms dominated the phytoplankton community, comprising 98.4% of the community. None of the other algal groups even composed 1% of the phytoplankton.

Overall organism abundance in 1999 ($5.9 X 10^6$ cells or colonies/liter) was higher than samples collected in May of 1998 ($3.7 X 10^5$ cells or colonies/liter).

Overall counts in 1998 were never as high as in 1997. *Cyclotella* sp. was easily the dominant species in all three years, although overall counts, and therefore *Cyclotella* counts, were considerably lower in 1998, especially in May and June (Table 8-2). Other species abundant in individual samples included *Synedra* sp., *Chlorochromonas* sp., and *Cryptomonas* spp.

Figure 8-5. Relative abundance of algal groups (common names given for Phyla) collected from the limnetic zone of North Bar Lake in A) 1997 and B) 1998. Triplicate samples were collected from 1 meter below the surface in the deepest part of the lake.

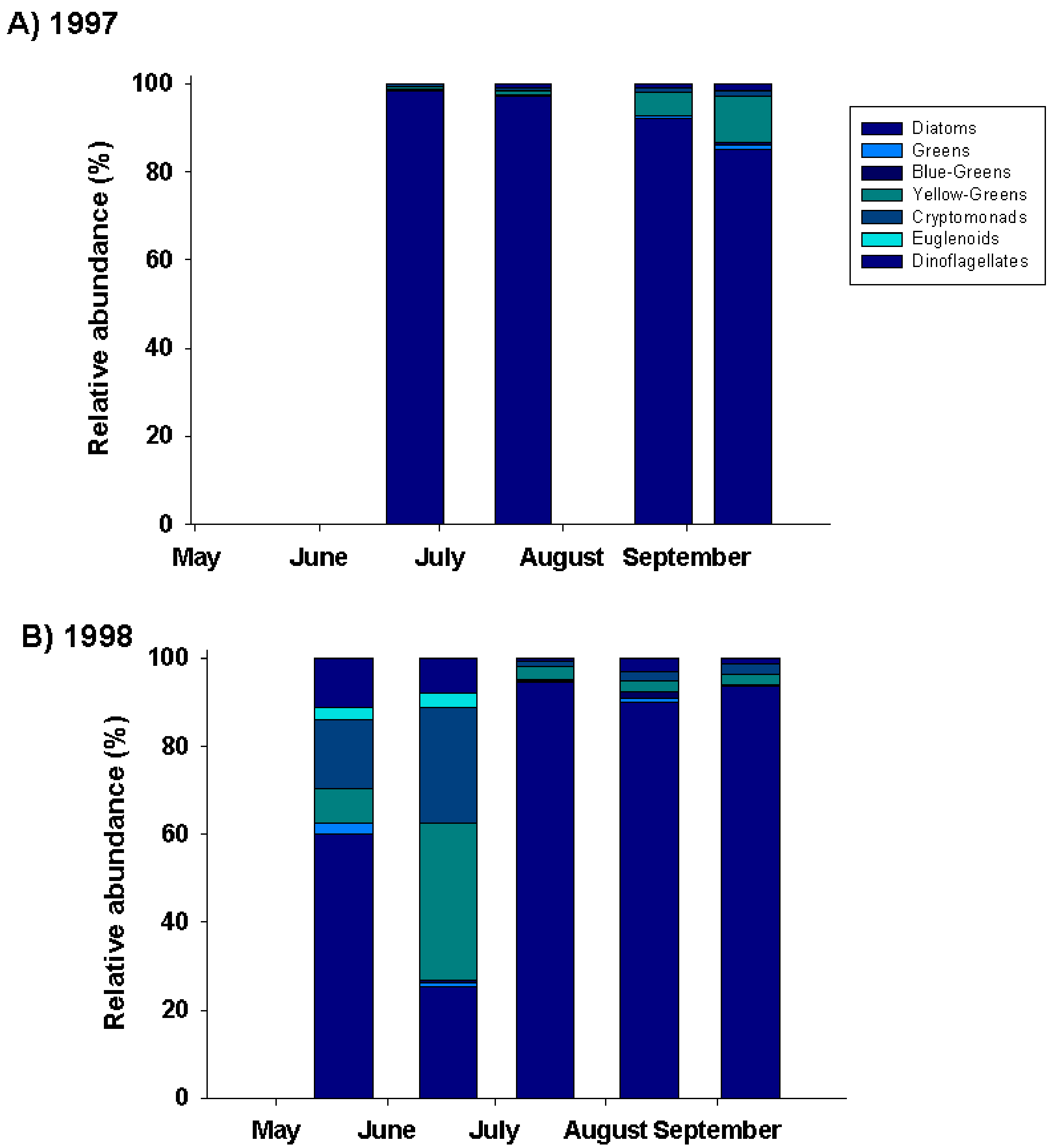


Figure 8-6. Relative abundance of algal groups (common names given for Phyla) collected from the littoral zone of North Bar Lake in 1997. Samples were collected from 1 meter below the surface.

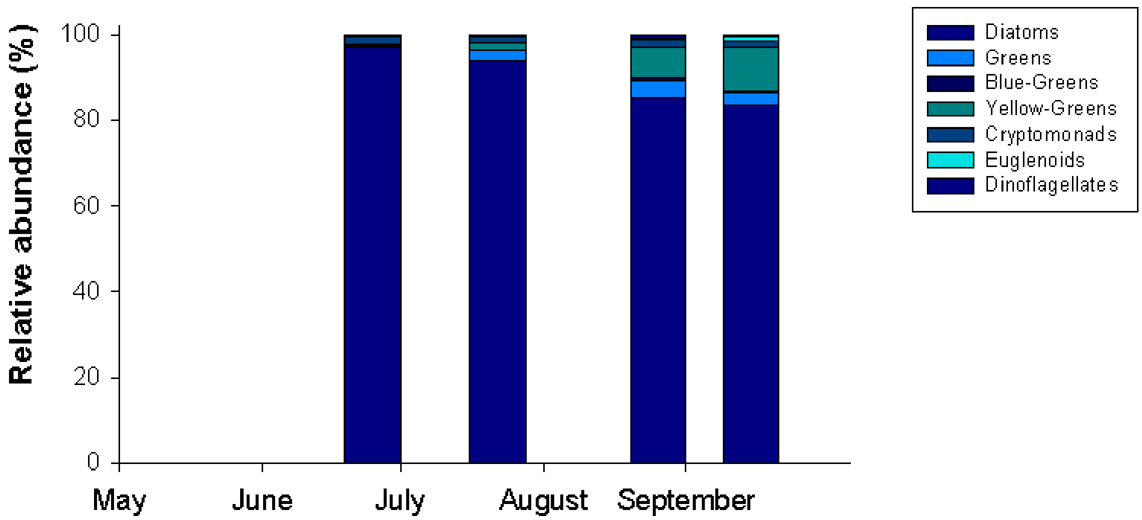


Figure 8-7. Chlorophyll *a* concentration in North Bar Lake. Data points represent samples taken from 1 meter below the surface in the deepest part of the lake.

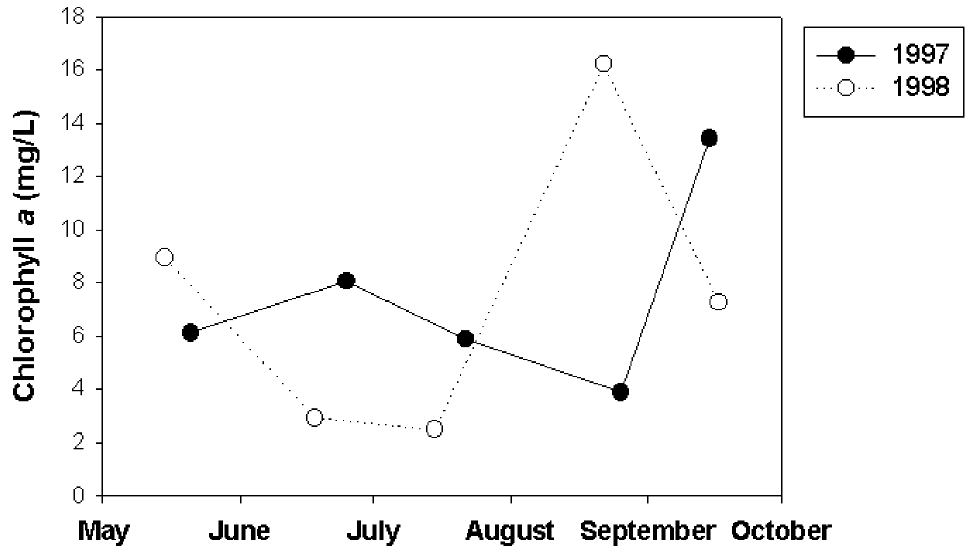


Figure 8-8. Total abundance of limnetic phytoplankton each month in North Bar Lake. Counts calculated from three composited samples.

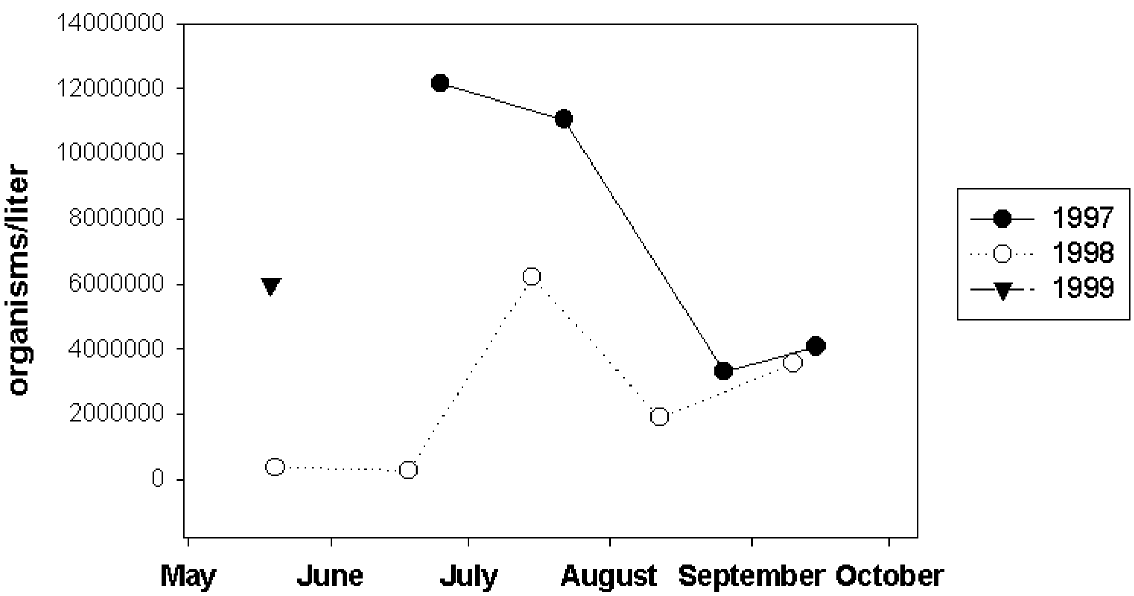


Table 8-2. Phytoplankton taxa that amounted to ≥ 10% of the relative abundance in the limnetic zone of North Bar Lake with percent relative abundance.

		Cyclotella spp.	*Synedra* spp.	*Chlorochromonas* spp.	*Cryptomonas* spp.	*Glenodinium/ Peridinium*
NorthBar	06/25/97	94				
NorthBar	07/22/97	95				
NorthBar	08/26/97	90				
NorthBar	09/15/97	81				
NorthBar	05/20/98	19			15	11
NorthBar	06/18/98	13		34	26	
NorthBar	07/15/98	92				
NorthBar	08/12/98	79				
NorthBar	09/10/98	67	21			
NorthBar	05/19/99	93				

Round Lake

Round Lake was added to the study in the second year, so results are only available for the 1998 sampling season and May of 1999. Diatoms clearly dominated throughout the season (Figure 8-9). Constituting 62% of the community in May, diatom domination rose to almost 89% in July with a gradual tapering off through the rest of the season. In May, cryptomonads constituted 26% of the community, but the numbers decreased dramatically in June to 8% and stayed relatively low for the rest of the season (2-5%). Dinoflagellates also maintained a presence throughout the season, composing 2-6% of the phytoplankton community. Although blue-greens were present only in low numbers from May until August, they increased to 10% by September.

Overall phytoplankton counts in Round Lake increased from May to July and then slowly decreased for the remainder of the season (Figure 8-11). In May, the total phytoplankton count was 5.3 X 10^5 cells or colonies/liter. The peak count in July was 3.5 X 10^6 colonies or cells/liter. In May of 1999, the total count was slightly higher but similar: 8.9 X 10^5 cells or colonies/liter. A total of 30 different species were identified for the 1998 season.

Limnetic chlorophyll *a* concentration remained relatively constant for the entire 1998 season but for a sharp rise in July, followed by a sharper decrease in August (Figure 8-10). The peak concentration, in July, was 15 mg/l, and for the rest of the season, concentration ranged from 5-9 mg/l.

In May 1999, yellow-green algae dominated the phytoplankton, composing 71.5% of the community. 16.7% of the community was diatoms, and 7.9% of the community was cryptomonads. Overall phytoplankton counts (8.7 X 10^5 cells or colonies/liter) were similar to May of 1998 (5.3 X 10^5 cells or colonies/liter).

The slow increase of diatoms followed by a gradual decrease may indicate that spring turnover did not result in the abundance of nutrients often seen in lakes. Round Lake is connected to Crystal Lake, so some interaction between the lakes could affect the nutrient balance. The abundance of cryptomonads is unusual, but they may have been more successful in spring competition than the diatoms. Like other Sleeping Bear Dunes lakes, Round Lake had an abundance of *Cyclotella* sp., composing more than 10% of the community every month sampled (Table 8-3). *Synedra* sp. was also abundant, and there were periodic high counts of *Chlorochromonas* sp. and *Chroococcus* sp.

Figure 8-9. Relative abundance of algal groups (common names given for Phyla) collected from the limnetic zone of Round Lake in A) 1998; no samples were collected in 1997. Triplicate samples were collected from 1 meter below the surface at the deepest point of the lake.

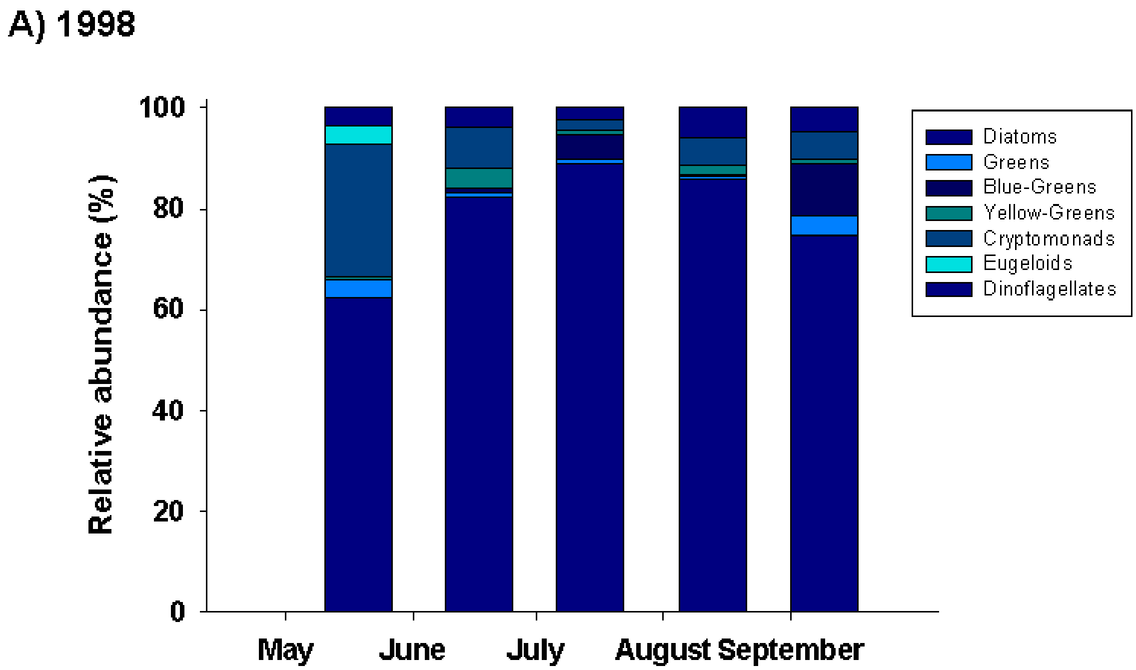


Figure 8-10. Chlorophyll *a* concentration in Round Lake. Data points represent samples taken from 1 meter below the surface at the deepest point of the lake. Chlorophyll a samples were only collected in 1998.

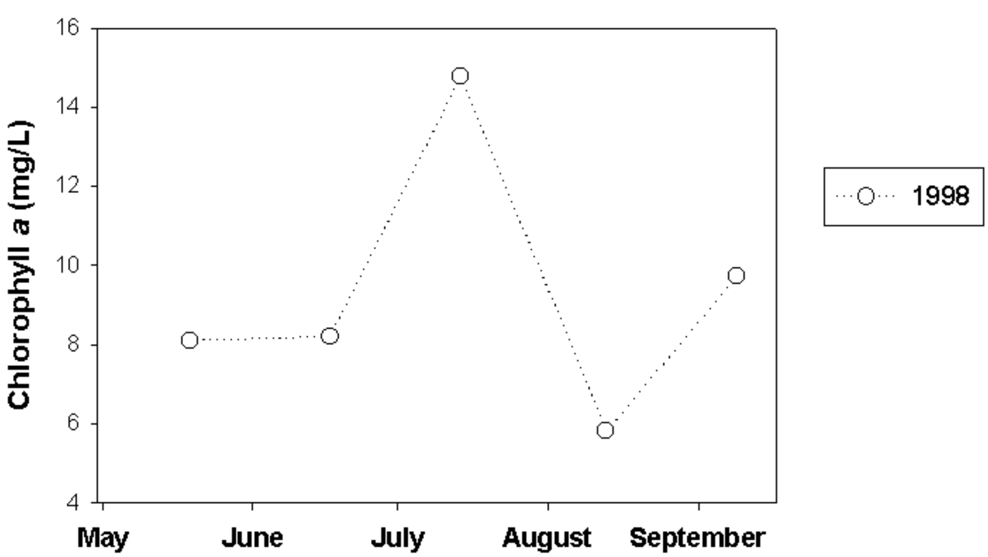


Figure 8-11. Total abundance of limnetic phytoplankton collected each month in Round Lake. Counts calculated from three composited samples taken from 1 meter below the surface at the deepest point in the lake.

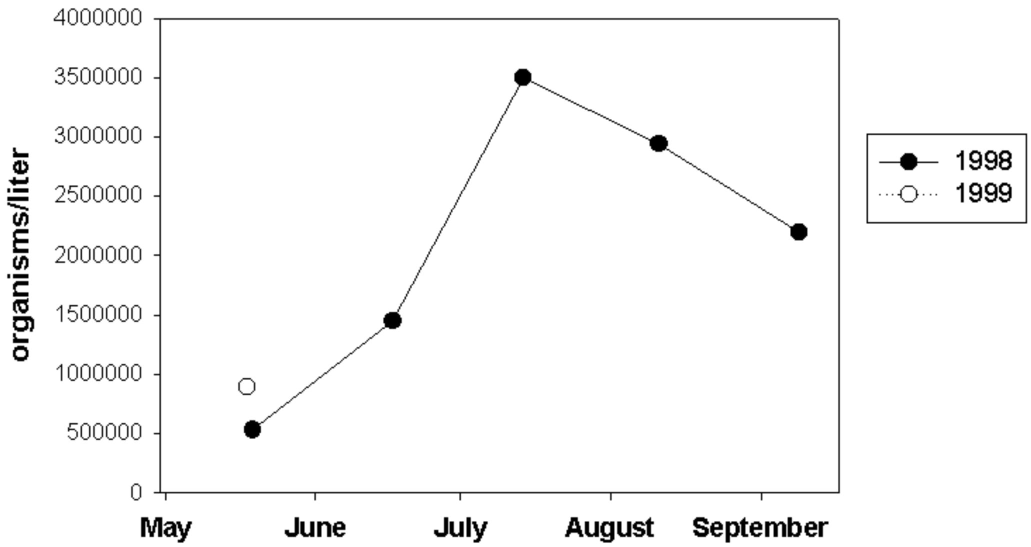


Table 8-3. Phytoplankton taxa that amounted to ≥ 10% of the relative abundance in limnetic zone of Round Lake with percent relative abundance.

		Cyclotella spp.	*Synedra* spp.	*Chlorochromonas* spp.	*Cryptomonas* spp.	*Chroococcus* spp.
Round	05/19/98	59			26	
Round	06/17/98	41	35			
Round	07/14/98	84				
Round	08/11/98	20	65			
Round	09/09/98	47	26			10
Round	05/18/99	14		68		

Beaver Lake

Samples were collected in Beaver Lake in the littoral and limnetic zones between July and September of 1997. In 1998, samples were collected in the limnetic zone between May and September. Additionally, limnetic samples were collected in May of 1999.

1997 Limnetic: In Beaver Lake, yellow-green algae dominated throughout most of the 1997 season (Figure 8-12). Only in July did diatoms composed a greater percentage of the phytoplankton community. In early July, the first time phytoplankton were collected, yellow-green algae made up 60% of the community with diatoms at 22%. By late July, those two groups had switched dominance: diatoms were 82% and yellow-greens were only 10%. For the remaining months of the season, yellow-greens dominated slightly, at 45% in August and 44% in September. Diatoms also remained relatively constant for those two months, composing 34% and 32% of the community. Green algae peaked at 12% of the community in September, but otherwise maintained low counts the rest of the season. Cryptomonads composed 8% of the phytoplankton community in early July and August with lower numbers in late July and September. Blue-greens were present as 8% of the community in early July, but were then absent in late July. Blue-greens returned in late July and September to make up 3% of the community.

Overall phytoplankton counts peaked in late July and then decreased (Figure 8-15). The early July phytoplankton count was 6.4×10^5 cells or colonies/liter, which quickly rose to 1.75×10^6 cells or colonies/liter in late July. Overall counts then dropped below the initial early July count to 4.3×10^5 and 5.3×10^5 cells or colonies/liter. A total of 40 taxa were identified from the 1997 Beaver Lake samples.

1997 Littoral: As in the limnetic zone, yellow-green algae dominated the phytoplankton in early July (Figure 8-13). Diatoms dominated in late July, composing 74% of the community. By August, yellow-green algae were again dominant, composing 57% of the community. In September, yellow-green algae were overtaken in abundance by the green algae, which composed 46% of the community. This was due in large part to a surge in numbers of *Actinastrum* sp. and *Sphaerocystis* sp. Dominant individual species throughout the season included *Cyclotella* sp., *Synedra* sp., *Chlorochromonas* sp., and *Dinobryon* sp.

In 1998, patterns of abundance among algal groups were strikingly similar to the 1997 results. A sample was collected in May of 1998, so early season activity could be compared to the rest of the season. Early season community composition was very different from any of the 1998 results. Blue-greens were almost equal to the dominant diatoms in May, composing 30% of the community to the diatoms 33%. The community changed dramatically by July, where community patterns were similar to July of 1997: yellow-green algae dominated (69% of the community) and diatoms and green algae were next in highest abundance (8%). Diatoms again surged in July, to constitute 72% of the community, and yellow-green numbers dropped to 17% of the community. Diatoms and yellow-greens remained relatively constant for the rest of the season but for a decrease in diatoms in September coupled with an increase in green algae. Blue-green numbers dropped sharply after May and remained at 1-2% of the community for the rest of the season. Crytptomonads and dinoflagellates remained relatively constant through the sampling season at approximately 4% and 2% of the community.

In 1998, overall phytoplankton counts peaked in July at 1.1×10^6 cells or colonies/liter. Early season was characterized by low counts (2.3×10^5 in May and 2.9×10^5 in June), and after the July peak, numbers dropped off only slightly (1.0 X 10^6 in August and 7.6 X 10^5 in September). A total of 42 taxa were identified from 1998 samples.

Patterns of limnetic chlorophyll *a* were different between years (figure 8-14). In 1997, concentrations ranged from 1-6 mg/l with peaks in early July and September. In 1998, the range and concentrations were higher. Peak chlorophyll *a* concentration in August was 12 mg/l, and the low, in June, was 2 mg/l.

In May of 1999, green algae dominated the phytoplankton community (75%), as was seen in June of 1998 and early July of 1999. Yellow-greens were also relatively abundant, comprising 12.7% of the community. Diatoms were a far third in abundance, comprising only 4% of the phytoplankton community. The overall count of organisms (2.8 X 10^5 cells or colonies/liter) was almost identical to May of 1998 (2.3 X 10^5 cells or colonies/liter).

The most abundant species was clearly *Chlorochromonas* sp., which composed greater than 10% of the community during most months sampled (Table 8-4). Other abundant species included *Cyclotella* spp., *Synedra* spp. and *Dinobryon* spp. In a study conducted by Limnetics, Inc. in 1970, the dominant species of phytoplankton in Beaver Lake was *Chrococcus limneticus*. Other frequently observed species included *Melosira islandica*, *Tabellaria fenestrata*, and *Pediastrum duplex* (Limnetics 1970). All of these genera were found in the lakes during 1997-1999 sampling seasons.

According to community composition patterns in Beaver Lake, it can be classified as an oligotrophic lake. Although the chlorophyll *a* concentrations are slightly high in 1998, other evidence indicates its low nutrient characteristic. The dominance of yellow-green algae through most of the 1997 season and the their clear presence in 1998 show that diatoms were unable to dominate the system through the summer season. Beaver Lake has been recognized as a polymictic lake, and as such, nutrients from the deeper waters are circulated periodically throughout the season. Because mixing takes place relatively often, the concentration of re-introduced nutrients is not as high as it would be if mixing only took place once a year. It is quite possible that mixing took place in July during both years and that diatoms were able to out-compete the yellow-green algae with the increased availability of nutrients.

Nutrient concentration in 1998 appeared to be higher overall, as is apparent in the increased biomass. The higher percentage of the community dominated by diatoms also gives an indication of the higher nutrient availability. In order for the diatom population to increase, there must be more nitrogen and phosphorus available, as for all phytoplankton, and also more silica. According to water chemistry results, silica was higher in 1997 than in 1998. Total phosphorus was also generally higher in 1998.

Figure 8-12. Relative abundance of algal groups (common names given for Phyla) collected from the limnetic zone of Beaver Lake in A) 1997 and B) 1998. Triplicate samples were collected from 1 meter below the surface at the deepest point of the lake.

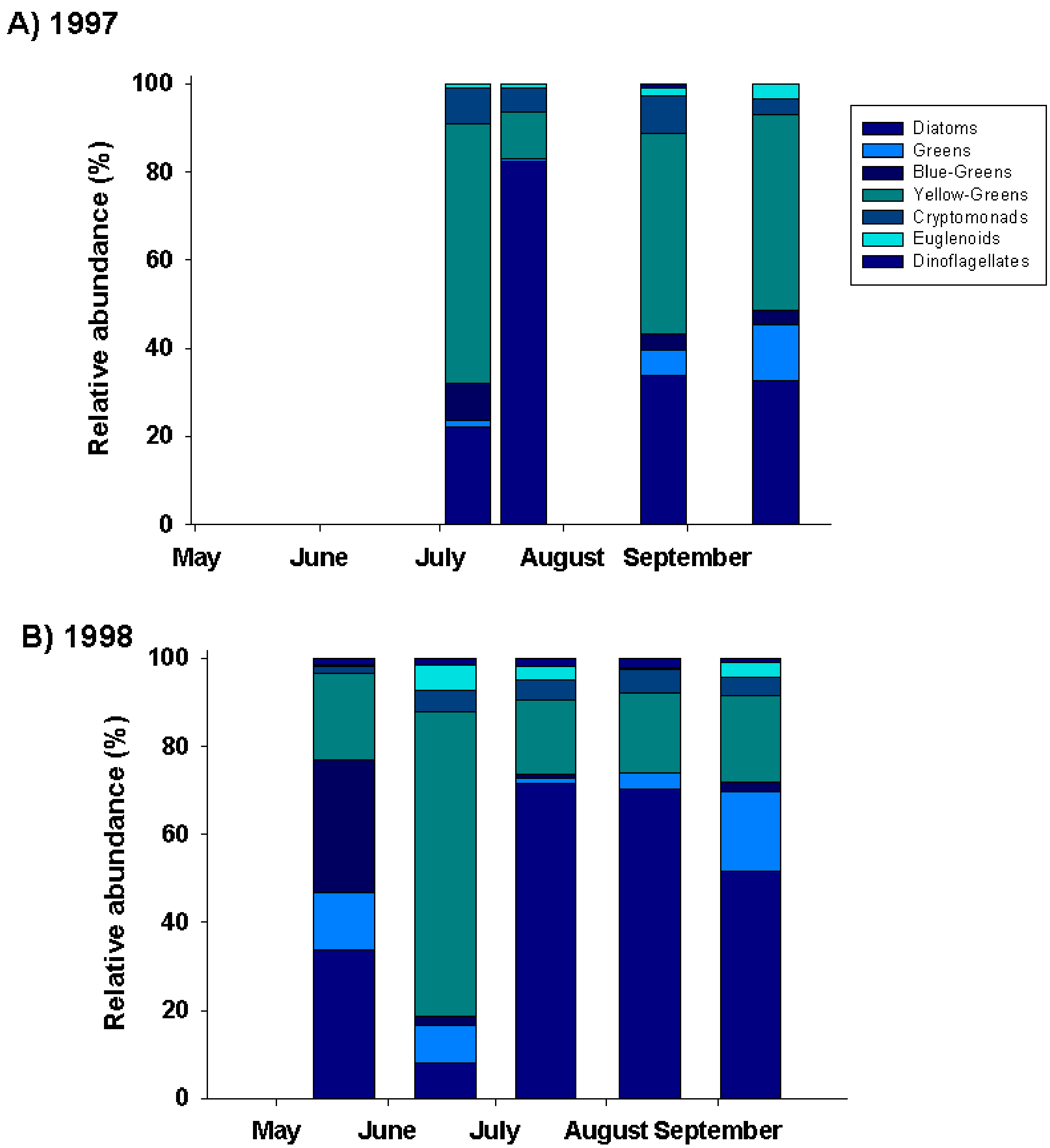


Figure 8-13. Relative abundance of algal groups (common names given for Phyla) collected in the littoral zone of Beaver Lake in 1997. Triplicate samples were collected from 1 meter below the lake surface.

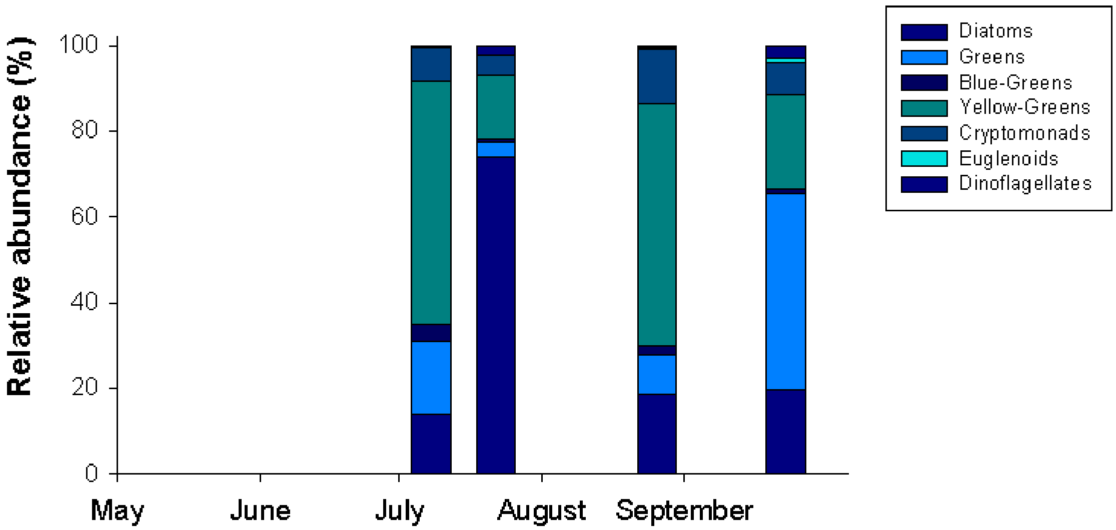


Figure 8-14. Chlorophyll *a* concentration in Beaver Lake. Data points represent samples taken from 1 meter below the surface at the deepest point in the lake.

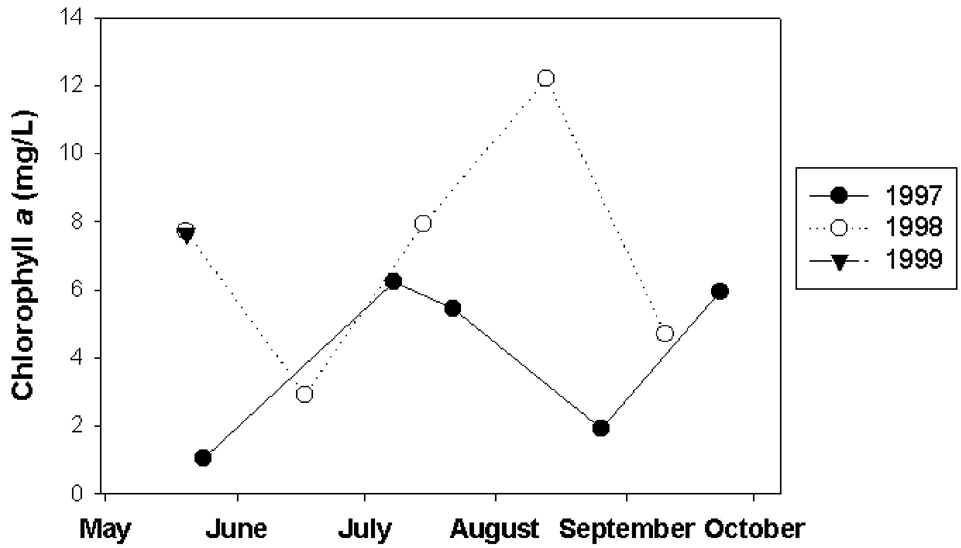


Figure 8-15. Total abundance of limnetic phytoplankton collected each month in Beaver Lake. Counts calculated from three composited samples taken from 1 meter below the surface at the deepest point in the lake.

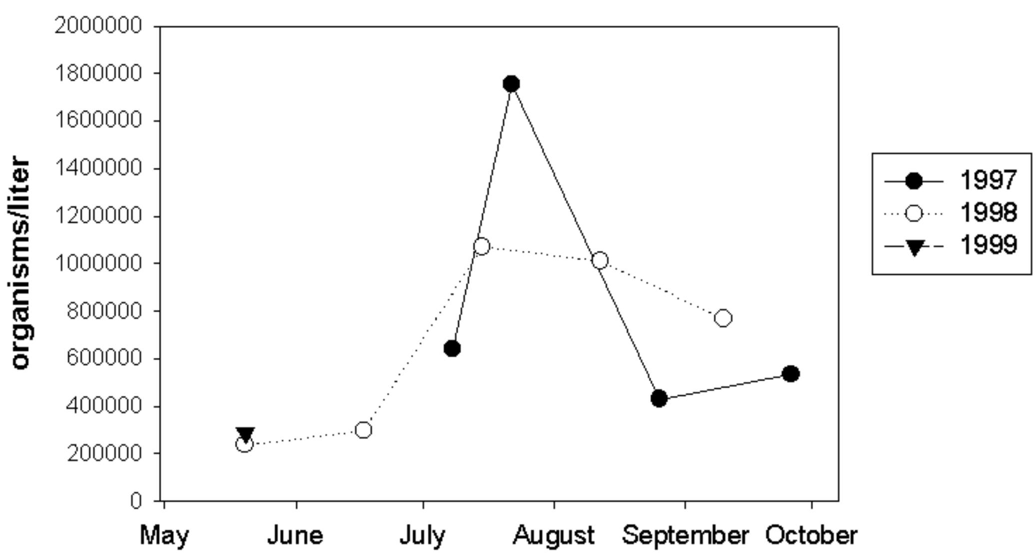


Table 8-4. Phytoplankton taxa that amounted to ≥ 10% of the relative abundance in limnetic zone of Beaver Lake with percent relative abundance.

		Cyclotella spp.	*Synedra* spp.	*Actinastrum* spp.	*Ankistrodesmus/ Quadrigula*	*Sphaerocystis schroeteri*	*Chlorochro-monas* spp.	*Dinobryon* spp.	*Anabaena* spp.
Beaver	7/8/97						39	20	
Beaver	7/22/97	29	47						
Beaver	8/26/97						26	19	
Beaver	9/26/97						43		
Beaver	5/20/98		17	11			18		30
Beaver	6/17/98						68		
Beaver	7/15/98	33	28						
Beaver	8/12/98	58					18		
Beaver	9/10/98		20				18		
Beaver	5/20/99				30	44	13		

Grand Sable Lake

Grand Sable Lake was sampled from July through September 1997 in the littoral and limnetic zones. In 1998, samples were collected in the limnetic zone from May to September. In 1999, limnetic samples were collected in May.

1997 Limnetic: In 1997, Grand Sable Lake was sampled for phytoplankton starting in early July. The community was clearly dominated by yellow-green algae (60%) in early July, and remained the dominant group through the season (Figure 8-16). Blue-greens started the season at 8% of the community and increased to 30% in August—when yellow-green algae composed 33% of the community. Numbers of blue-greens then decreased slightly in September. Diatoms were present through the season but were never the dominant group. The percent of the community composed of diatoms increased slowly from 15% in early July to 23% in September. Green algae and cryptomonads composed 2-9% of the community throughout the season.

Overall counts of phytoplankton in 1997 increased from May to August, with a slight decrease in September (Figure 8-19). There were never any sharp peaks in abundance. Counts ranged from 5.4×10^5 cells or colonies/liter in May to 9.3×10^5 cells or colonies/liter in August. A total of 40 taxa were identified in 1997.

1997 Littoral: The phytoplankton community in the littoral zone was similar to that in the limnetic zone (Figure 8-17). Yellow-green algae were dominant most months, composing between 29% and 62% of the community relative abundance. Blue-greens were present in all samples collected, and increased to be the most dominant group by September, composing 33% of the community. Diatoms generally increased throughout the sampling season to compose 23% of the phytoplankton community. Dominant species in the littoral zone included *Rhizoselenia* sp., *Chlorochromonas* sp. and *Chroococcus* sp.

The phytoplankton community in Grand Sable Lake in 1998 was very similar to the community in 1997 (Figure 8-16). In May, there was some deviation from the later summer pattern—diatoms constituted 81% of the community, and yellow-green algae only made up 14% of the community. This relationship quickly changed in June when yellow-greens increased to make up 63% of the community while diatom percent community composition decreased to 14%. Blue-greens again became prominent later in the season as they increased from 1% of the community in May to a peak of 41% of the community in August, when they were the dominant algal group. Cryptomonads increased to 9% of the community in July and then slowly decreased. Greens remained constant through the season at 3-4%.

Phytoplankton counts were actually highest in May, and numbers decreased through the rest of the season. Counts in May were 1.4×10^6 cells or colonies/liter and decreased to 5×10^5 cells or colonies/liter. In all, 50 taxa were identified in 1998.

Limnetic chlorophyll *a* concentrations were highly variable in both 1997 and 1998 (Figure 8-18). Concentrations in 1997 ranged from 8-13 mg/l, with a rise and fall pattern. In 1998, the range of concentrations was 4-11 mg/l, but the pattern was again rising and falling concentrations each month. The high concentrations are unusual given that Grand Sable has characteristics of an oligotrophic lake, but similar concentrations have been recorded in previous studies (Kamke, 1987).

Diatoms dominated the phytoplankton community in May of 1999 (40%), but there was a diverse community of algae present in abundance. Yellow-green algae composed 29% of the community, and blue-greens composed 15% of the community. 7% of the community was green algae, 6% was cryptomonads, and 2% was dinoflagellates. Overall, there were 3.7 X 10^5 cells or colonies/liter in May. This count was considerably lower than the May 1998 overall abundance of 1.4 X 10^6 cells or colonies/liter.

As in Beaver Lake, *Chlorochromonas* spp. was abundant in most months sampled (Table 8-5). *Chroococcus* spp., *Cyclotella* spp., and *Dinobryon* spp. were also abundant periodically in Grand Sable Lake. In a 1970 study of Grand Sable Lake, abundant phytoplankton included *Asterionella formosa*, *Fragilaria crotenensis*, *Melosira islandica*, *Tabellaria fenestrata*, and *Aphanocapsa pulchra* (Limnetics 1970). All of these genera were found in Grand Sable Lake except for *Aphanocapsa pulchra*.

The seasonal patterns in Grand Sable Lake are typical of an oligotrophic northern lake. In these lakes, spring mixing results in a diatom bloom—as seen in May of 1998. Due to nutrient competition and predation, the diatoms then decrease in number as yellow-green algae become dominant—June of 1997 and 1998. The community then becomes dominated by blue-greens—August of 1997 and 1998—until fall mixing. If samples had been taken in Grand Sable Lake in May of 1997, it is extremely likely that diatoms would have dominated the community.

Figure 8-16. Relative abundance of algal groups (common names given for Phyla) collected from the limnetic zone of Grand Sable Lake in A) 1997 and B) 1998. Triplicate samples were collected from 1 meter below the surface above the deepest part of the lake.

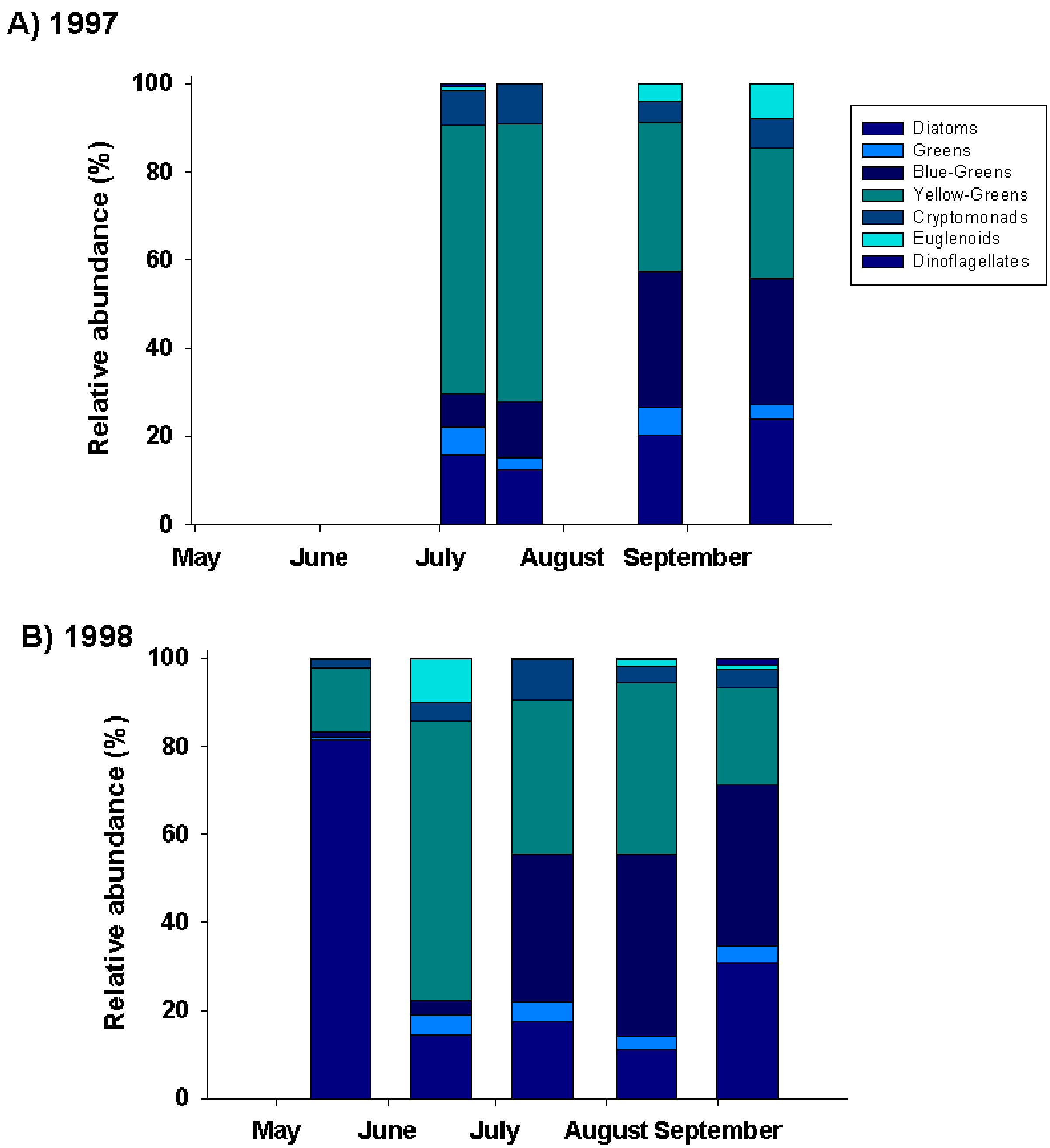


Figure 8-17. Relative abundance of algal groups (common names given for Phyla) collected from the littoral zone of Grand Sable Lake in 1997. Triplicate samples were collected from 1 meter below the surface.

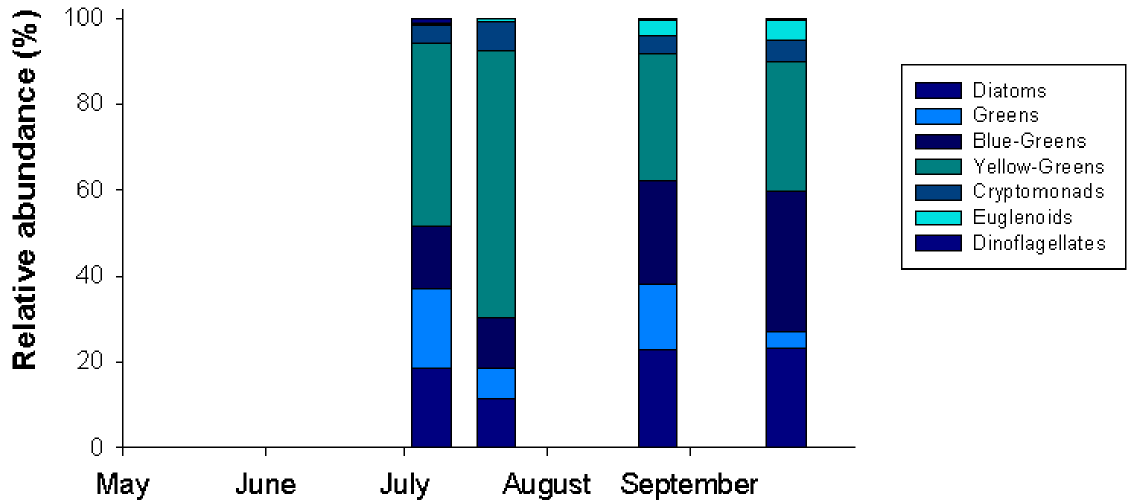


Figure 8-18. Chlorophyll *a* concentration in Grand Sable Lake. Data points represent samples taken from 1 meter below the surface at the deepest point in the lake.

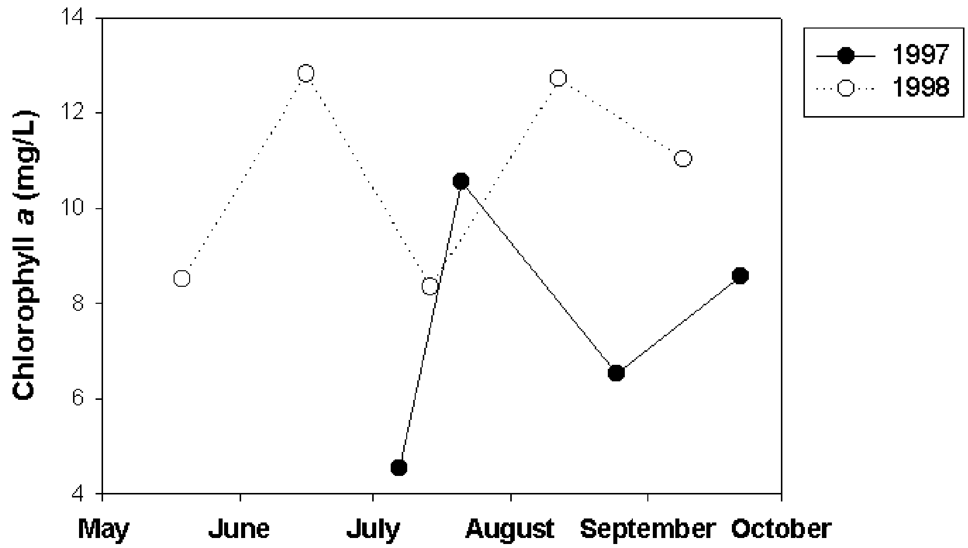


Figure 8-19. Total abundance of limnetic phytoplankton collected each month in Grand Sable Lake. Counts calculated from three composited samples taken from 1 meter below the surface at the deepest point in the lake.

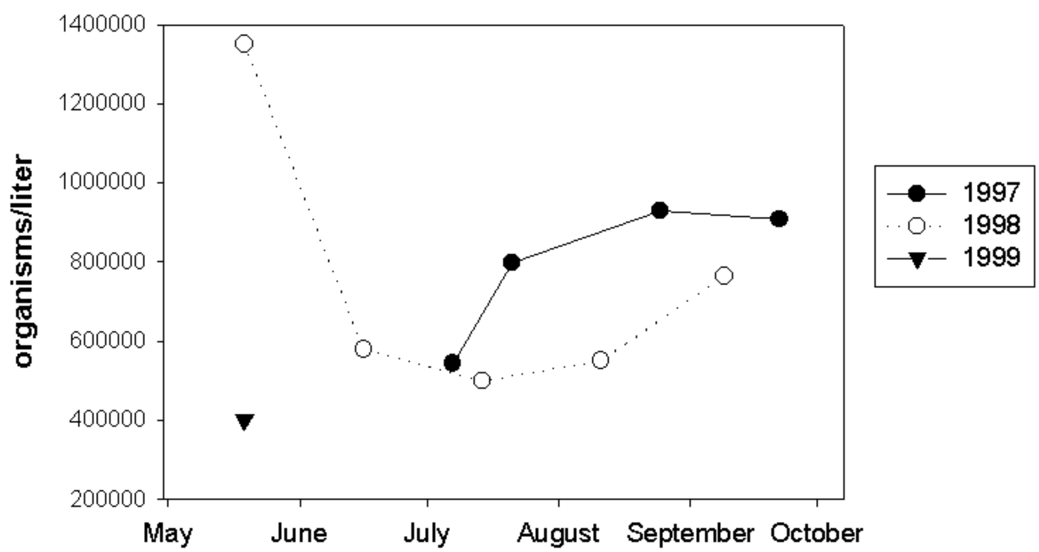


Table 8-5. Phytoplankton taxa that amounted to ≥ 10% of the relative abundance in limnetic zone of Grand Sable Lake with percent relative abundance.

		Cyclotella spp.	*Rhizosolenia eriensis*	*Synedra* spp.	*Chlorochro-monas* spp.	*Dinobryon* spp.	*Aphanizomenon flos-aquae*	*Chroococcus* spp.	*Oscillatoria* spp.
GrndSabl	7/7/97				60				
GrndSabl	7/21/97				52	11			
GrndSabl	8/25/97				32			26	
GrndSabl	9/22/97		13		27			26	
GrndSabl	5/19/98	50		13		14			
GrndSabl	6/16/98				62				
GrndSabl	7/14/98				34		11	14	
GrndSabl	8/11/98				36			34	
GrndSabl	9/9/98	22			21				
GrndSabl	5/19/99	19			13	14			12

Sargent Lake

Phytoplankton were collected in Sargent Lake between July and September of 1997 in the littoral and limnetic zones. In 1998, phytoplankton were collected in the limnetic zone between May and September.

1997 Limnetic: Phytoplankton communities in Sargent Lake were varied with few obviously dominant groups in 1997 (Figure 8-20). In June, yellow-green algae composed 33% of the community and diatoms 32%. Green algae were also relatively abundant, making up 23% of the community. This equality among groups shifted in July to dominance by greens (36%) followed by diatoms (29%). Yellow-green algae numbers had decreased to compose 20% of the community. Green algae maintained their slight dominance through the rest of the season, increasing to 39% of the community in August and then decreasing to 34% in September. Simultaneously, yellow-green algae increased and diatoms decreased slightly. Blue-greens increased in August but never dominated the system.

Overall phytoplankton counts were also relatively stable through the season (Figure 8-23). Phytoplankton counts were highest in September at 8.2×10^5 cells or colonies/liter, but the lowest count was 5.9×10^5 cells or colonies/liter in July—not much difference. A total of 55 taxa were identified from Sargent Lake in 1997.

1997 Littoral: There were similarities between the littoral and limnetic phytoplankton communities, although the littoral zone tended to include more diatoms (Figure 8-21). Diatoms accounted for 31% of the phytoplankton community in early July and quickly increased to 48% of the community by late July. Green algae had similar relative compositions, accounting for 26% to 31% of the community throughout the 1997 season. Yellow-green algae were relatively abundant, with a high of 20% of the community in early July. Blue-greens were also present in appreciable amounts throughout the season, with a high of 16% in early July that fell to 4% in August. Phytoplankton species that were abundant included *Cyclotella* sp., *Sphaerocystis* sp., *Chlorochromonas* sp., and *Synedra* sp. (in August).

In 1998, there were more changes in dominance and population size among the phytoplankton. In May, yellow-green algae were clearly dominant in Sargent Lake, constituting 62% of the community. Diatoms were a strong presence, making up 20% of the community. Other groups were present only in low concentrations. In June, diatoms increased slightly, and green algae increased appreciably as yellow-greens began to decrease. The community was 39% yellow-green algae, 26% diatoms, and 24% green algae. This trend continued in July. In August, however, green algae and blue-green numbers decreased as diatoms continued to increase along with yellow-greens. By September, the final month of sampling, diatoms were clearly the dominant group and composed 51% of the community. Yellow-greens had increased to 13% of the community from the low of 7% in July. Green algae and blue-greens continued to decrease.

Phytoplankton counts overall decreased from May to August with a slight increase in September (Figure 8-23). The highest count of organisms was 1.1×10^6 cells or colonies/liter, and the lowest count, in August, was 5.5×10^5 cells or colonies/liter. 57 taxa were identified in the 1998 Sargent Lake samples.

Limnetic chlorophyll *a* samples were lower in 1997 than in 1998, as were the overall phytoplankton counts (Figure 8-22). Concentrations in 1997 ranged from 0.4-7 mg/l. In 1998, concentrations ranged from 6-7.5 mg/l, but sampling was limited to only three months: May, August, and September. In June and July, field technicians filtered the samples incorrectly, so there were no results. In 1997, chlorophyll *a* decreased from June to July and then remained relatively stable before peaking in September. Patterns are difficult to distinguish in 1998 because of the limited number of samples collected. Samples collected in 1999 were not transported appropriately, so the filters were thawed and degraded upon arrival at the laboratory; therefore, they could not be analyzed adequately.

Yellow-green algae dominated the phytoplankton in May of 1999, composing 38.6% of the community. Also abundant in the May samples were diatoms, which composed 31.8% of the community. Euglenoids actually made up a portion of the community (13%), and greens were also relatively abundant (9%). The overall count of phytoplankton (8.6 X 10^5 cells or colonies/liter) was lower than in May of 1998 (1.1 X 10^6 cells or colonies/liter).

Dominant phytoplankton species included *Chlorochromonas* spp. and *Sphaerocystis schroeteri* throughout the sampling months (Table 8-6). Other species rarely composed greater than 10% of the community.

The minor and varied shifts in dominance and percent community composition in 1997 are evidence of a stable system not subject to much nutrient addition during spring mixing or throughout the season. Community dominance switched among groups, and no phytoplankton group showed any large growth peaks through the season.

Community structure in 1998 went through more noticeable changes in both group dominance and overall counts. In 1998 diatoms increased throughout the season. Yellow-greens started out as the dominant group in May but were out-competed by diatoms, green algae, and in a minor sense, blue-greens. Between July and August, that trend reversed itself slightly and lasted through the season as yellow-green algae increased again at the expense of the green and blue-greens and the ever-increasing diatoms. Green algae were able to compete with yellow-green algae in June and July, but by August, diatoms and yellow-greens together limited green phytoplankton growth. Unlike other lake systems, where diatoms bloomed early in the season and then were out-competed for the rest of the season, diatoms increased from May to September. Spring mixing may not have provided the nutrients necessary for a spring diatom bloom because the lake is clearly oligotrophic.

Figure 8-20. Relative abundance of algal groups (common names given for Phyla) collected from the limnetic zone of Sargent Lake in A) 1997 and B) 1998. Triplicate samples were collected from 1 meter below the surface above the deepest part of the lake.

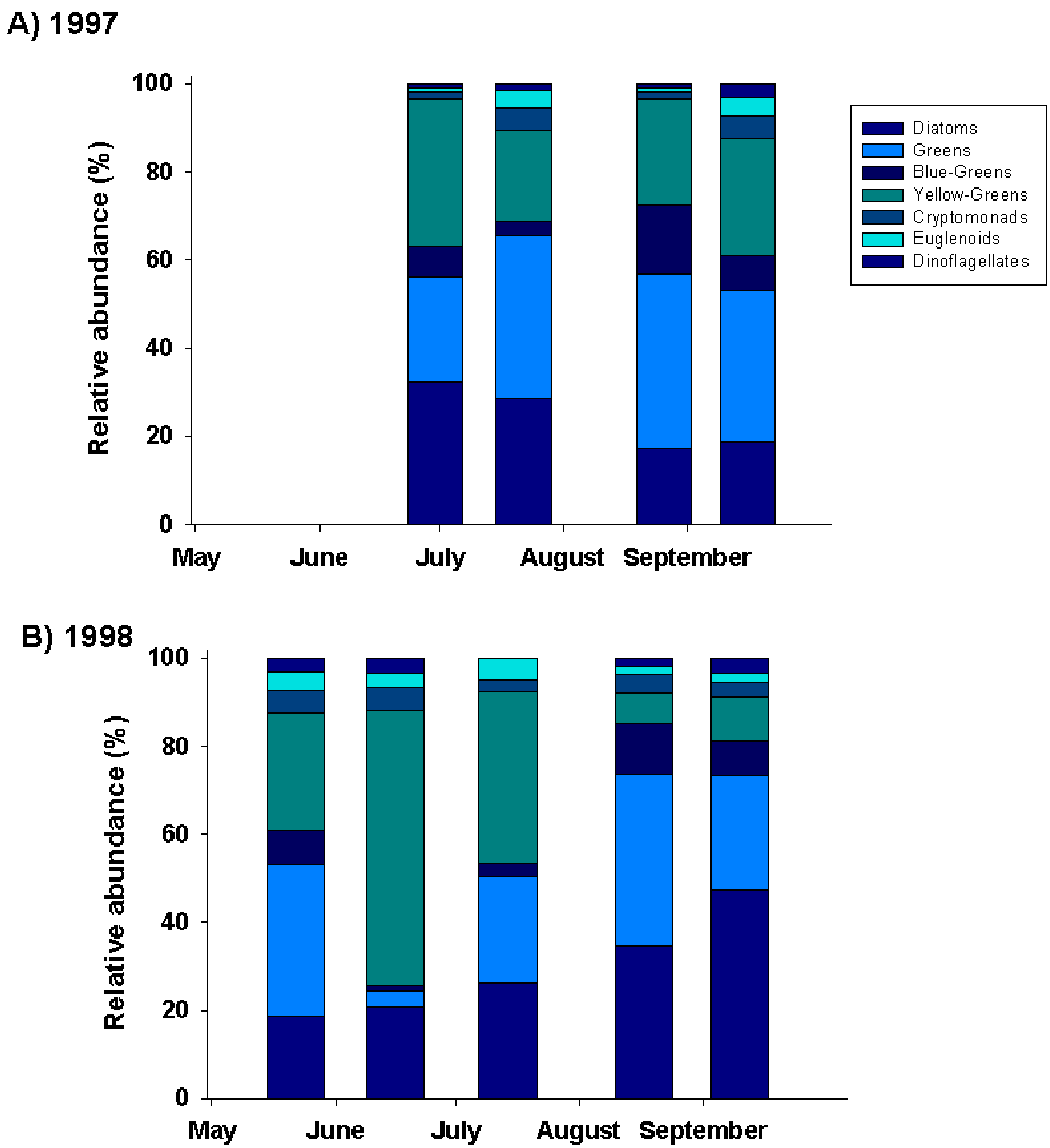


Figure 8-21. Relative abundance of algal groups (common names given for Phyla) collected from the littoral zone of Sargent Lake in 1997. Triplicate samples were collected from 1 meter below the surface.

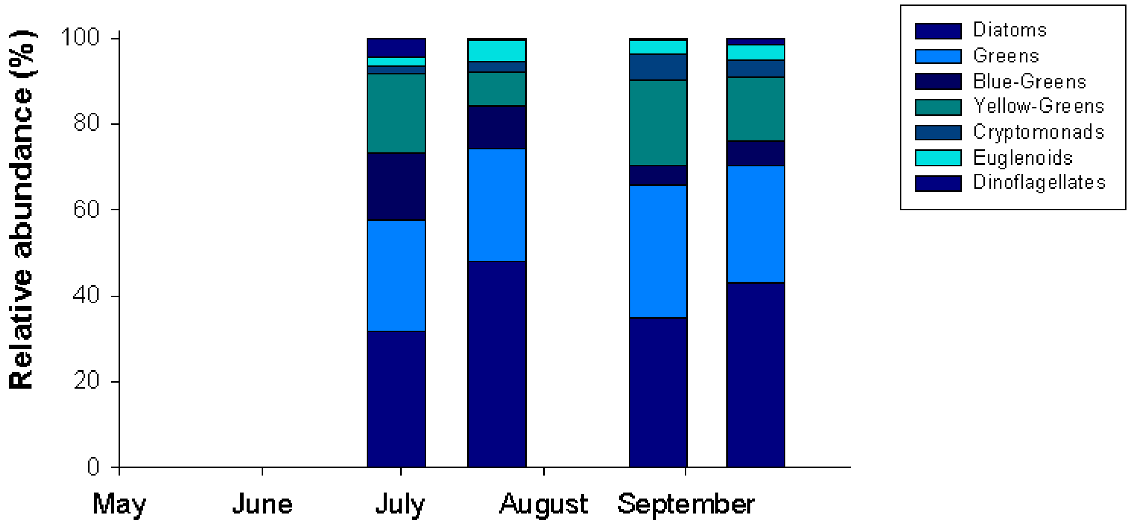


Figure 8-22. Chlorophyll *a* concentration in Sargent Lake. Data points represent samples taken from 1 meter below the surface at the deepest point in the lake.

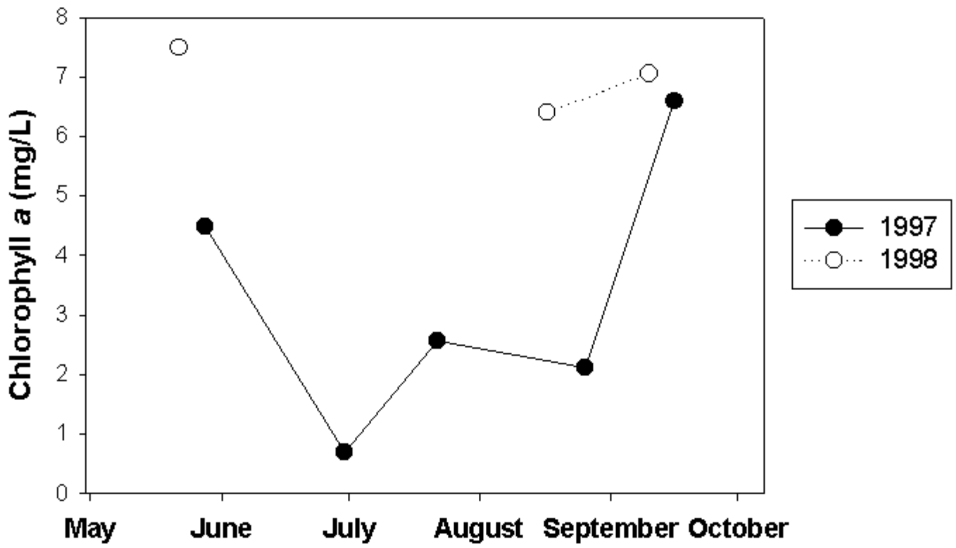


Figure 8-23. Total abundance of limnetic phytoplankton collected each month in Sargent Lake. Counts calculated from three composited samples taken from 1 meter below the surface at the deepest point in the lake.

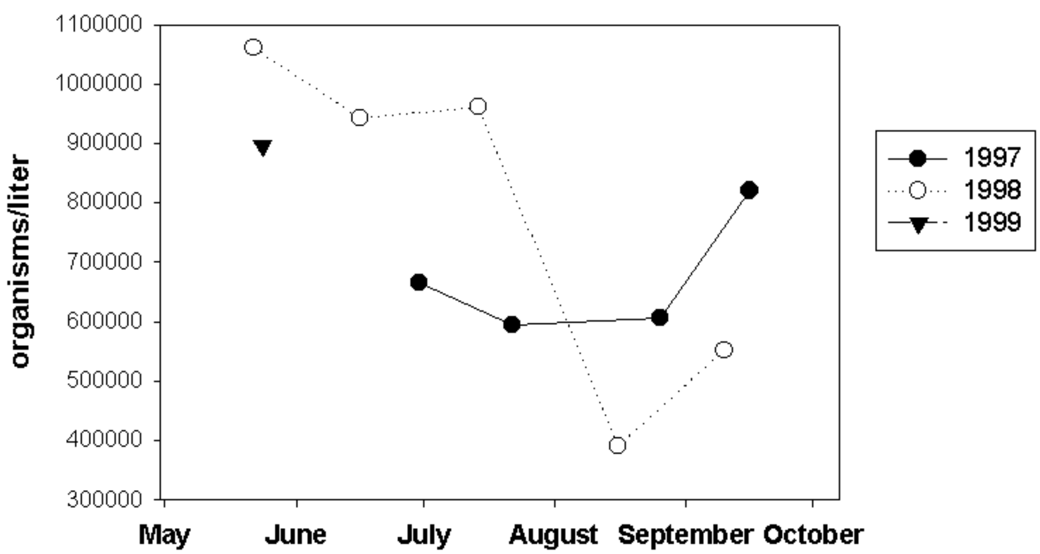


Table 8-6. Phytoplankton taxa that amounted to ≥ 10% of the relative abundance in limnetic zone of Sargent Lake with percent relative abundance.

		Cyclotella comta	*Cyclotella* spp.	*Fragilaria crotonensis*	*Synedra* spp.	*Oocystis* spp.	*Sphaerocystis schroeteri*
Sargent	6/30/97	11					10
Sargent	7/22/97						17
Sargent	8/26/97					11	20
Sargent	9/16/97						21
Sargent	5/22/98						
Sargent	6/16/98						13
Sargent	7/14/98						16
Sargent	8/16/98			16			19
Sargent	9/10/98		12	14			
Sargent	5/24/99				14		

		Chlorochromonas spp.	*Dinobryon bavaricum*	*Dinobryon* spp.	*Trachelomonas* spp.
Sargent	6/30/97	32			
Sargent	7/22/97	19			
Sargent	8/26/97	24			
Sargent	9/16/97	26			
Sargent	5/22/98		32	25	
Sargent	6/16/98	39			
Sargent	7/14/98				
Sargent	8/16/98				
Sargent	9/10/98	12			
Sargent	5/24/99	35			13

Siskiwit Lake

Phytoplankton were collected from Siskiwit Lake from June through September, 1997 in the littoral and limnetic zones and from May to September 1998 in the limnetic zone. Additionally, a phytoplankton sample was collected in May of 1999 in the limnetic zone.

1997 Limnetic: In 1997, diatoms dominated Siskiwit Lake. In the June samples, 89% of the phytoplankton was diatoms (Figure 8-24). That percentage decreased for a short time in July to 60%, but the counts rebounded in August and September to 75% and 80%. The decrease in July was coupled with an increase in yellow-green algae (26% of the community). The yellow-green algae subsequently decreased as the diatoms again dominated the community in August and September. At no time in the season did the green algae, blue-greens, or cryptomonads constitute more than 6% of the phytoplankton community.

Phytoplankton counts overall peaked in August at 1.5 X 10^6 cells or colonies/liter. Counts were lowest in May (6 X 10^5 cells or colonies/liter), increased through August, and decreased slightly in September (Figure 8-27). A total of 40 different taxa were identified in 1997.

1997 Littoral: Littoral zooplankton had a lower percentage of diatoms in June of 1997, but otherwise, a similar pattern was apparent between littoral and limnetic phytoplankton (Figure 8-25). In June of 1997, diatoms comprised 54% of the littoral phytoplankton in June and steadily increased throughout the summer, whereas limnetic phytoplankton in June composed 87% of the phytoplankton then decreased in July and slowly increased in August and September. Green algae were abundant in June (40% of the community), but numbers decreased significantly for the rest of the sampling season (5-10%). Although yellow-green algae only composed 2% of the community in June, the group increased to 20% of the community in July, and remained above 10% for the rest of the season. Abundant species included *Cyclotella* sp. and *Dinobryon* sp.

Diatoms were again the dominant algal group in 1998 (Figure 8-24). In May of 1998, diatoms composed 72% of the community. After a slight decrease in June, diatoms increased through the rest of the season to a high community percentage of 86% in September. Yellow-green algae were the only other phytoplankton group to constitute a noticeable percentage of the community. The highest percentage of yellow-green algae in the phytoplankton community was in May (20%). Numbers decreased for the rest of the season to 6% in September, coinciding with the gradual increase in diatoms. Green algae, blue-greens, and cryptomonads never made up more than 4% of the phytoplankton community.

Interestingly, phytoplankton counts peaked in June at 1.1 X 10^6 cells or colonies/liter and then decreased to 7.3 X 10^5 cells or colonies/liter until a slight reversal in September. In 1998, 49 different taxa were identified from Siskiwit Lake.

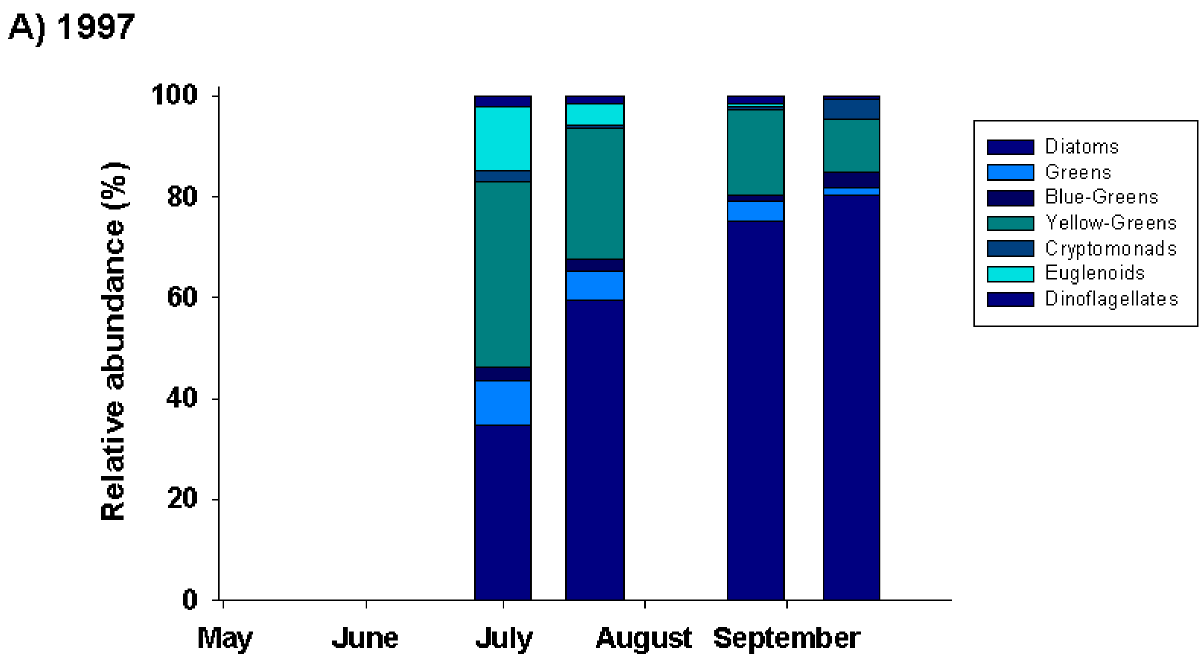
Limnetic chlorophyll *a* was generally higher in 1997 (Figure 8-26). There was very little variation between months in 1997, and chlorophyll *a* ranged from 1.5-3 mg/l. In 1998, chlorophyll results were only available for May, August, and September, so patterns were not discernable. In June and July, field technicians filtered the samples incorrectly, so there were no results. A high concentration in May (7.7 mg/l) was followed by very low concentrations in August and September (0.5 mg/l and 1.2 mg/l). Samples collected in 1999 were not transported appropriately, so the filters were thawed and degraded upon arrival at the laboratory; therefore, they could not be analyzed adequately.

In May of 1999, diatoms (70%) dominated Siskiwit Lake with yellow-green algae present in next highest abundance (17%). Euglenoids were present to compose 7% of the community, and 4% of the community was green algae. Total number of organisms was 7.7 X 10^5 cells or colonies/liter, which was higher than counts in May of 1998 but still well within the range of counts collected from Siskiwit Lake over the course of the study.

Between years, phytoplankton counts were similar, but counts were slightly higher in 1997. Peak counts were in different months each year (August in 1997 and June in 1998). The dominant species was *Cyclotella* spp., which composed greater than 40% of the community in every month sampled (Table 8-7). Other than *Cyclotella* spp., only *Dinobryon* spp. and *Asterionella formosa* (individual) ever composed greater than 10% of the phytoplankton community, and only on rare occasions.

Siskiwit Lake is clearly dominated by diatoms with yellow-green algae in a secondary position. In June of 1997, yellow-green algae increased at the expense of diatoms, but diatom counts rebounded the following month. Aside from that occurrence, diatoms remained the dominant algal group throughout both seasons. Siskiwit Lake appears to have a ready supply of nutrients that support constant algal growth through the season. All three of the macronutrients are available in constant, if low, supply from May to September. Siskiwit Lake is clearly an oligotrophic northern lake. Although diatoms dominate the lake throughout the season, the lake clearly has low nutrients.

Figure 8-24. Relative abundance of algal groups (common names given for Phyla) collected from the limnetic zone of Siskiwit Lake in A) 1997 and B) 1998. Triplicate samples were collected from 1 meter below the surface above the deepest part of the lake.



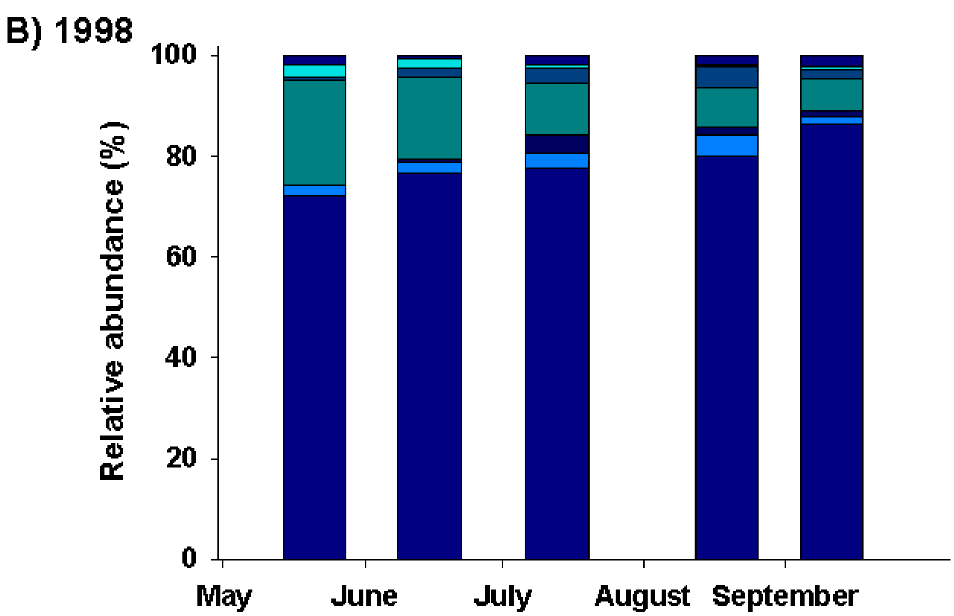


Figure 8-25. Relative abundance of algal groups (common names given for Phyla) collected from the littoral zone of Siskiwit Lake in 1997. Triplicate samples were collected from 1 meter below the surface.

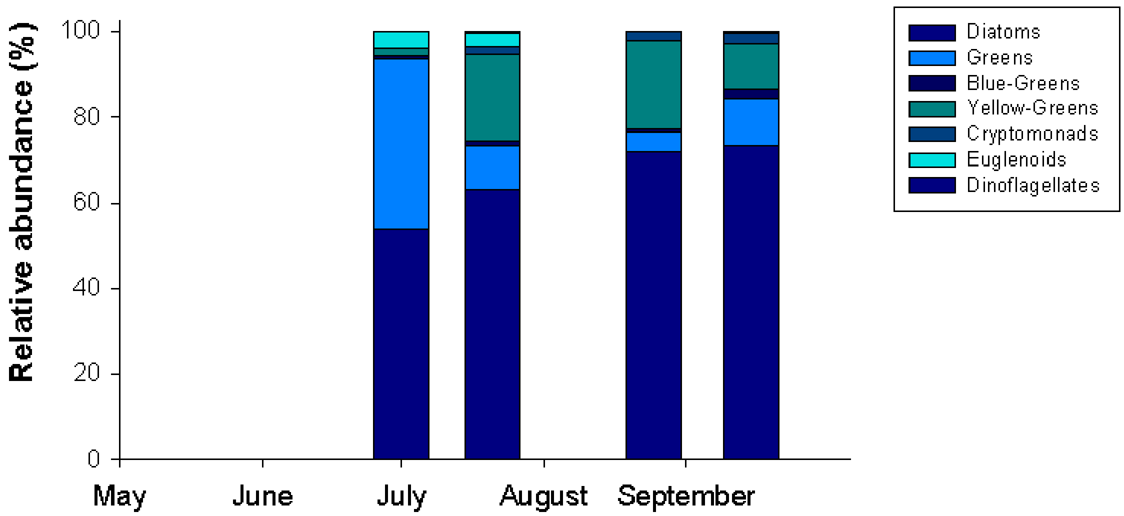


Figure 8-26. Chlorophyll *a* concentration in Siskiwit Lake. Data points represent samples taken from 1 meter below the surface at the deepest point in the lake.

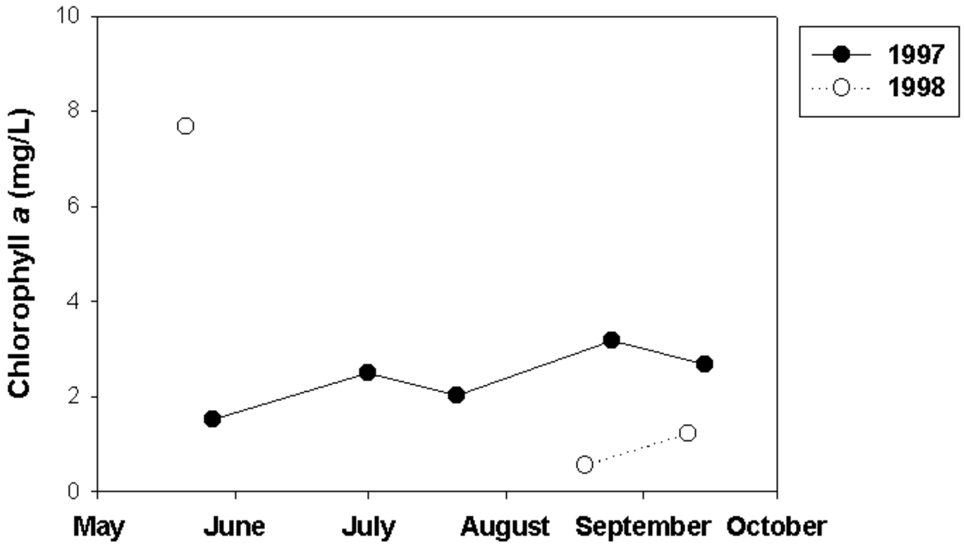


Figure 8-27. Total abundance of limnetic phytoplankton collected each month in Siskiwit Lake. Counts calculated from three composited samples taken from 1 meter below the surface at the deepest point in the lake.

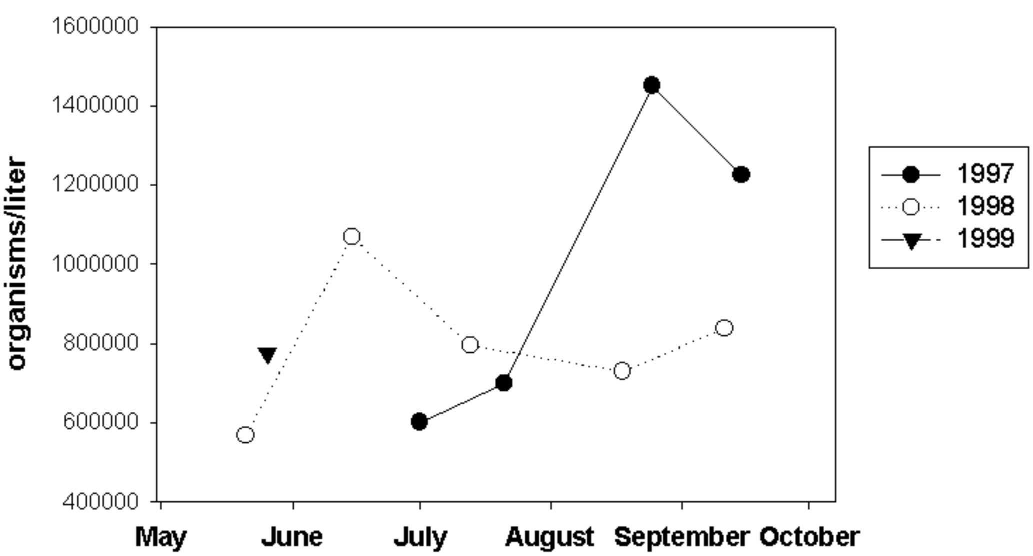


Table 8-7. Phytoplankton taxa that amounted to ≥ 10% of the relative abundance in limnetic zone of Siskiwit Lake with percent relative abundance.

		Asterionella formosa (indiv)	*Cyclotella* spp.	*Dinobryon* spp.
Siskiwit	07/01/97		71	
Siskiwit	07/21/97		52	
Siskiwit	08/25/97		71	11
Siskiwit	09/15/97		78	
Siskiwit	05/21/98		46	
Siskiwit	06/15/98		64	11
Siskiwit	07/13/98		69	
Siskiwit	08/18/98		72	
Siskiwit	09/11/98		79	
Siskiwit	05/26/99	18	41	

Locator Lake

Phytoplankton were collected from the littoral and limnetic zones of Locator Lake from June through September, 1997. In 1998, phytoplankton were collected in the limnetic zone from May through September, and in 1999, phytoplankton were collected in May from the limentic zone.

1997 Limnetic: In 1997, the month of June started with nearly identical percent community composition for the diatoms (27%), blue-greens (26%), and green algae (26%) (Figure 8-28). Yellow-green algae composed a noticeable percentage of the community also (13%). By July, the percent composition for yellow-green algae decreased by half (to 13%) and diatoms increased (to 37%). In July, cryptomonads (5%) and dinoflagellates (6%) were also significant components of the phytoplankton community. Diatoms increased to make up 47% of the community by August, and green algae rebounded from the July low to compose 24% of the community. Blue-greens decreased to make up only 5% of the community. By September, yellow-green algae increased to become the most common algal group in the community at 33% while diatoms and green algae decreased. By September, dinoflagellates had decreased to make up only 1% of the phytoplankton community.

Overall counts of phytoplankton in Locator Lake decreased every month from June to September from a high count of 1.4×10^6 cells or colonies/liter to a low count of 6.5×10^5 cells or colonies/liter (Figure 8-31). A total of 39 taxa were identified from Locator Lake in 1997.

1997 Littoral: Diatoms dominated through much of the season, as in the littoral zone (Figure 8-29). Starting at 45% of the community in June, diatoms increased until September when percent relative abundance fell to 14%. With this decrease in diatoms was an enormous increase in green algae. Through the season, green algae composed 7-13% of the community, but in September, they increased to make up 59% of the littoral community. Blue-greens were fairly abundant, with concentrations near 26% of the community in June and July, followed by a decrease to 4% in August and a slight rebound to 11% in September. Abundant species included *Rhizoselenia* sp. in June, *Merismopedia tenuissima* in June and July, *Tabellaria fenestrata* in July and August, and *Asterionella formosa* in August.

Patterns in the phytoplankton community were quite different in 1998 (Figure 8-28). In May, yellow-green algae were the dominant group, composing 41% of the phytoplankton community. Yellow-green algae then decreased through the rest of the season to a low in September of 21% of the community. Meanwhile, blue-greens increased from May to September from a low of 3% of community composition to 28%. Diatoms increased from May (25%) through July (34%) and then dropped off slightly in August (20%). Like the diatoms, the green algae increased for most of the season to a high of 24% in August but then decreased in September to 13% of the community. As in 1997, both cryptomonads and dinoflagellates composed a noticeable portion of the community for part of the season. In May, cryptomonads were 12% of the community, but the counts decreased dramatically for the rest of the season. Dinoflagellates were in low abundance in May, but in June and July, they accounted for 10% of the phytoplankton community. In August and September, that percentage dropped to 4-6%.

The interactions among algal groups led to a community in August where the four major groups were almost equal, with percent community compositions ranging from 20-24%. By September, the diatoms and blue-greens slightly out-competed the other two groups.

Phytoplankton counts overall followed a pattern similar to that seen in 1997 (Figure 8-31). The highest counts were early in the season and decreased consistently through September.

The highest count was actually in June (1.1 X 10^6 cells or colonies/liter), although it was only marginally higher than the May count (1.0X 10^6 cells or colonies/liter). The counts then decreased to a low of 8.1 X 10^5 cells or colonies/liter in September. A total of 38 taxa were identified in Locator Lake in 1998.

Patterns and concentrations of limnetic chlorophyll *a* were similar between years in Locator Lake (Figure 8-30). Except for a July 1998 peak, chlorophyll *a* increased from May to September during both years. Concentrations in 1997 were slightly higher than in 1998. In 1997, concentration increased from 3.7 to 8.9 mg/l. Concentrations in 1998 increased from 0.9-8.3 mg/l, with a peak in July of 11.4 mg/l.

In May of 1999, four groups of algae were almost equal in their percent community composition: diatoms (24.5%), yellow-green algae (22.2%), cryptomonads (24%), and euglenoids (22.2%). Green algae only composed 4% of the community. This pattern was similar to the pattern seen in June of 1997. Organism abundance in May of 1999 (7.3 X 10^5 cells or colonies/liter), however, was considerably lower than previous early-summer samples from Locator Lake.

The phytoplankton community in Locator Lake was quite varied. During many months there was a dominant algal group, but the percent community composition was never as high as seen in some lakes in this study. Further, there were numerous months in which two or three (or even four) groups were about equal in percent community composition. Commonly, *Chlorochromonas* spp. and *Rhizoselenia eriensis* composed greater than 10% of the community, but there were numerous other species that periodically composed greater than 10% of the community (Table 8-8). There was seasonal variation among the algal groups, but it was more limited than in some of the other lakes. One of the more interesting characteristics of Locator Lake was the abundance of dinoflagellates in the samples. Locator Lake had the highest percentage of dinoflagellates among all the lakes included in this study.

The abundance of yellow-green algae early in the season coupled with the high percentage of yellow-green algae and blue-greens indicate a low concentration of available silica as a result of spring mixing. According to the water chemistry results, silica concentrations are relatively low in Locator Lake and are low at the beginning of the sampling season. The other macronutrients are available in higher relative concentrations, which would favor yellow-green algae, green algae, or blue-greens.

Figure 8-28. Relative abundance of algal groups (common names given for Phyla) collected from the limnetic zone of Locator Lake in A) 1997 and B) 1998. Triplicate samples were collected from 1 meter below the surface above the deepest part of the lake.

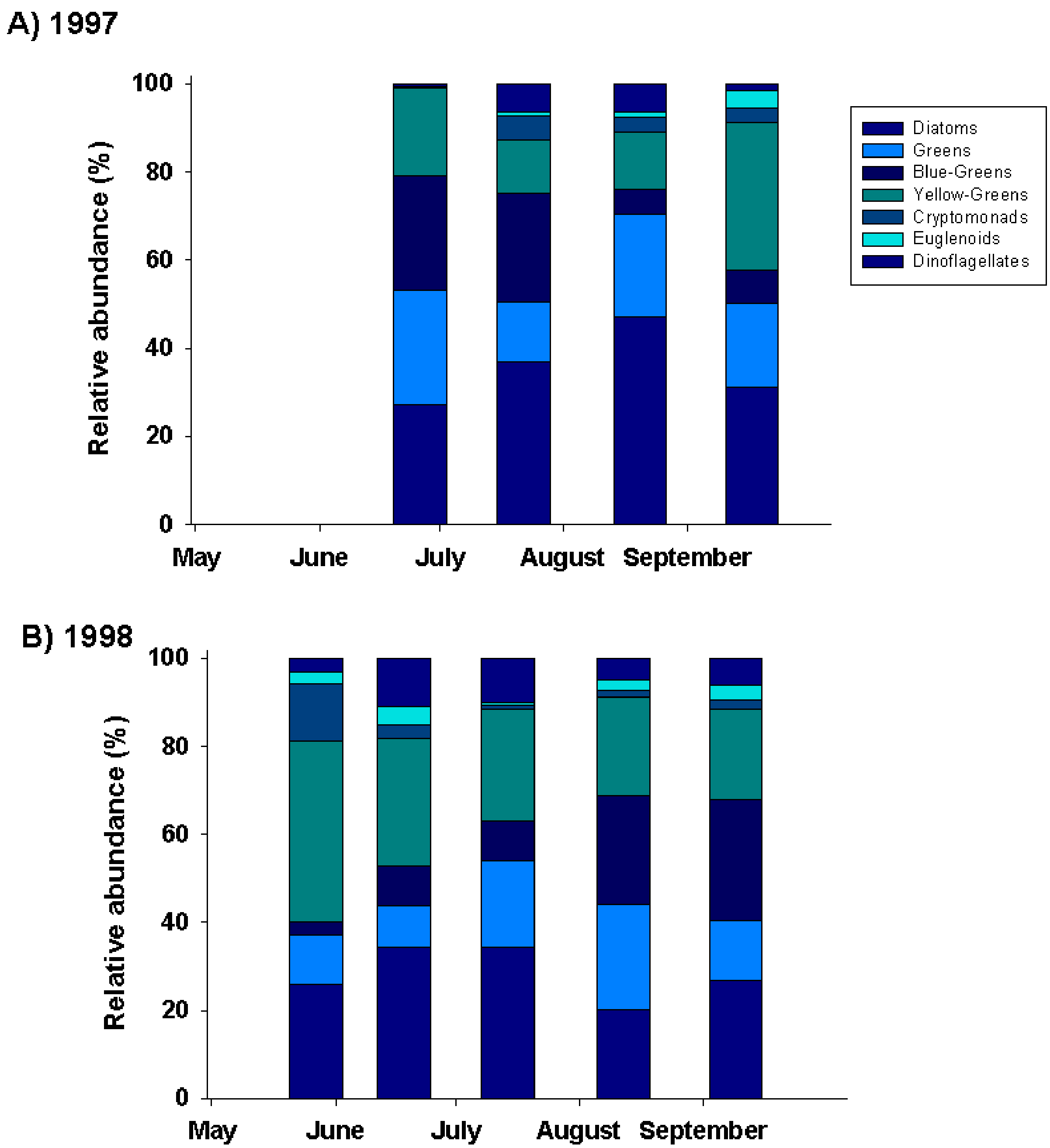


Figure 8-29. Relative abundance of algal groups (common names given for Phyla) collected from the littoral zone of Locator Lake in 1997. Triplicate samples were collected from 1 meter below the surface.

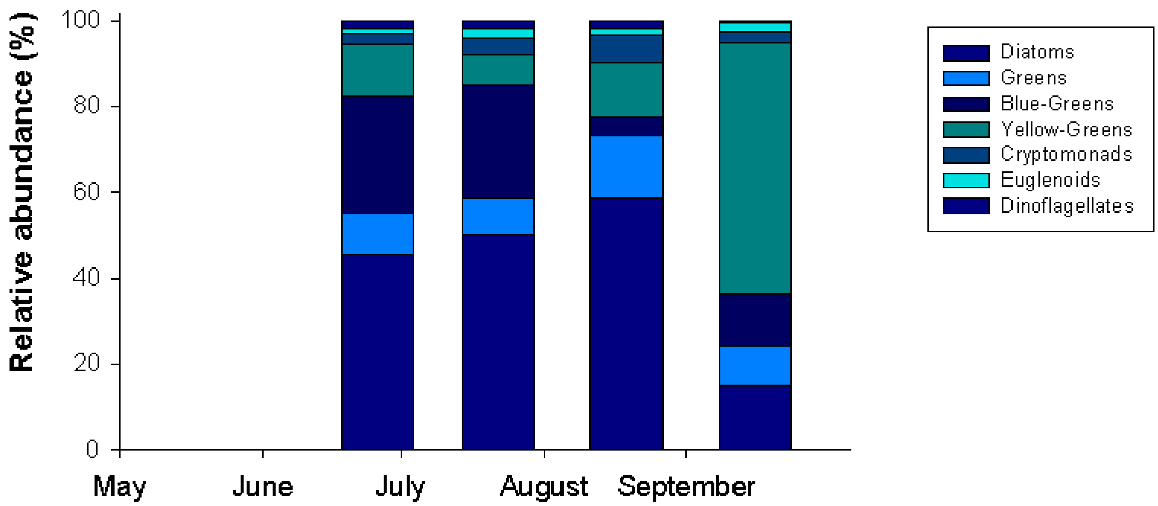


Figure 8-30. Chlorophyll *a* concentration in Locator Lake. Data points represent samples taken from 1 meter below the surface at the deepest point in the lake.

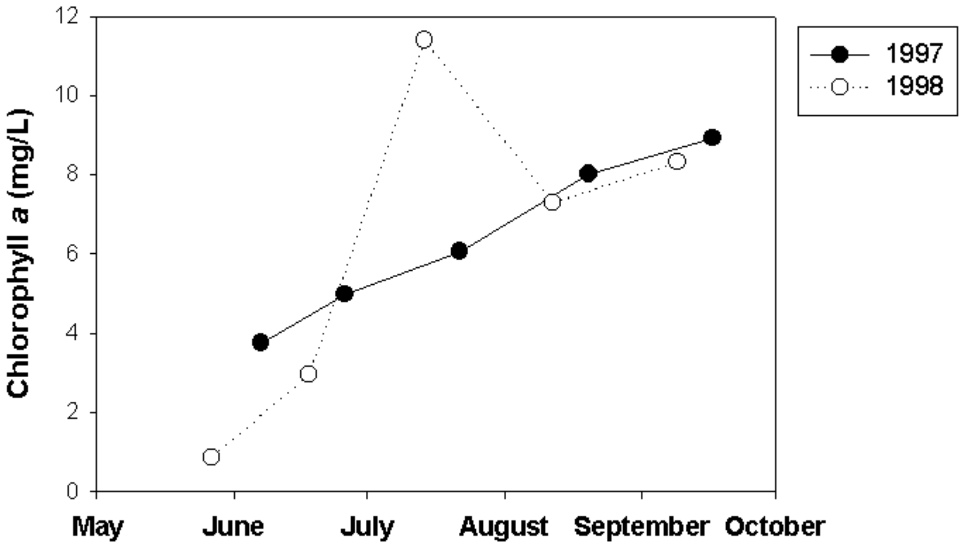


Figure 8-31. Total abundance of limnetic phytoplankton collected each month in Locator Lake. Counts calculated from three composited samples taken from 1 meter below the surface at the deepest point in the lake.

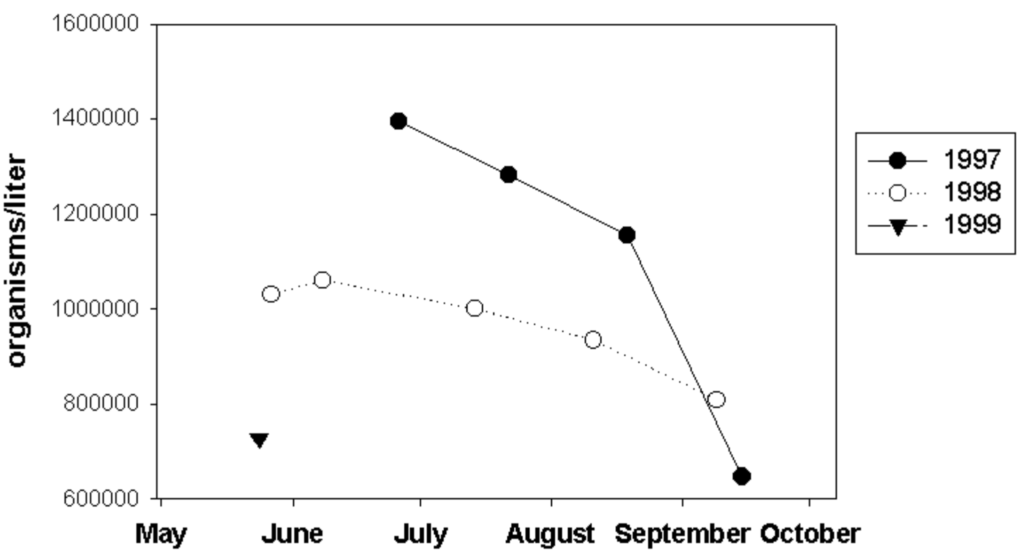


Table 8-8. Phytoplankton taxa that amounted to ≥ 10% of the relative abundance in limnetic zone of Locator Lake with percent relative abundance.

		Asterionella formosa (indiv)	*Cyclotella* spp.	*Rhizosolenia eriensis*	*Tabellaria fenestrata*	*Sphaerocystis schroeteri*	*Chlorochromonas* spp.
Locator	6/26/97			11		12	19
Locator	7/22/97			11	14		11
Locator	8/19/97	20	56			15	11
Locator	9/15/97			21		12	22
Locator	5/27/98					10	38
Locator	6/8/98			15			23
Locator	7/14/98			19		11	22
Locator	8/11/98						16
Locator	9/9/98			12	10		16
Locator	5/24/99			12			16

		Cryptomonas spp.	*Chroococcus* spp.	*Merismopedia tenuissima*	*Glenodinium/ Peridinium*	*Trachelomonas* spp.
Locator	6/26/97			17		
Locator	7/22/97			13		
Locator	8/19/97					
Locator	9/15/97					
Locator	5/27/98	13				
Locator	6/8/98				10	
Locator	7/14/98					
Locator	8/11/98			13		
Locator	9/9/98		11			
Locator	5/24/99	24				22

Mukooda Lake

Mukooda Lake was sampled for phytoplankton between June and September 1997 in the littoral and limnetic zones. Samples were collected in the limnetic zone between May and September 1998 and in May of 1999.

1997 Limnetic: In 1997, the phytoplankton community in Mukooda Lake was mostly stable (Figure 8-32). In June, the phytoplankton percent community composition was closely distributed among the four major algal groups: yellow-green algae 27%, blue-greens 26%, green algae 20%, and diatoms 22%. By July, the blue-greens had increased dramatically (to 45%) and the diatoms slightly (to 27%) at the expense of the yellow-green algae (decreased to 4%). In August, yellow-greens increased slightly as the other three groups decreased only slightly. By September, the blue-greens decreased dramatically (to 15%) corresponding with an increase in green algae, yellow-green algae, and a slight increase in diatoms. Cryptomonads were present only in low numbers, and dinoflagellates were absent all months except for August when they constituted only 0.48% of the phytoplankton community.

Overall phytoplankton abundance peaked in July at 1.2 X 10^6 cells or colonies/liter, and then numbers decreased to 6 X 10^5 cells or colonies/liter in August before a slight increase in September (Figure 8-35). A total of 38 taxa were identified from Mukooda Lake.

1997 Littoral: The phytoplankton community in the littoral zone was similar to the limnetic zone (Figure 8-33). There was not one group of algae that clearly dominated all season. Diatoms were more prevalent in the littoral zone and increased through the season from composing 10% of the community to composing 60% of the community. Blue-greens composed between 30 and 37% of the community until September, when their concentration decreased to 10% of the community. Green algae reached their high concentration in July at 41% relative abundance and otherwise remained below 23% community composition. Yellow-green algae were abundant in June, composing 23% of the community, but otherwise were relatively nonexistent, with concentrations below 2%. Abundant individual phytoplankton species in the littoral zone included *Aphanizomenon flos-aquae* in June and July, *Actinastrum* sp. in July, *Anabaena* sp. in August, *Synedra* sp. in August and September, and *Tabellaria fenestrata* in September.

Patterns in 1998 for Mukooda Lake were quite different from 1997 (Figure 8-32). In May, diatoms, green algae, and yellow-green algae made up equal percentages of the phytoplankton community. By June, the green algae successfully out-competed both of those groups to compose 62% of the phytoplankton community. Diatoms had decreased to 17% of the community, and yellow-green algae decreased to 4% of the community. The success of the green algae was scaled back in July (to 43%) as diatoms and yellow-greens increased to 28% and 13%, and blue-greens in July constituted 10% of the community. Phytoplankton groups evened out in August and September, although different groups were abundant. In August, blue-greens (34%), green algae (24%), and diatoms (25%) were present in comparable concentrations, and in September, yellow-green algae (28%), blue-greens (29%), and green algae (31%) were similarly abundant. Diatoms had decreased to only 5% of the community by September.

The pattern of overall phytoplankton abundance in 1998 was similar to the results in 1997. Overall counts generally increased through the season until a major decrease in August that was followed by slight increase. As in 1997, the lowest count was recorded in August (3.9 X 10^5 cells or colonies/liter); however, the highest phytoplankton count was in June (6.8 X 10^5

cells or colonies/liter). A total of 40 taxa were identified from Mukooda Lake in 1998. Counts in 1997 were generally higher than in 1998.

Limnetic chlorophyll *a* concentrations were comparable between years in Mukooda Lake (Figure 8-34). In 1997, chlorophyll *a* rose from 2.6 mg/l in May to a high of 9.7 mg/l in August then decreased slightly in September to 8.8 mg/l. In 1998, chlorophyll *a* rose between May and June but then decreased to a low of 1.4 mg/l in July. Concentration increased for the rest of the season to a high of 11 mg/l in September.

In May 1999, blue-greens (36%) and diatoms (34%) dominated the phytoplankton community. Yellow-green algae were also abundant, composing 21% of the community. Euglenoids (5%) and cryptomonads (2%) also composed more than 1% of the community. Total organism abundance in May of 1999 (6.3×10^5 cells or colonies/liter) was similar to counts in May of 1998 (5.6×10^5 cells or colonies/liter).

The phytoplankton community in Mukooda Lake was highly variable (Table 8-9). *Chlorochromonas* spp. and *Sphaerocystis schroeteri* were very common, and most months, they composed greater than 10% of the phytoplankton community. Twelve other species of phytoplankton composed greater than 10% of the community during at least one month of sampling.

Differences in the phytoplankton community between years were quite pronounced in Mukooda Lake. The 1997 community was mostly dominated by blue-greens and the 1998 community was primarily dominated by yellow-green algae. The low percentage of diatoms in the lake can be explained in part by the very low concentrations of available silica. In such a situation, groups that do not require silica can easily out-compete the diatoms.

Figure 8-32. Relative abundance of algal groups (common names given for Phyla) collected from the limnetic zone of Mukooda Lake in A) 1997 and B) 1998. Triplicate samples were collected from 1 meter below the surface above the deepest part of the lake.

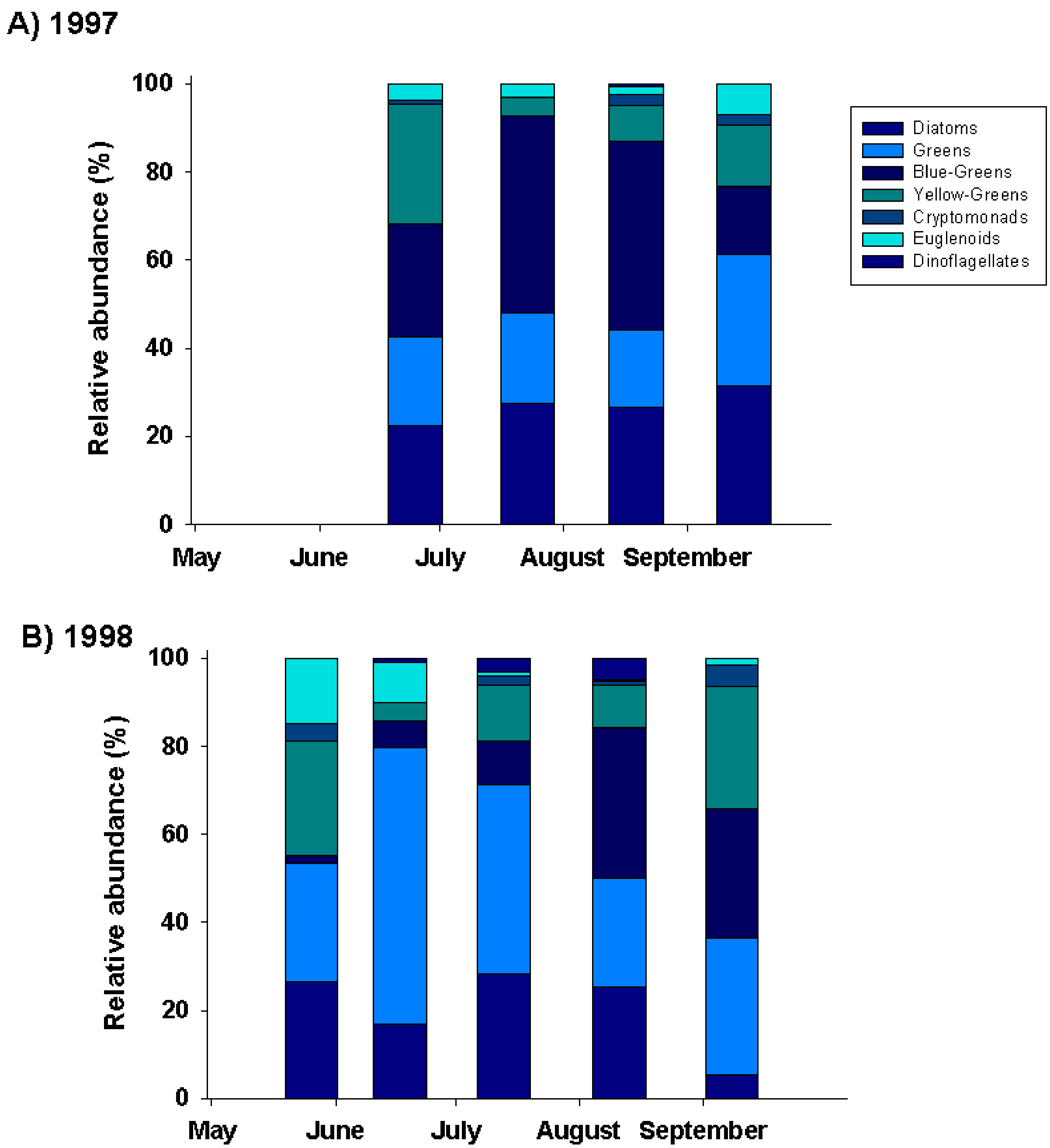


Figure 8-33. Relative abundance of algal groups (common names given for Phyla) collected from the littoral zone of Mukooda Lake in 1997. Triplicate samples were collected from 1 meter below the surface.

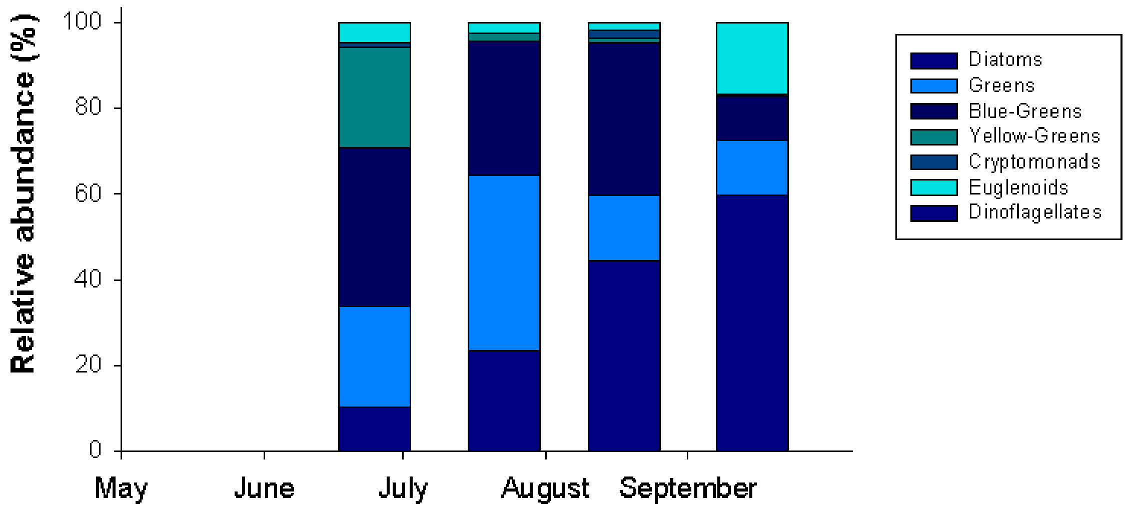


Figure 8-34. Chlorophyll a concentration in Mukooda Lake. Data points represent samples taken from 1 meter below the surface at the deepest point in the lake.

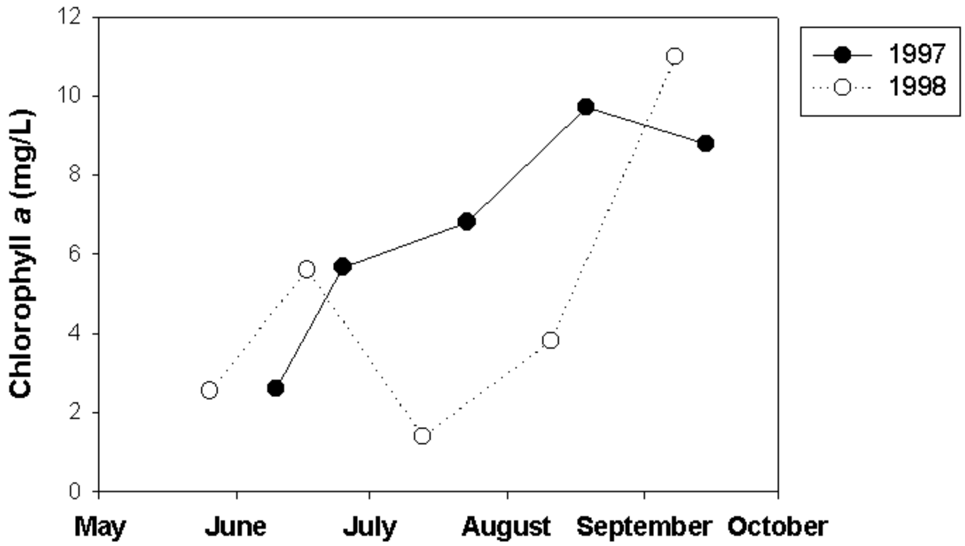


Figure 8-35. Total abundance of limnetic phytoplankton collected each month in Mukooda Lake. Counts calculated from three composited samples taken from 1 meter below the surface at the deepest point in the lake.

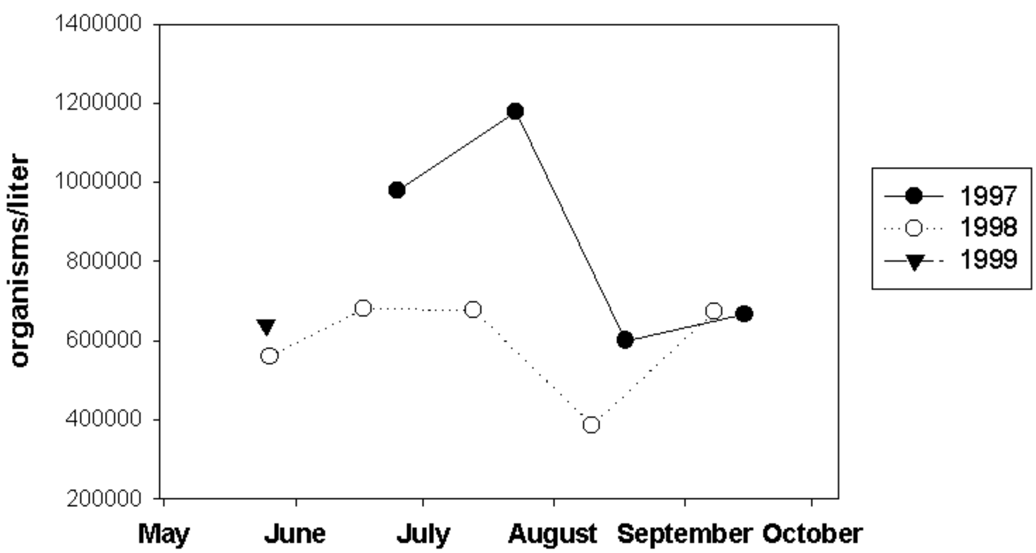


Table 8-9. Phytoplankton taxa that amounted to ≥ 10% of the relative abundance in limnetic zone of Mukooda Lake with percent relative abundance.

		Asterionella formosa (colony)	*Asterionella formosa* (indiv)	*Cyclotella* spp.	*Rhizosolenia eriensis*	*Ankistrodesmus/ Quadrigula*	*Oocystis* spp.	*Sphaerocystis schroeteri*
Mukooda	6/25/97							10
Mukooda	7/23/97							12
Mukooda	8/18/97							11
Mukooda	9/15/97							22
Mukooda	5/26/98			11				14
Mukooda	6/17/98					12	21	18
Mukooda	7/13/98				20	10		10
Mukooda	8/10/98				10	11		
Mukooda	9/8/98							11
Mukooda	5/25/99	13	17					

		Chlorochromonas spp.	*Dinobryon bavaricum*	*Anabaena* spp.	*Aphanizomenon flos-aquae*	*Chroococcus* spp.	*Oscillatoria* spp.	*Trachelomonas* spp.
Mukooda	6/25/97	23			21			
Mukooda	7/23/97			13	28			
Mukooda	8/18/97			32				
Mukooda	9/15/97	14						
Mukooda	5/26/98	24						15
Mukooda	6/17/98							
Mukooda	7/13/98	13						
Mukooda	8/10/98					27		
Mukooda	9/8/98	27		12		13		
Mukooda	5/25/99		20				35	

Chapter 9 Zooplankton Results

Long Lake

The Long Lake zooplankton were sampled in 1997 in the littoral zone. Summer-wide mean (± 1 SE) zooplankton density was 336 ± 196 individuals/liter. Zooplankton peaked in early July to approximately 1100 individuals/liter. During other samplings, the density was near or below 200 individuals/liter (Figure 9-1). The high abundance in July reflected increases across most of the dominant taxa listed in Table 9-1. Abundance was much higher in Long Lake than other lakes in the study. This may be due to the presence of high densities of macrophytes. The 3-dimensional complexity created by the plants offers zooplankton significant refuge from predatory fish. The high primary production also may contribute to the zooplankton abundance by supplying adequate energy sources.

Zooplankton diversity was comparable to the other lakes in the study, with a Margalef's richness of 4.0 and a Shannon-Wiener index of 1.5-1.8. In total, 36 taxa were identified. The taxa were dominated by rotifers, which composed around 75% of the individuals (Figure 9-2). The dominant taxa included the rotifers *Conochilus unicornis* and *Keratella* spp., and the cladoceran *Bosmina longirostris*. The rotifers are very common taxa for lakes in the Great Lakes area (Hutchinson 1967). *C. unicornis* tends to be at maximum densities in early spring then again in July (Hutchinson 1967), as was seen in Long Lake. *B. longirostris* is commonly associated with lakes that are becoming eutrophic (Hutchinson 1967). Long Lake is a eutrophic lake so the high abundance of *B. longirostris* is as expected. Common zooplankton collected from colder, deeper lakes were not found in Long Lake, which also makes it similar to other characteristics of the lake. In general, it appears Long Lake zooplankton reflect the conditions of the lake.

Table 9-1. Zooplankton taxa that amounted to ≥ 10% of the relative abundance in Long Lake.

Zone	Date	*Bosmina longirostris*	*Collotheca* spp.	*Conochilus unicornis*	*Keratella* 1 post. spine	*Polyarthra* spp.
Litt	4-Jun-97	35	10		31	
Litt	24-Jun-97	16		67		
Litt	22-Jul-97			32	45	
Litt	26-Aug-97	17		15	11	26
Litt	16-Sep-97			55	29	

Figure 9-1. Total zooplankton densities in Long Lake. Means ± 1 standard error (error bars) are calculated from triplicate samples.

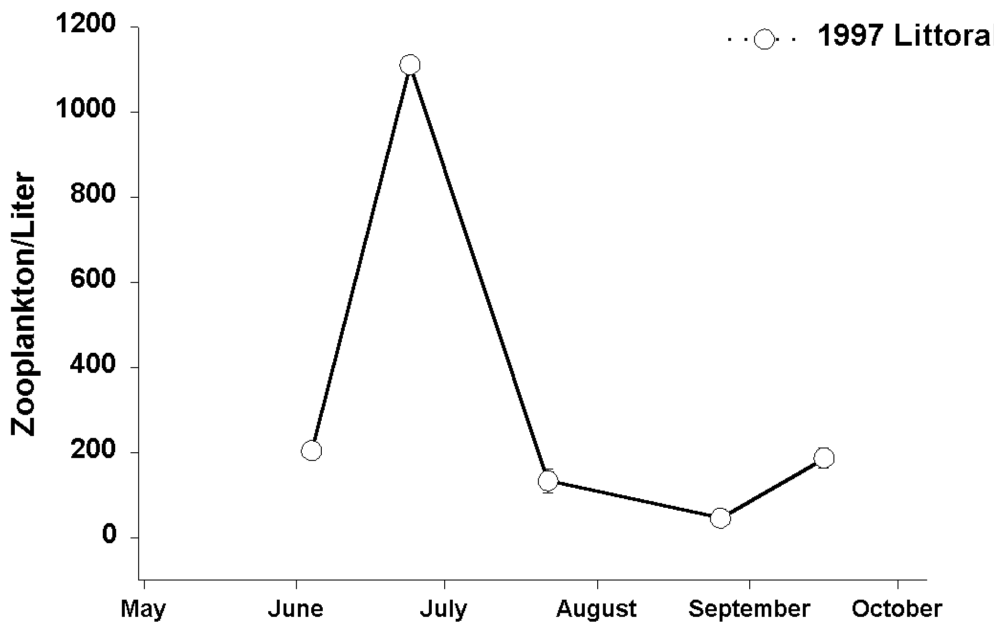
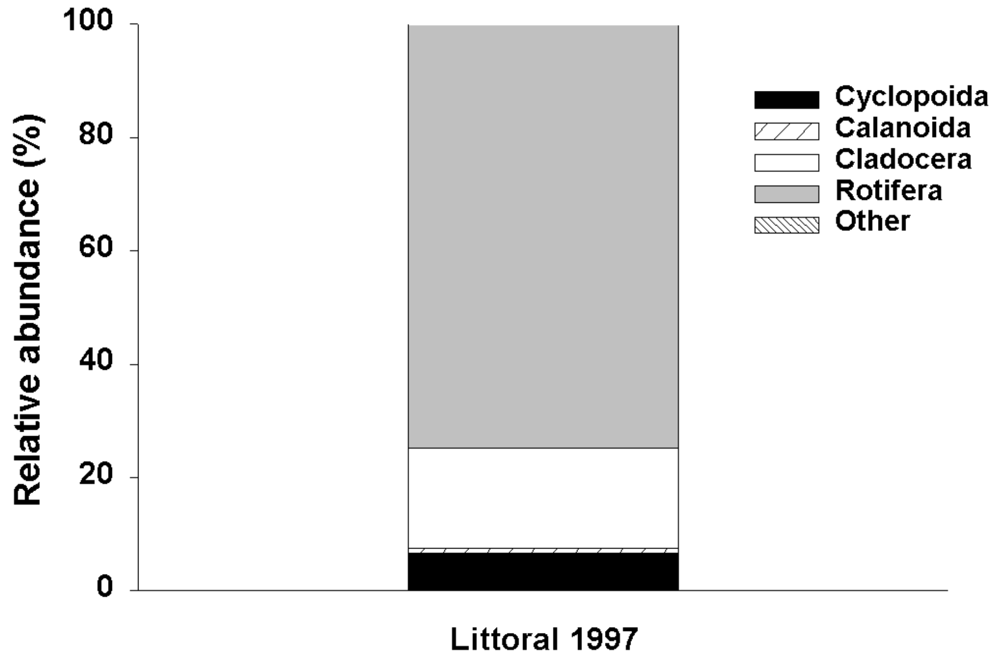


Figure 9-2. Relative abundance of zooplankton groups in Long Lake. Only littoral sampling was done, and sampling was done only in 1997.



Loon Lake

Limnetic zooplankton densities in Loon Lake in 1997 (range: 90-379 individuals/liter) peaked in early July at near 400 individuals/liter. This peak abundance was largely caused by high numbers of *Conochilus unicornis*. The high relative abundance of *Keratella* spp. in May appear to decrease with an associated increase in many other species, especially *Bosmina longirostris* and *C. unicornis*. In contrast, 1998 densities (range: 19-194 individuals/liter) were at a maximum in May and steadily declined through summer (Figure 9-3). *C. unicornis* was not as prominent in 1998 samples as in 1997. Instead, *Keratella* spp. tended to be more important in the overall zooplankton dynamics (Table 9-2). In general, limnetic densities were about two times greater in 1997 (mean = 229, SE = 47) than in 1998 (mean = 77, SE = 31). Total organisms in littoral samples (range: 62-170 individuals/liter) taken in 1997 tended to be fewer than the limnetic densities on all sampling dates. *Keratella* spp. tended to be the only dominant zooplankton in the littoral zone. The daphnids were particularly few in the littoral compared to the limnetic zone. The September sampling did demonstrate high similarity (88%) between the littoral and limnetic community, whereas in May the zones were not particularly similar (54%).

We encountered 35 different taxa in the lake during the study. The Margalef's index was 3.75 and the Shannon-Wiener index was about 2.0-2.3. Community similarity between the littoral and limnetic zones, based on relative abundances, was only 66%. The taxa were dominated by rotifers, which composed about 75-80% of the samples numerically (Figure 9-4). *C. unicornis*, *Kellicottia longispina*, *Keratella* spp., and *Polyarthra* spp. were the most numerous rotifers in the limnetic zone. *Keratella* spp. dominated the littoral rotifers. *Bosmina longirostris* and *Diacyclops thomasi* were the most common cladocerans and copepods in the limnetic samples, and *B. longirostris* was also numerous in littoral samples on a single date. The occurrence of *Polyarthra* spp. was interesting because this genus is typically found in deep waters (Hutchinson 1967). Some shifts in the taxa abundance, such as the relative changes in *Kellicottia longispina* and *C. unicornis* from May to July 1997, could suggest that some competition for similar resources is occurring. In general, the diversity of zooplankton was high in Loon Lake, which could indicate that the system is not particularly stressed.

Table 9-2. Zooplankton taxa that amounted to ≥ 10% of the relative abundance in Loon Lake.

Site	Date	*Bosmina longirostris*	*Daphnia retrocurva*	*Conochilus unicornis*	*Gastropus* spp.	*Filinia terminalis*	*Kellicottia longispina*	*Keratella quadrata*	*Keratella* 1 post. spine	*Ploesoma truncatum*	*Polyarthra* spp.	*Synchaeta* spp.
Limn	21-May-97						14		72			
Limn	26-Jun-97	35		10				10	20			
Limn	21-Jul-97			74					11			
Limn	25-Aug-97		12	27					26			
Limn	14-Sep-97		31						42			
Limn	20-May-98			11			34		30			
Limn	17-Jun-98	12					52					
Limn	14-Jul-98			12					11		44	
Limn	12-Aug-98	16							33			
Limn	9-Sep-98	20		20					23	16		
Limn	19-May-99					16			47		14	10
Litt	21-May-97								82			
Litt	26-Jun-97	33			19							18
Litt	14-Sep-97								67			

Figure 9-3. Total zooplankton densities in Loon Lake. Means ± 1 standard error (error bars) are calculated from triplicate samples.

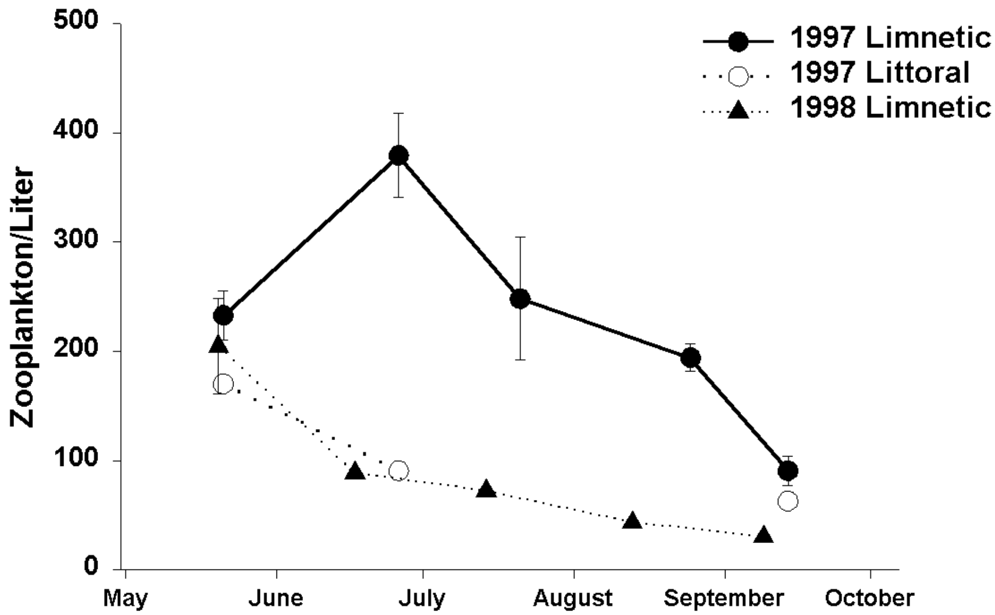
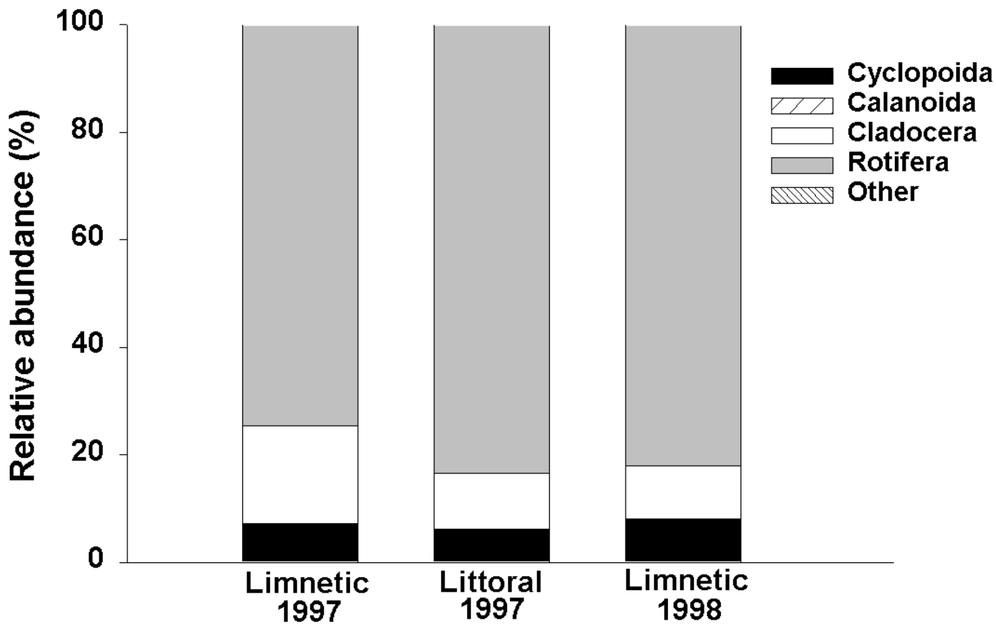


Figure 9-4. Relative abundance of zooplankton groups in Loon Lake calculated from averages of each year.



North Bar Lake

Summer mean zooplankton densities were similar in the limnetic zone in North Bar Lake in 1997 and 1998. Total individuals were 221 ± 72 in 1997 and 251 ± 52 in 1998. However, the change in densities differed between the two years. In 1997, densities ranged from 74-490 individuals/liter, and the peak density occurred in late September, whereas densities ranged from 19-458 individuals/liter and peaked in May then remained around 225 individuals/liter during other months in 1998 (Figure 9-5). Littoral densities were much lower than limnetic densities in 1997; densities ranged from 8 to 71 individuals/liter. In addition, only a few samples could be processed from the littoral zone, which made further comparisons difficult. The peak in 1997 was caused mostly by a high number of nauplii in samples. In contrast, the peak in 1998 was due largely to high abundance of *Kellicottia longispina*. The cladoceran, *Bosmina longirostris*, was high in the spring in both 1997 and 1998.

In total, 38 taxa were encountered in the lake during the study, which is similar to most other lakes in the study. The Margalef's index was 4.1 (slightly higher than the other Sleeping Bear Dunes lakes), and the Shannon-Wiener index was around 2.3 to 2.5 (comparable to other lakes). As in other lakes, the rotifers generally were the numerically dominant zooplankton group (Figure 9-6). Some interesting taxa were found. For example, *Trichocerca* spp. was present at times, and densities peaked in September and June 1997; no definite peak was detected in 1998. This genus is usually associated with sandy habitats (Hutchinson 1967), but we collected it mostly in the limnetic samples and not the littoral samples. *Polyarthra* spp. was present in late summer 1997, but the genus is known to peak in mid-summer (Hutchinson 1967), as it did in 1998. They were also present in high numbers in the littoral zone, which is unusual, too, because they are typically cold-water species (Hutchinson 1967). *Asplanchna* spp., also a cold-water genus (Hutchinson 1967), was also found in the limnetic samples after the lake had stratified. It is possible that some individuals are moving between the hypolimnion and epilimnion. This movement could add variation to what is collected. Compared to other lakes in this study, *Conochilus unicornis* was not very abundant. Instead, *Kellicottia longispina* and *Keratella* spp. were common (Table 9-3). *Keratella* and *Polyarthra* spp. were also found in high numbers in the littoral zone, along with *Ploesoma truncatum*, which was not particularly numerous in the limnetic zone. In general, the littoral and limnetic zones were not very similar (approx. 23%), except in May (75%). Two copepods were particularly abundant in the limnetic zone, *Diacyclops thomasi* and *Tropocyclops prasinus mexicanus*. The cladoceran *Bosmina longirostris* was abundant in both the limnetic and littoral zones. Of particular note was the occurrence of the exotic *Dreissena polymorpha* veligers in samples from the limnetic zone in August and September of 1998. This species was not found in any samples in 1997. Between-year similarity was only 54%, which is relatively low compared to the other lakes studied. The implication for this is that yearly variation is high (possibly because of North Bar Lake's connection with Lake Michigan) and short-term monitoring will be difficult to interpret. Longer (possibly decades) data sets will be needed to compensate for this variation.

Table 9-3. Zooplankton taxa that amounted to ≥ 10% of the relative abundance in North Bar Lake.

Site	Date	*Tropocyclops prasinus mexicanus*	*Bosmina longirostris*	*Eubosmina coregoni*	*Daphnia retrocurva*	*Conochilus unicornis*	*Gastropus* spp.	*Kellicottia longispina*	*Keratella* 1 post. spine	*Polyarthra* spp.	Synchaeta spp.	Trichocerca *multicrinis*
Limn	21-May-97		23							44		
Limn	25-Jun-97		38			13						30
Limn	22-Jul-97	12	24						37	13		
Limn	26-Aug-97	11			10	11			12	10		
Limn	15-Sep-97								24	25		16
Limn	20-May-98		11					58	21			
Limn	18-Jun-98		16		27		12					
Limn	15-Jul-98			13	15				21	25		
Limn	12-Aug-98		10						65			
Limn	10-Sep-98		27						29	11		
Limn	19-May-99		42						12	28		
Litt	21-May-97		12						17	55		
Litt	26-Aug-97		23						66			
Litt	15-Sep-97								35		16	

Figure 9-5. Total zooplankton densities in North Bar Lake. Means ± 1 standard error (error bars) are calculated from triplicate samples.

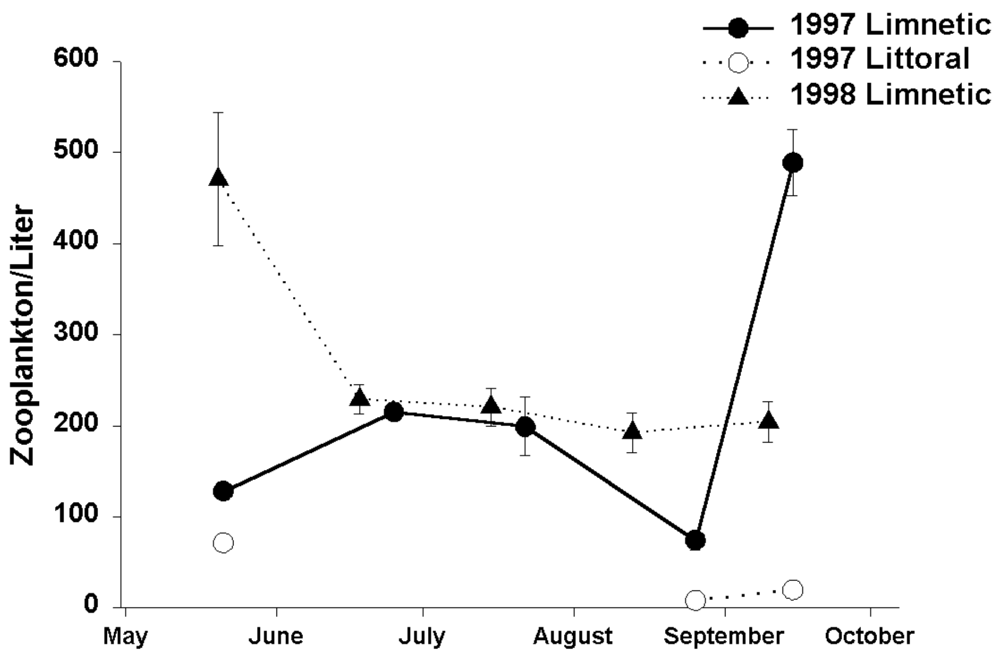
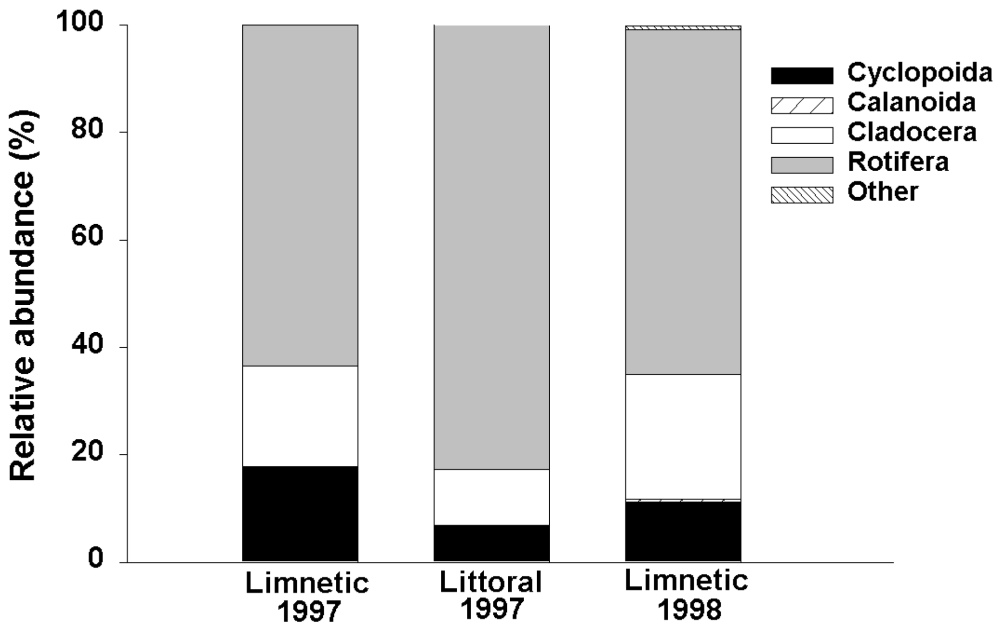


Figure 9-6. Relative abundance of zooplankton groups in North Bar Lake calculated from averages of each year.



Round Lake

Round Lake was sampled only in 1998, so littoral samples were not collected. In the limnetic zone, summer density was 189 ± 50 individuals/liter, and the range was 89-383 individuals/liter. Zooplankton peaked around 400 individuals/liter in May then remained 3 to 4 times lower during the remainder of the sampling period (Figure 9-7). This peak and subsequent decline was mostly due to the dynamics of *Kellicottia longispina.* The decline in *K. longispin*a was followed with an increase in *Keratella* spp. and *Bosmina longirostris*.

We recorded 31 taxa during the study, which translates into a Margalef's index of 3.8 and a Shannon-Wiener index of about 2.0-2.1. These indices were not particularly high, but the data set was substantially reduced compared to the other lakes. The taxa were dominated by the rotifers (Figure 9-8), which were composed mostly of *Keratella* spp. *Conochilus unicornis*, a species that was common in most lakes studied, was abundant only in spring, when it is known to be at peak densities (Table 9-4). *Collotheca* spp. was abundant on a single date, but was present in lower numbers throughout the study. The presence of *Polyarthra* spp. in higher numbers in fall was usual for lakes in this region. The Cyclopoid copepods were the next most abundant group, and these were composed mostly of *Tropocyclops prasinus mexicanus*. However, they were abundant only in the September samples. *B. longirostris* was the most common of the cladocerans, but also found in abundance were *Daphnia retrocurva* and *Diaphanosoma birgei* (a species not found in high abundance in other lakes in the study). In general, the zooplankton community was representative of typical lakes in the region. The presence of this diverse zooplankton community is a good indicator of a healthy ecosystem.

Table 9-4. Zooplankton taxa that amounted to ≥ 10% of the relative abundance in Round Lake.

Site	Date	*Tropocyclops prasinus mexicanus*	*Bosmina longirostris*	*Daphnia retrocurva*	*Diaphanosoma birgei*	*Asplanchna* spp.	*Collotheca* spp.	*Conochilus unicornis*	*Kellicottia longispina*	*Keratella* 1 post. spine	*Polyarthra* spp.	*Trichocerca multicrinis*
Limn	19-May-98							10	75	10		
Limn	17-Jun-98								11	70		10
Limn	14-Jul-98	12	37							23		
Limn	11-Aug-98					10	29			23		
Limn	9-Sep-98	25			18					23	14	
Limn	18-May-99			59				20				

Figure 9-7. Total zooplankton densities in Round Lake. Means ± 1 standard error (error bars) are calculated from triplicate samples.

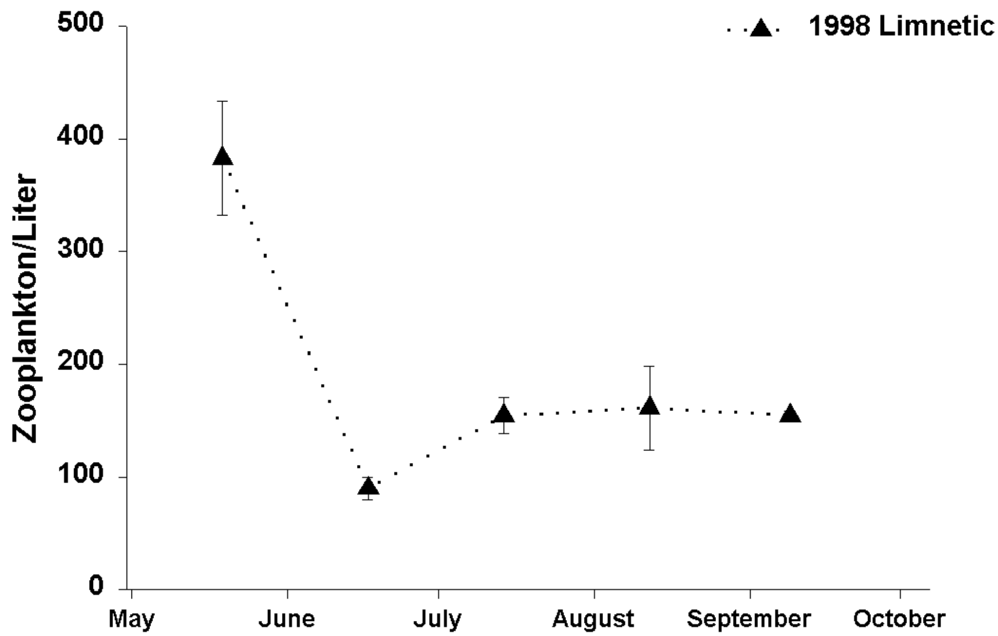
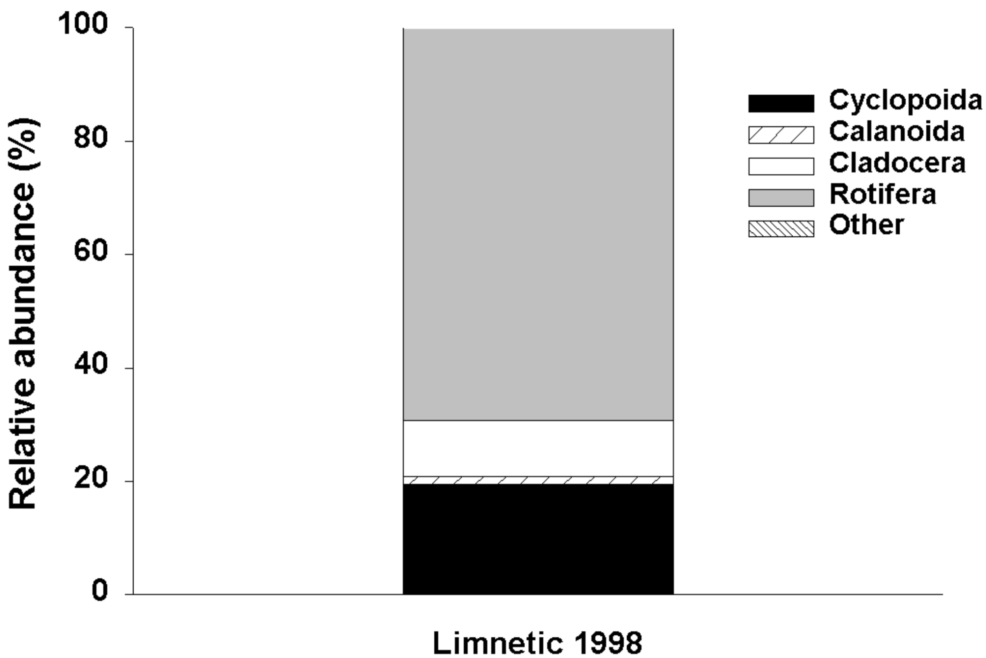


Figure 9-8. Relative abundance of zooplankton groups in Round Lake. Only limnetic sampling was done, and sampling was done only in 1998.



Beaver Lake

Although summer mean zooplankton densities were fairly similar in 1997 (95 ± 12 individuals/liter) and 1998 (188 ± 84 individuals/liter), the range was noticeably greater in 1998 (49-422 individuals/liter) than in 1997 (62-133 individuals/liter). The range of densities in the littoral zone (46-478) was even larger than in the limnetic zone in 1998 (Table 9-5). This means that there is a high amount of temporal variability in the data, and this variability could be expected in future monitoring. For example, zooplankton densities were consistent throughout the study period in 1997, whereas the densities were bimodal in 1998, with peaks in both May and August. The 1998 peak was due mostly to the abundant *Conochilus unicornis* and *Keratella quadrata.* These two species composed 40% and 54% of the sample, which is unusual because a single dominant rotifer is more often collected. However, the abundance of *K. quadrata* dropped to near 0 individuals/liter after this sample date. The second peak in 1998 was mostly *C. unicornis* (96%). Interestingly, littoral densities in 1997 also showed a possible bimodal distribution, in contrast to the limnetic densities from the same year. Again, the peak in June was due to *C. unicornis* (97%), but for the remainder of the sampling *Keratella* spp. were dominant (Table 9-5).

We encountered 32 different taxa during the study. The Margalef's index was 3.6 and the Shannon-Wiener index could be between 1.7 and 2.1, depending on the number of samples collected (i.e., the curve generated from the accumulated data was not uniform in direction). These indices were some of the lowest among the lakes in this study. The lake is not very heterogenous as far as habitats are concerned. This lack of structure may have contributed to the low diversity in the zooplankton community. The rotifers dominated the taxa, composing over 90% of the animals collected (Figure 9-10). In particular abundance were *Keratella quadrata* and an unknown *Keratella* sp., *Conochilus unicornis* (especially high in 1998), and *Kellicottia longispina* (not particularly abundant in 1998). Similiarity of the communities, based on relative abundance, between 1997 and 1998 was only 47%. This was the lowest for all lakes studied and indicates a high amount of variability between years. In a practical sense, short-term monitoring information would be less useful and more difficult to interpret. Longer-term monitoring will be needed to identify trends in the zooplankton community. The littoral species were also mostly rotifers, although similarity between littoral and limnetic samples was fairly low (37-52%). Of interest in the whole lake were *Epischura lacustris* and *Skistodiaptomus oregonensis* (Copepoda) that are species typical of deep northern lakes (Hutchinson 1967). Another cold-water species, *Daphnia pulex* (Hutchinson 1967), was also found in dominant numbers (89% in May 1999). This community of zooplankton commonly occurs in this geographic area and indicates a lake that is mostly oligotrophic (Hutchinson 1967).

Table 9-5. Zooplankton taxa that amounted to ≥ 10% of the relative abundance in Beaver Lake.

Site	Date	*Daphnia pulex*	*Collotheca* spp.	*Conochilus unicornis*	*Gastropus* spp.	*Kellicottia longispina*	*Keratella hiemalis*	*Keratella quadrata*	*Keratella* 1 post. spine	*Ploesoma truncatum*	*Polyarthra* spp.	*Synchaeta* spp.	Unknown rotifer
Limn	24-May-97					24	26	22			14		
Limn	8-Jul-97			35				21				15	
Limn	22-Jul-97		19		11	35			12				
Limn	26-Aug-97			74					11				
Limn	23-Sep-97			28				11	56				
Limn	20-May-98			39				54					
Limn	17-Jun-98			94									
Limn	15-Jul-98									69		14	
Limn	12-Aug-98			96									
Limn	10-Sep-98			74		11							
Limn	20-May-99	89											
Litt	24-May-97					17	13		17		43		
Litt	8-Jul-97			97									
Litt	22-Jul-97		11			16				21			21
Litt	26-Aug-97			13					41			26	
Litt	23-Sep-97			42					50				

Figure 9-9. Total zooplankton densities in Beaver Lake. Means ± 1 standard error (error bars) are calculated from triplicate samples.

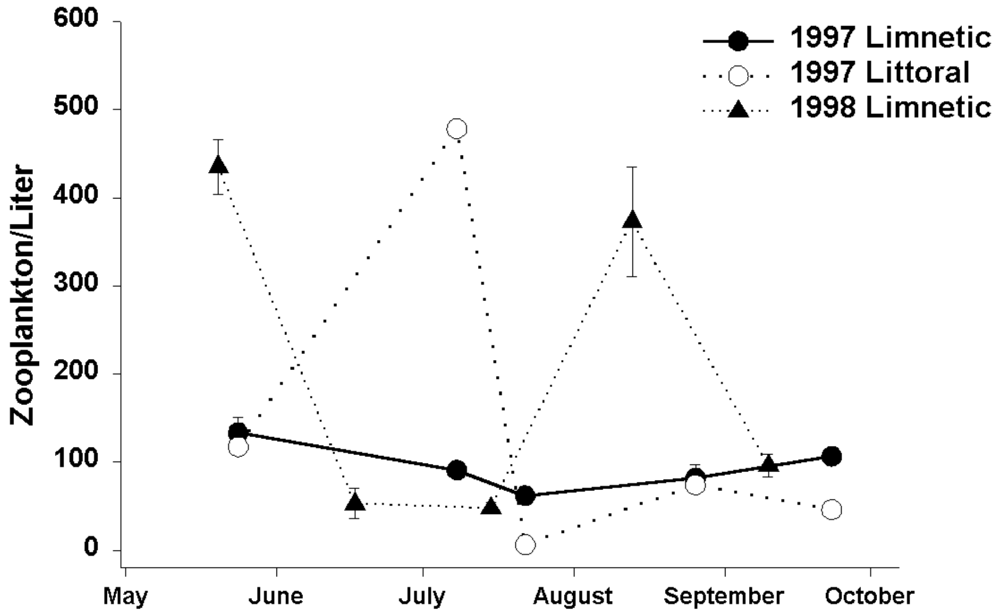
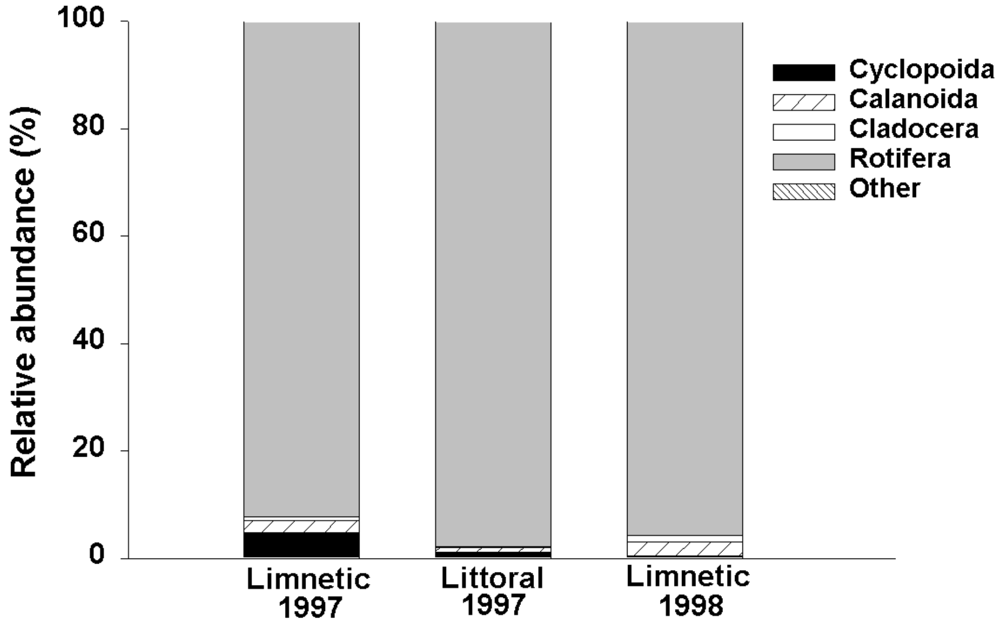


Figure 9-10. Relative abundance of zooplankton groups in Beaver Lake calculated from averages of each year.



Grand Sable Lake

Grand Sable Lake had low zooplankton densities relative to Beaver Lake and many of the other lakes in the study. Sampling at Grand Sable began in early July 1997, and the summer peak may have been missed. The summer peak density in 1998 was during late June (Figure 9-11) and was predominantly the result of a high number of nauplii in the sample (approx. 25% of the total). Summer mean densities were 26 ± 1 individuals/liter in 1997 (range: 23-28 individuals/liter) and 45 ± 11 individuals/liter in 1998 (range: 26-86 individuals/liter). Littoral densities tended to be slightly lower than the limnetic samples. Because personnel at Pictured Rocks changed the lake to be sampled after the season had begun, the smaller number of data points from Grand Sable Lake limits the ability to draw conclusions for comparisons of the limnetic and littoral and between years.

Margalef's index was 3.9 and Shannon-Wiener index was 2.3-2.5. Despite the low abundance of zooplankton, taxa diversity was still comparable to other lakes. We encountered 31 different taxa during the study. Rotifers were the dominant taxa in the limnetic zone in 1998, but the cladocerans were particularly numerous in 1997 limnetic samples (Figure 9-12). The cladocerans from 1997 samples were composed mostly of *Bosmina longirostris*, *Daphnia galeata mendotae*, and *D. longiremis*, but there were other numerous taxa contributing as well. Community similarity calculated from relative abundance data was 67% between years. Comparing the littoral community and the limnetic community in 1997, we found that *Tropocyclops prasinus mexicanus* was more abundant in the littoral zone, as was the genus *Synchaeta* in spring of 1999. Likewise, *Conochilus unicornis* was rarely collected from the littoral zone (Table 9-6). *Diaphanosoma birgei* was found in the littoral zone, although it is a species more common in open-water zones. Community similarity ranged from 37-52% between the littoral and limnetic zone.

The cold-water genera *Skistodiaptomus* and *Asplanchna* (Hutchinson 1967) were collected in the lake, which is consistent with the general lake conditions. Other species were found that are general indicators of the water chemistry. However, in Grand Sable Lake there are some inconsistencies. For example, the rotifer *Lecane* spp. and the cladoceran *Holopedium gibberum* are normally found in acid waters (Hutchinson 1967), but Grand Sable Lake is only acidic in the hypolimnion at certain times based on pH profiles (see other section).

Table 9-6. Zooplankton taxa that amounted to ≥ 10% of the relative abundance in Grand Sable Lake.

Site	Date	*Tropocyclops prasinus mexicanus*	*Bosmina longirostris*	*Daphnia galeata mendotae*	*Daphnia longiremis*	*Diaphanosoma birgei*	*Conochilus unicornis*	*Filinia terminalis*	*Kellicottia longispina*	*Keratella* 1 post. spine	*Polyarthra* spp.	*Synchaeta* spp.
Limn	7-Jul-97		16				31		29			
Limn	21-Jul-97						40		17		12	
Limn	25-Aug-97		15	15			21		14			
Limn	22-Sep-97		13	13			15		21	15		
Limn	19-May-98						12		27	28		
Limn	16-Jun-98						69		10			
Limn	14-Jul-98						47					
Limn	11-Aug-98				11					24		
Limn	9-Sep-98		13		10			22		17		
Limn	19-May-99								14			53
Litt	21-Jul-97								26		26	
Litt	25-Aug-97	17				35			30			
Litt	2-Sep-97								55	23		

Figure 9-11. Total zooplankton densities in Grand Sable Lake. Means ± 1 standard error (error bars) are calculated from triplicate samples.

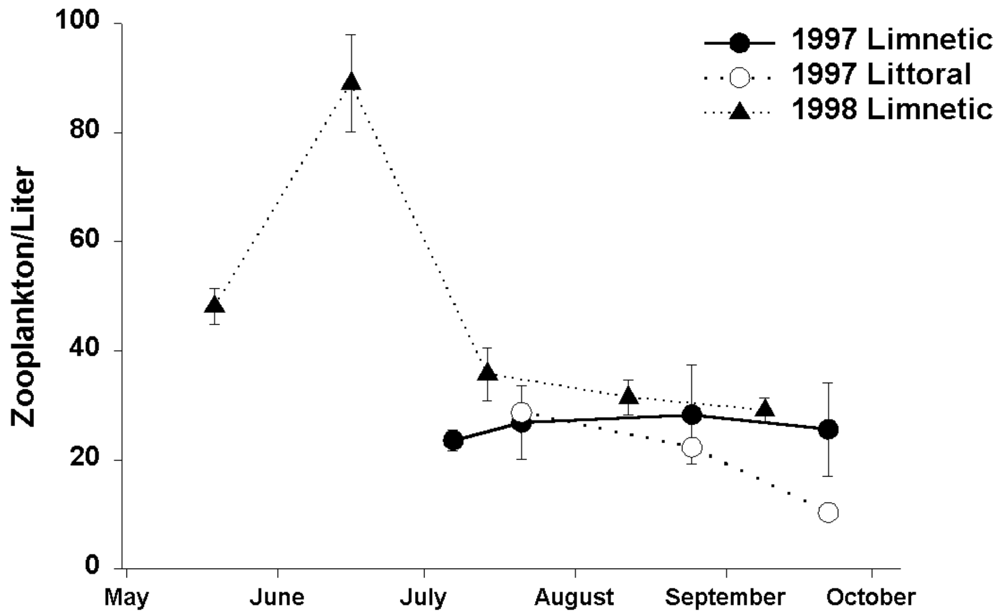
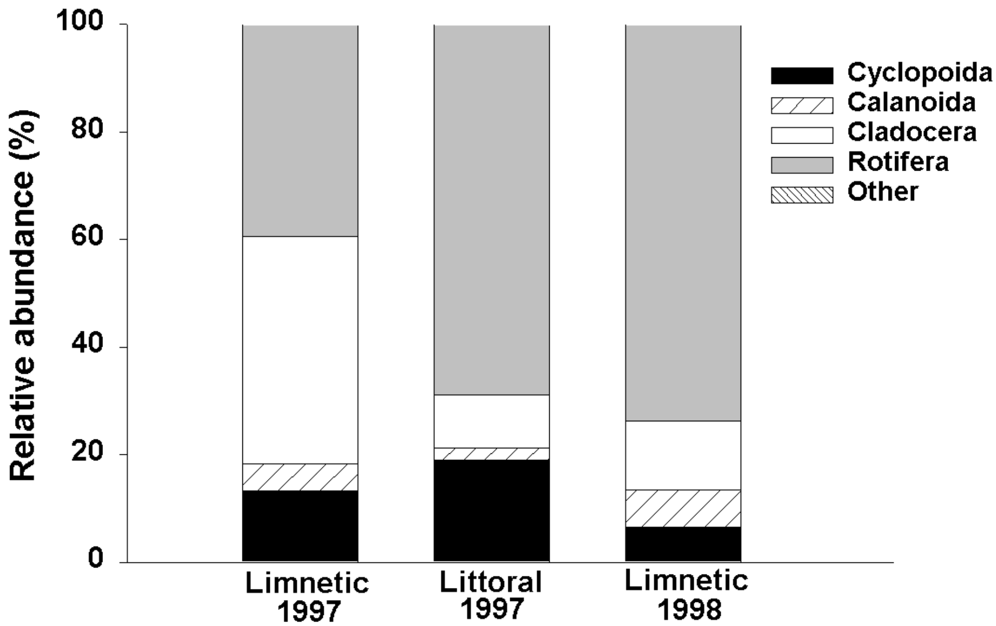


Figure 9-12. Relative abundance of zooplankton groups in Grand Sable Lake calculated from averages of each year.



Sargent Lake

Zooplankton densities in Sargent Lake were similar in 1997 (summer mean = 82 ± 10 individuals/liter) and 1998 (summer mean = 104 ± 20 individuals/liter. Even though densities were more variable in 1998, the range for 1997 (63-111 individuals/liter) and 1998 (50-160 individuals/liter) were also similar. In both 1997 and 1998, the zooplankton showed a bimodal distribution with apparent peaks in early June and again in late September (Figure 9-13). Comparing years, though, *Conochilus unicornis* was low in spring and high in fall in 1997, whereas they were much higher in the spring in 1998. Similarly, *Keratella* spp. were higher in spring than fall in 1997 but just the opposite in 1998. The two peaks of the bimodal distribution were generally because of high numbers of *C. unicornis* in spring and high numbers of *Keratella* spp. in August and September, although *Bosmina longirostris* also contributed to the high abundance in the August and September samples of both years. The decline in numbers in the September 1997 samples was due to reduced numbers of many taxa (i.e., no one taxa was driving this decline). The littoral zone zooplankton followed a pattern similar to the limnetic zooplankton. Densities ranged from 15-91 individuals/liter. The dynamics among taxa in the littoral also were similar to the limnetic zone but more closely resembled 1998 than 1997. In general, the abundance of zooplankton was a combination of interactions among the three main taxa, *C. unicornis*, *Keratella* spp., and *Kellicottia longispina*, but the variation in numbers makes it difficult to be sure what is considered normal for the lake (Table 9-7). Long-term monitoring would be needed to begin to see biologically meaningful patterns.

Margalef's index was 4.5 and the Shannon-Wiener index was 2.4-2.5 for Sargent Lake. In total, 39 taxa of zooplankton were encountered. These are some of the highest taxa richness numbers of the study, which indicates a very diverse zooplankton community. In both years and lake zones, the rotifers were the most numerous group of zooplankton (Figure 9-14). The community similarity was 72% between years, which was relatively high for lakes in this study. The main taxa were *Keratella* sp. and *C. unicornis*. These are very common rotifers and tend to be present in dominant numbers (Hutchinson 1967). Also present in strong abundance were the cladocerans *B. longirostris* and *Holopedium gibberum*. *B. longirostris* is generally an indicator of a lake in transition from oligotrophic to eutrophic (Hutchinson 1967), although we are not certain this is the case for Sargent Lake. The lake does have some areas (e.g., small embayments) that could be more productive than other open-water areas. *H. gibberum* is found only in calcium-poor lakes (Hutchinson 1967), so it is not unusual to find it in the Isle Royale lakes that we sampled. The littoral taxa were similar to the limnetic taxa with the addition of relatively abundant *Tropocyclops prasinus mexicanus* (Copepoda) and *Kellicottia longispina* (Rotifera), but few *C. unicornis* were found here. These differences could explain the rather low similarity of communities (as low as 35%) between the limnetic and littoral zones. *Leptodora kindtii* (Cladocera) was collected, although they were in very low densities and only in June 1998 samples. This species was not commonly collected in any of the lakes included in this study. It is a large predatory cladoceran that can at times control populations of smaller zooplankton. In general, the community of zooplankton in Sargent Lake reflected a large, deep, cold, northern lake. Taxa such as *Epischura*, *Leptodiaptomus*, *Skistodiaptomus*, and *Polyarthra* are common inhabitants of such lakes (Hutchinson 1967).

Table 9-7. Zooplankton taxa that amounted to ≥ 10% of the relative abundance in Sargent Lake.

Site	Date	*Tropocyclops prasinus mexicanus*	*Bosmina longirostris*	*Conochilus unicornis*	*Gastropus* spp.	*Kellicottia Longispina*	*Keratella hiemalis*	*Keratella* 1 post. spine	*Polyarthra* spp.	*Synchaeta* spp.
Limn	28-May-97						10	66		
Limn	30-Jun-97			19		13		24		
Limn	22-Jul-97							34		
Limn	26-Aug-97		19	24				22		
Limn	16-Sep-97		36	12				17		
Limn	22-May-98			50				17		
Limn	16-Jun-98			60						
Limn	14-Jul-98			16				33	15	
Limn	16-Aug-98		40					46		
Limn	10-Sep-98							56	16	
Limn	24-May-99			11	14	33		16		13
Litt	28-May-97							78		
Litt	30-Jun-97							52		
Litt	22-Jul-97	18						72		
Litt	26-Aug-97		25	26				29		

Figure 9-13. Total zooplankton densities in Sargent Lake. Means ± 1 standard error (error bars) are calculated from triplicate samples.

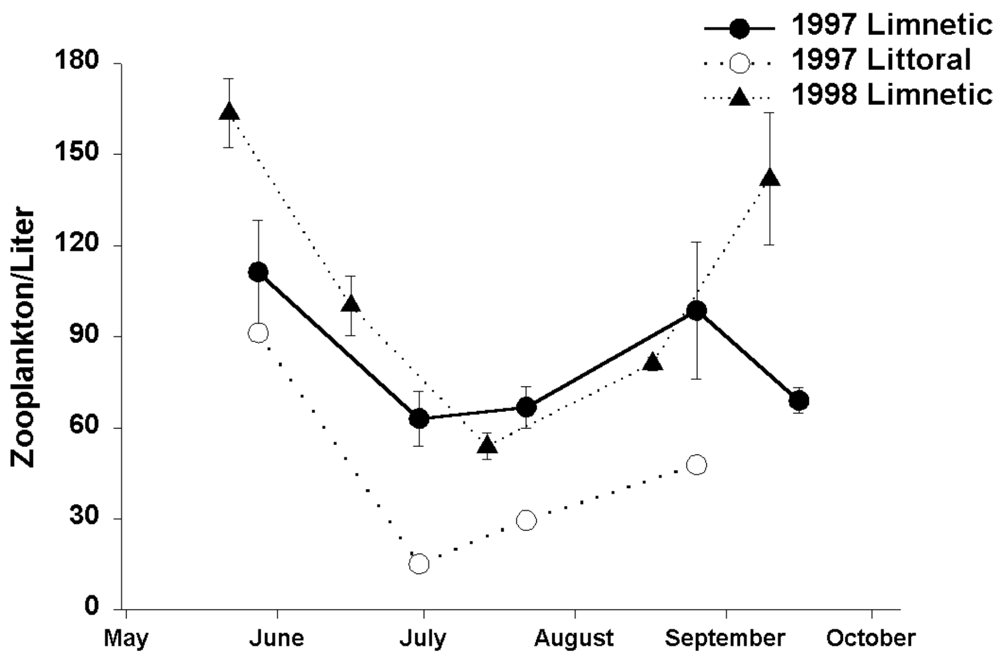
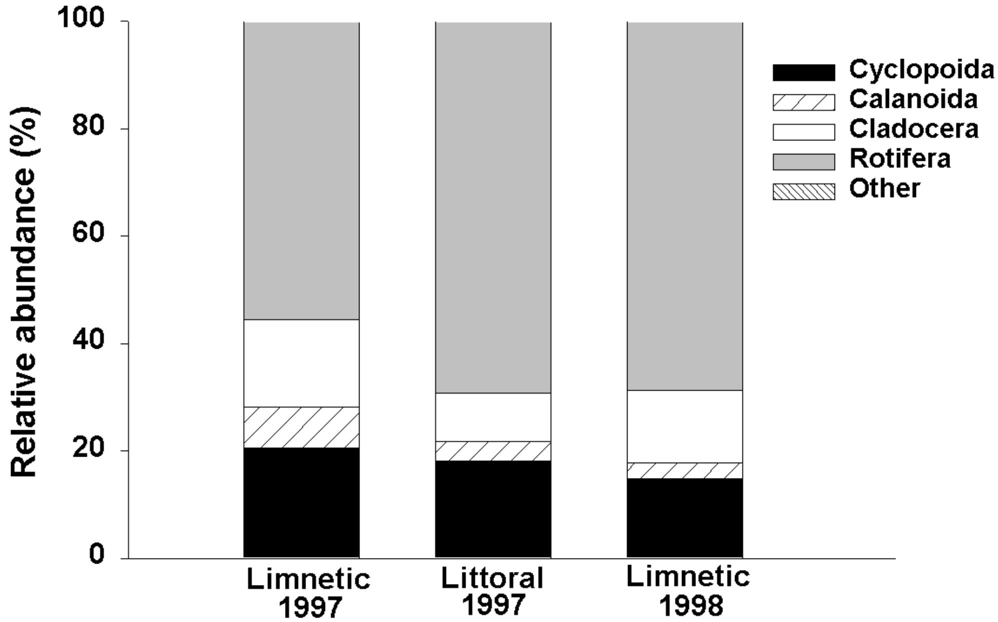


Figure 9-14. Relative abundance of zooplankton groups in Sargent Lake calculated from averages of each year.



Siskiwit Lake

Zooplankton densities were lower in Siskiwit Lake than all other lakes studied. This is probably because of the low primary productivity of Siskiwit, i.e., the zooplankton are being controlled by bottom-up influences. Summer mean densities in the limnetic zone were 22 ± 4 individuals/liter in 1997 (range: 8-31 individuals/liter) and only 17 ± 3 individuals/liter in 1998 (range:12-30 individuals/liter). Peak zooplankton densities occurred in late June to early July in both 1997 and 1998 samples (Figure 9-15). Specific taxa were not responsible for the dynamics here. Instead, the number of nauplii was often 30-40% of the total individuals in samples. The changes in density within each year were generally due to changes across numerous taxa. Littoral densities remained below 10 individuals/liter throughout the sampling period in 1997.

Although densities were low, we still encountered 30 different taxa of zooplankton. The Margalef's index was 4.7 and the Shannon-Wiener index was 2.4-2.5, which was similar to Sargent Lake and high compared with other lakes of the study. Dominance of a single taxa is spurious when abundance is so low. For example, in a sample with 1.5 individuals/liter of a certain species, it could constitute 50-60% of the taxa. Thus, caution has to be used when interpreting the community composition data. Unlike other lakes in the study that were completely dominated by rotifers, the relative taxa abundance was fairly evenly distributed among the four main zooplankton groups in Siskiwit (Figure 9-16). *Daphnia galeata mendotae*, a species common in glacial lakes (Hutchinson 1967), was collected. In addition, numerous taxa common to large, cold-water lakes in the north were found, including *Skistodiaptomus oregonensis* (Hutchinson 1967). The cladoceran *Holopedium gibberum* was present in extremely low numbers but indicates a low-calcium lake (Hutchinson 1967). Similar to Sargent Lake, the rotifers *Ploesoma truncatum*, *Polyarthra* spp. and *Synchaeta* spp. were encountered in appreciable numbers in the limnetic zone. Although no taxon clearly dominated the littoral zone, *Diacyclops thomasi*, *S. oregonensis*, and *Bosmina longirostris* (Cladocera) and the rotifers *Keratella* spp. and *Kellicottia longispina* clearly were the most numerous zooplankton in the limnetic zone (Table 9-8). *B. longirostris* and *Tropocyclops prasinus mexicanus* were more common in the littoral than the limnetic, possibly because production in this zone may be high. These species tend to indicate a more eutrophic habitat (Hutchinson 1967), but Siskiwit is still considered an oligotrophic lake. No *Conochilus unicornis* were collected in the littoral zone, whereas they were relatively common in the limnetic. These differences contribute to the low similarity between the littoral and limnetic zones (24-43%). Between-year comparisons of the community revealed the two years to be fairly similar (64%). Although there is some variability in the samples, the low abundance and the high community indices could indicate that intensifying the sampling regime for Siskiwit Lake would not add much information about the lake. Minor changes in the taxa composition can occur without any substantial change to the lake. However, if zooplankton densities increase greatly, it could indicate that something in the lake is changing.

Table 9-8. Zooplankton taxa that amounted to ≥ 10% of the relative abundance in Siskiwit Lake. The 1 July littoral sample also had 11% each of Tropocyclops prasinus mexinaus and Ploesomas truncatums.

Site	Date	*Diacyclops thomasi*	*Skistodiaptomus oregonensis*	*Bosmina longirostris*	*Daphnia galeata mendotae*	*Asplanchna* spp.	*Collotheca* spp.	*Conochilus unicornis*	*Kellicottia longispina*	*Keratella* 1 post. spine	Polyarthra spp.	Synchaeta spp.
Limn	27-May-97	54							14			
Limn	1-Jul-97			17				26	20	10		
Limn	21-Jul-97			22				13	19	13	11	
Limn	25-Aug-97		15		10				12	20		
Limn	15-Sep-97	12		10	25					12		
Limn	21-May-98								77			
Limn	15-Jun-98							38	42			
Limn	13-Jul-98				26			23	25			
Limn	18-Aug-98			12	24			20		18		
Limn	11-Sep-98			12		10		20	15			
Limn	26-May-99		13						40			
Litt	27-May-97			15					38			23
Litt	1-Jul-97								44	11	22	
Litt	21-Jul-97			54						12		12
Litt	25-Aug-97			25		13	25		38			
Litt	15-Sep-97			25		17	33					

Figure 9-15. Total zooplankton densities in Siskiwit Lake. Means ± 1 standard error (error bars) are calculated from triplicate samples.

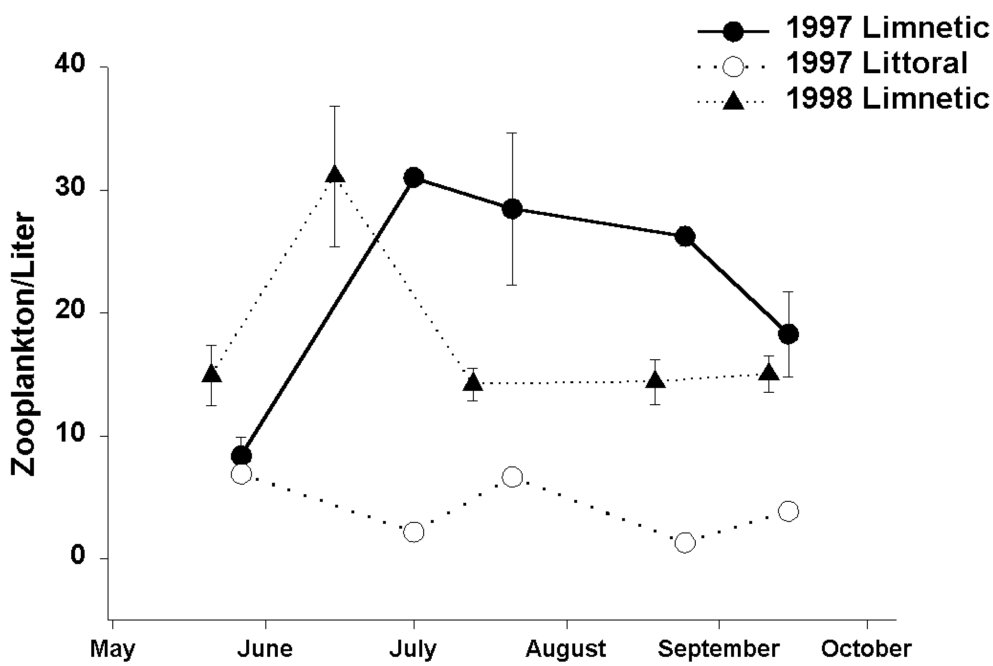
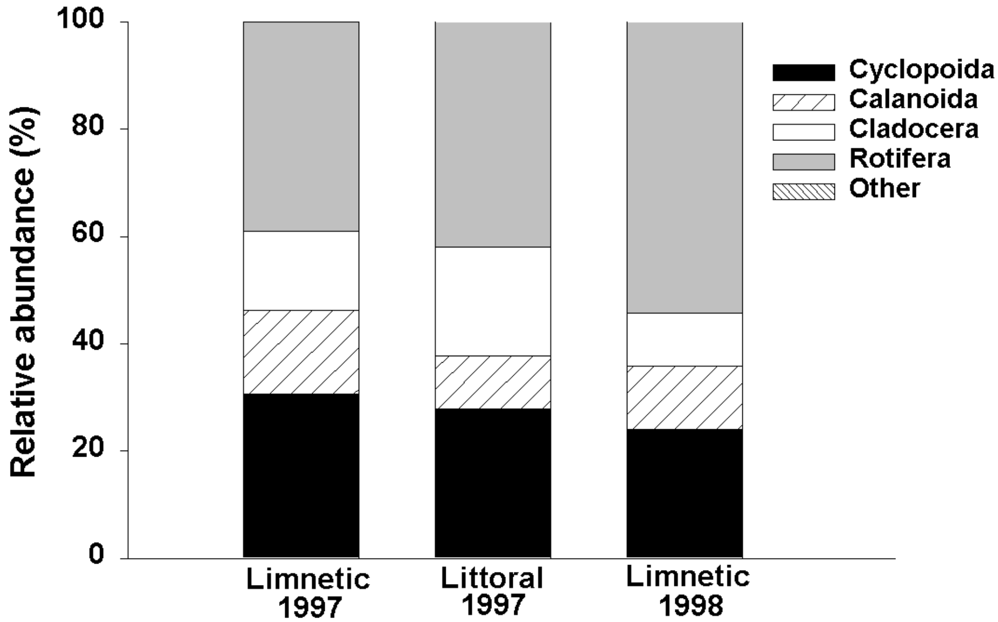


Figure 9-16. Relative abundance of zooplankton groups in Siskiwit Lake calculated from averages of each year.



Locator Lake

Zooplankton density in the limnetic zone was about twice as high in 1998 as in 1997. Mean summer density was 115 ± 41 individuals/liter in 1997 (range: 55-274 individuals/liter), whereas mean summer density was 63 ± 8 individuals/liter in 1998 (range: 36-121 individuals/liter). Both 1997 and 1998 limnetic data showed possible bimodal distribution through the sampling period (Figure 9-17). Likewise, the littoral data from 1997 showed a possible bimodal distribution. The data demonstrate that slight inter-year variation can be expected, especially if samples are taken at different periods of the year. The monthly variation was driven primarily by changes in *Conochilus unicornis* densities, whereas other taxa did not noticeably affect the total zooplankton density because of their relatively low abundance (see below). Locator Lake and Mukooda Lake had very similar patterns of zooplankton abundance. Both lakes showed the highest densities in June and rapid density reductions through the summer.

We encountered 34 total taxa in the lake throughout the study. The Margalef's index was 4.2 and the Shannon-Wiener index was 2.2-2.4. These indices are similar to Mukooda Lake's indices. The taxa composition was 67% similar between years. The zooplankton community was numerically dominated by the rotifers (Figure 9-18). *Conochilus unicornis* was the only dominant taxa, making up between 30-60% of the zooplankton in 1997 and 7-60% in 1998. *Kellicottia longispina* was also present from 1-30% at times during 1997, but was more prominent in 1998 when it made up 7-60% of the zooplankton in samples. Both *C. unicornis* and *K. longispina* are ubiquitous species in the Great Lakes region and beyond. The commonly occurring cyclopoid copepods, *Diacyclops thomasi*, and the cladoceran, *Bosmina longirostris*, were also common, but in abundances far less than the commonly occurring rotifers. *Tropocyclops prasinus mexicanus* was also more common in the littoral than in the limnetic zone, and this species is also commonly collected throughout lakes. *Synchaeta* spp. (Rotifera) was found only in littoral samples but can be found throughout lakes normally. Compared to the limnetic zone, the cladoceran *B. longirostris* and copepod *T. prasinus mexicanus* were more prominent in the littoral zone, at times composing up to 43% of the zooplankton (Table 9-9). *Holopedium gibberum*, a species associated with cold, calcium-poor, acidic waters was present in the lake (Hutchinson 1967), but in low numbers. Based on taxa relative abundances, the littoral and limnetic community had relative similarities from 43 to 62%, indicating some differences between the communities.

Table 9-9. Zooplankton taxa that amounted to ≥ 10% of the relative abundance in Locator Lake.

Site	Date	*Bosmina longirostris*	*Asplanchna* spp.	*Conochilus unicornis*	*Kellicottia bostoniensis*	*Kellicottia longispina*	*Keratella* 1 post. spine	*Ploesoma truncatum*	*Synchaeta* spp.
Limn	5-Jun-97			60		18			
Limn	26-Jun-97			53		30			
Limn	22-Jul-97			33		19	13		
Limn	19-Aug-97			36			18		
Limn	16-Sep-97			15	10		26		
Limn	27-May-98					54	11		
Limn	18-Jun-98			15		50			
Limn	14-Jul-98			57					
Limn	11-Aug-98			21		13	14		
Limn	9-Sep-98			20			23		
Limn	24-May-99			18		41	22		
Litt	5-Jun-97		22						
Litt	26-Jun-97	43		21		15			
Litt	22-Jul-97	13					15	13	
Litt	19-Aug-97	14		22					13
Litt	16-Sep-97						43		21

Figure 9-17. Total zooplankton densities in Locator Lake. Means ± 1 standard error (error bars) are calculated from triplicate samples.

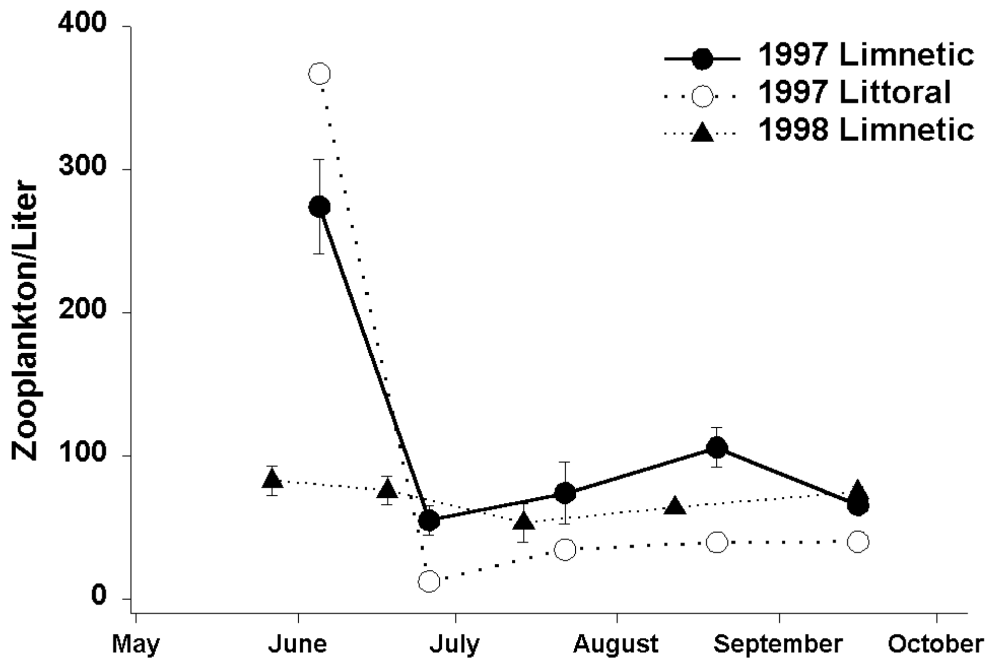
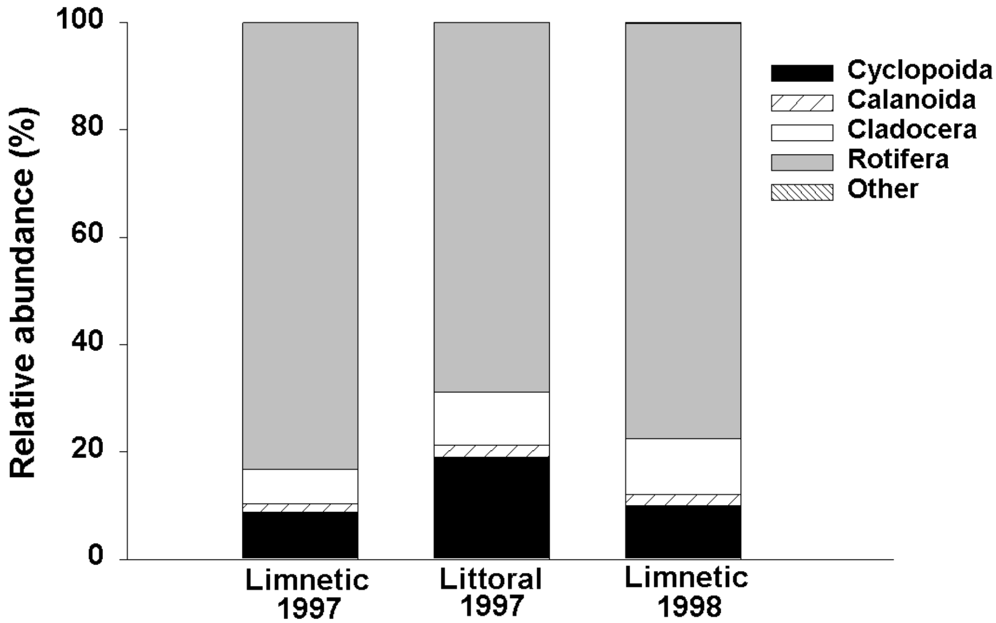


Figure 9-18. Relative abundance of zooplankton groups in Locator Lake calculated from averages of each year.



Mukooda Lake

Zooplankton densities tended to decline in a logarithmic fashion in both 1997 and 1998 in the limnetic zone and in the littoral zone (Figure 9-19). Densities were generally higher in 1997 (summer mean = 187 individuals/liter, SE = 86) than in 1998 (summer mean = 74 individuals/liter, SE = 28). Abundance was more variable in 1997 (60-512 individuals/liter) than in 1998 (range: 24-173 individuals/liter). The higher densities in the spring reflected the abundance of a few taxa. The most important of these were *Asplanchna*, *Kellicottia* and *Keratella*. All three taxa were extremely high on the earliest sampling date, then decreased in abundance (often to zero) throughout summer. In general, the decline in abundance was representative for most taxa, with the possible exception of *Conochilus unicornis*. *C. unicornis* was also a numerous zooplankton, but it reached its peak abundance in the late June samples.

In total, we encountered 37 different taxa in Mukooda Lake during the study. The Margalef's index was 4.2 and the Shannon-Wiener index was 2.2-2.4. As in most other lakes, the rotifers were the numerically dominant taxon group, although the calanoid copepods were more abundant (relative to other taxa) in Mukooda Lake than other lakes studied (Figure 9-20). The commonly occurring rotifers *Asplanchna* spp., *C. unicornis*, and *Kellicottia longispina* were dominant taxa in the limnetic zone, whereas *Collotheca* spp. was a common rotifer in the littoral zone (Table 9-10). The littoral zone was quite different from the limnetic zone in community composition. At times the similarity between the two zones was as low as 24% and only showed a maximum similarity of 43% (lowest of all the lakes studied). Obvious differences in the taxa included abundant *Collotheca* spp. in the littoral zone but not in the limnetic zone. The dominant taxa in the limnetic zone mentioned above were collected in the littoral zone but not in the high abundances. *Bosmina longirostris* was a common species in most of the other lakes studied, but it was not very abundant in Mukooda and Locator lakes. This species is generally an indicator of a lake in transition to a more eutrophic state (Hutchinson 1967), so its absence from Voyageurs lakes could indicate that the lakes are not changing trophic status.

In general, many taxa common to large, cold-water lakes were encountered. For example, *Daphnia galeata mendontae* is common in glacier-formed lakes, and the copepods *Leptodiaptomus minutus* and *Limnocalanus macrurus* are common species found in northern latitudes (Hutchinson 1967). *Holopedium gibberum*, which was collected often in samples, is a species that indicates an acidic, low-calcium lake (Hutchinson 1967). Although lakes in Voyageurs do tend to be low-calcium lakes (see water chemistry section), the occurrence of *H. gibberum* is somewhat surprising because Mukooda Lake is more alkaline than acidic, as is Locator Lake. But other acidophilic species were found, such as *Trichocerca* spp. and *Lecane* spp. (Hutchinson 1967). Thus, it appears that the zooplankton data should be interpreted along with other data, such as water chemistry, in further monitoring programs because there are some inconsistencies in the tolerances of some taxa. There were differences between years, particularly in magnitude of zooplankton abundance.

Table 9-10. Zooplankton taxa that amounted to ≥ 10% of the relative abundance in Mukooda Lake.

Site	Date	*Diacyclops thomasi*	*Bosmina longirostris*	*Asplanchna* spp.	*Collotheca* spp.	*Conochilus unicornis*	*Kellicottia longispina*	*Keratella quadrata*	*Keratella* 1 post. spine	*Ploesoma truncatum*	*Polyarthra* spp.	Trichocerca *cylindrica/ elongata*
Limn	4-Jun-97			25		11	33	22				
Limn	25-Jun-97			11		67						
Limn	23-Jul-97					57		17				
Limn	18-Aug-97				34	15		16				11
Limn	15-Sep-97	12			36	29						
Limn	26-May-98		11	18		34	24					
Limn	17-Jun-98			28		33	22					
Limn	13-Jul-98					23	41		18			
Limn	12-Aug-98				29	22			12			
Limn	8-Sep-98	18				17	14					
Limn	25-May-99					17	34		14			
Litt	4-Jun-97		66				10				15	
Litt	25-Jun-97					32	50					
Litt	23-Jul-97						21	14		29		
Litt	18-Aug-97				78							10
Litt	15-Sep-97				88							

Figure 9-19. Total zooplankton densities in Mukooda Lake. Means ± 1 standard error (error bars) are calculated from triplicate samples.

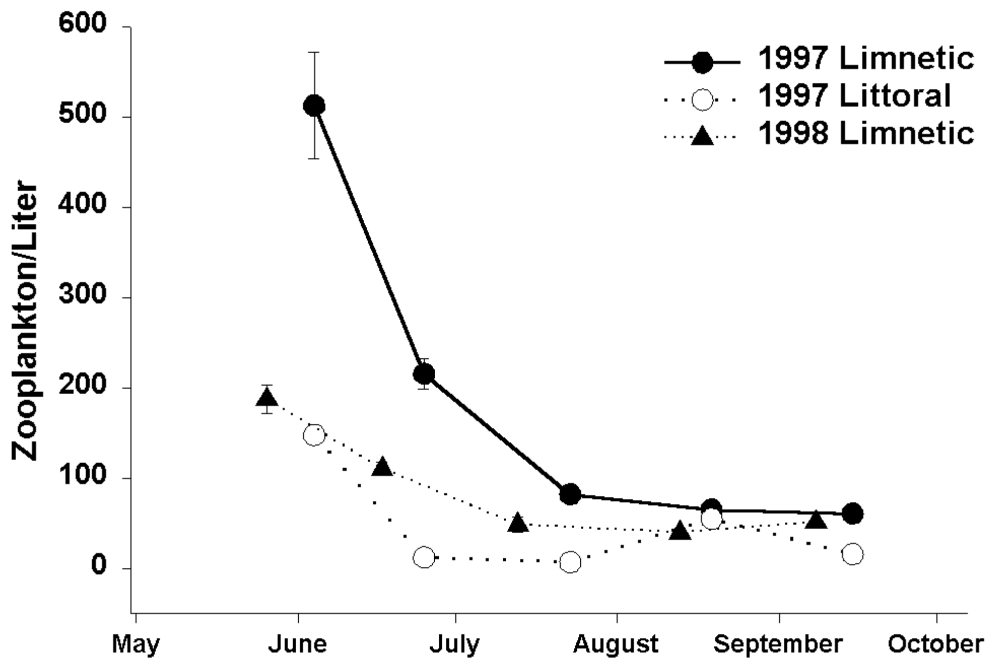
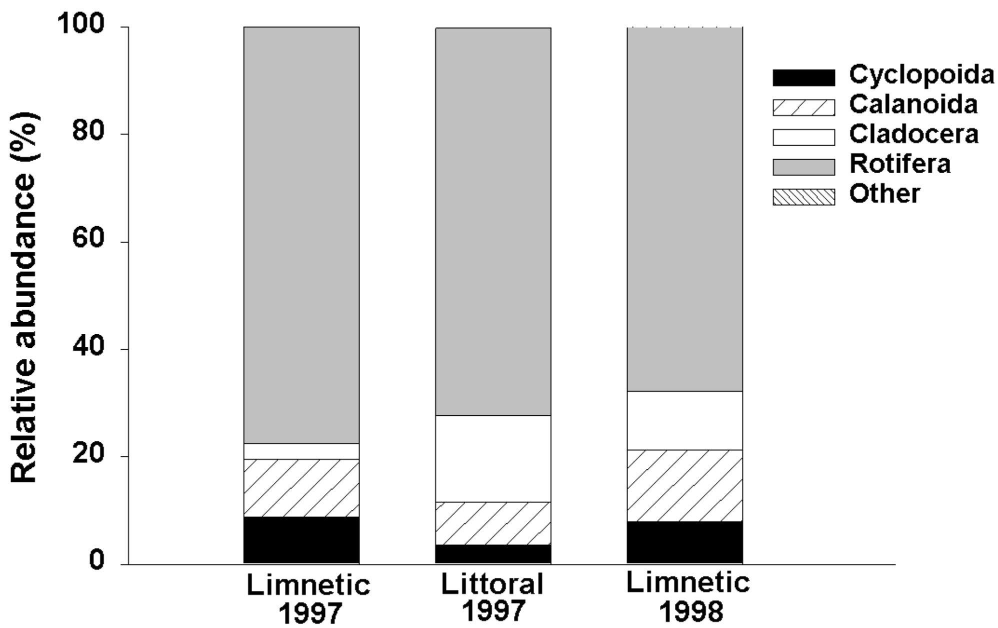


Figure 9-20. Relative abundance of zooplankton groups in Mukooda Lake calculated from averages of each year.



Chapter 10 Benthic Macroinvertebrates Results

Sleeping Bear Dunes National Lakeshore

1997

The total invertebrate density in Loon Lake was significantly higher ($p = 0.042$) in the limnetic zone than in the littoral zone (Figure 10-1). Despite the higher density, Margalef's index was lower in the limnetic zone compared to the littoral zone (Figure 10-2), and the Shannon-Wiener indexes were similar (Figure 10-3). The community was predominantly *Chaoborus*, which composed 45% of the community, and Tubificid worms, which composed 44% of the community (Figure 10-4). The littoral community was composed mostly of sphaeriid clams (60% of the community). Chironomids were the major insect taxa (13% of the total community), and *Hexagenia* mayflies also were consistently abundant. The Crustacea (8% of community) was made-up of both *Hyalella* and *Gammarus* amphipods, and the Oligochaeta (14% of the community) were mostly Tubificidae.

The total invertebrate density in North Bar Lake was about 2000 m^{-2},and there was no difference ($p = 0.407$) between limnetic and littoral densities (Figure 10-1). Margalef's and Shannon-Wiener indexes were slightly higher in the littoral zone compared to the limnetic zone (Figures 10-2 and 10-3). Insects dominated the communities in both the littoral (76% of the community) and limnetic (70% of the community) zones (Figure 10-5). Chironomids were the most abundant taxon in both zones, constituting over half the insect community. *Chaoborus* also contributed to both zones, although more in the limnetic than the littoral zone. Tubificid worms were the second most abundant taxon in both zones, being 28% of the community in the limnetic zone and 15% of the taxa in the littoral zone.

1998

In Loon Lake, the invertebrate density was significantly higher ($t = 4.87$, $df = 4.8$, $p = 0.005$) in the limnetic zone than in the littoral zone (Figure 10-6). Despite the difference, Margalef's index was twice as high in the littoral zone (Figure 10-7), and the Shannon-Wiener index was similar between the two zones (Figure 10-8). Most of the organisms collected from the littoral zone were insects, which in turn were mostly chironomids. Chironomid density steadily declined from about $1800/m^2$ in May to about $200/m^2$ by September. Greater than 50% of the organisms collected in the limnetic zone were oligochaetes (Figure 10-9). Chironomids and chaoborids were each about 50% of the insects collected, whereas a few ceratopogonids also were present in a few samples (see Appendix 5b).

Mean invertebrate density in North Bar Lake was slightly higher in the limnetic zone than the littoral zone (Figure 10-6), although the difference was not significant ($t = 1.98$, $df = 7.8$, $p = 0.84$). The Margalef's index was nearly three times higher in the littoral zone than in the limnetic zone (Figure 10-7), whereas the Shannon-Wiener index was just over half as high (Figure 10-8). The majority of the invertebrates collected from the littoral sediments were oligochaetes (Figure 10-10). Other organisms collected were sparse and not consistently found in samples, with the exception of chironomids and chaoborids. In general, density of all organisms was low, but some present taxa were noteworthy, such as the dragonflies (Odonata: Gomphidae and Corduliidae) and beetles (Coleoptera: *Berosus* sp.).

The invertebrate density in Round Lake was significantly higher (t = 9.07, df = 5.5, p<0.001) in the littoral zone than in the limnetic zone (Figure 10-6). Margalef's index was about three times higher in the littoral samples than in the littoral samples (Figure 10-2), and the Shannon-Wiener index was about twice as high (Figure 10-8). The mollusks were very evident in the littoral zone, with snails (Gastropoda) being the dominant benthic invertebrate (Figure 10-5). *Valvata* sp. and *Amnicola* sp. were consistently abundant throughout the sampling (see Appendix 5b). Of the insects collected, the chironomids were most prevalent, although mayflies (Ephemeroptera; mostly *Caenis* sp.) were also occasionally present in high numbers. The limnetic invertebrates were all Diptera, composed of chaoborids, chironomids, and ceratopogonids (in order of decreasing abundance).

Figure 10-1. Macroinvertebrate densities for the sampled lakes in 1997. Shared letters (capital for limnetic and small case for littoral) indicate no significant differences. Limnetic and littoral samples are not compared.

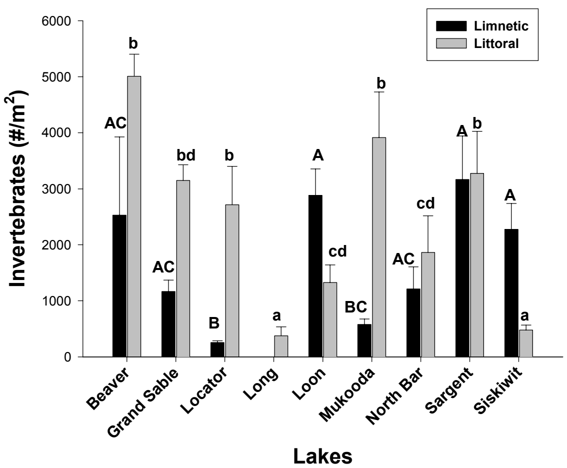


Figure 10-2. Margalef's index for the sampled lakes. This richness index was calculated by pooling all data from 1997.

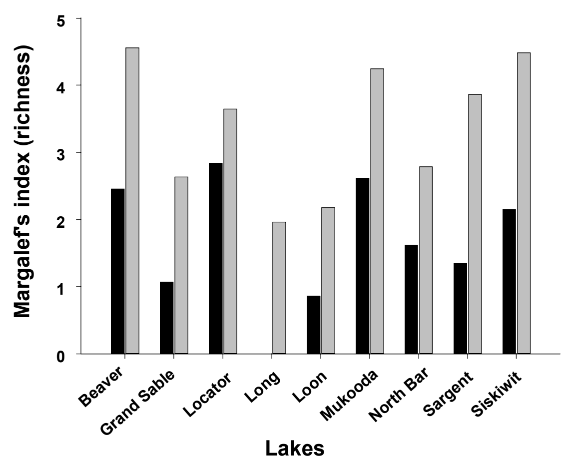


Figure 10-3. Shannon-Wiener index for sampled lakes. This index was calculated by pooling all data from 1997.

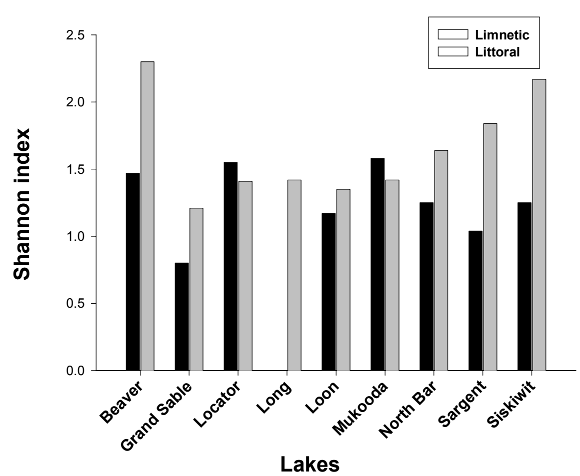


Figure 10-4. Relative abundance of macroinvertebrates from Loon Lake. Relative abundance was calculated by pooling all data from 1997.

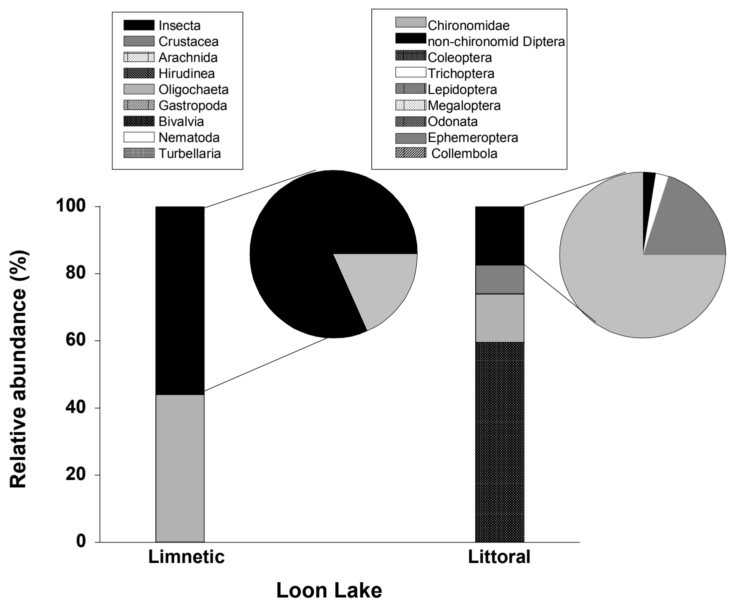


Figure 10-5. Relative abundance of macroinvertebrates from North Bar Lake. Relative abundance was calculated by pooling all data from 1997.

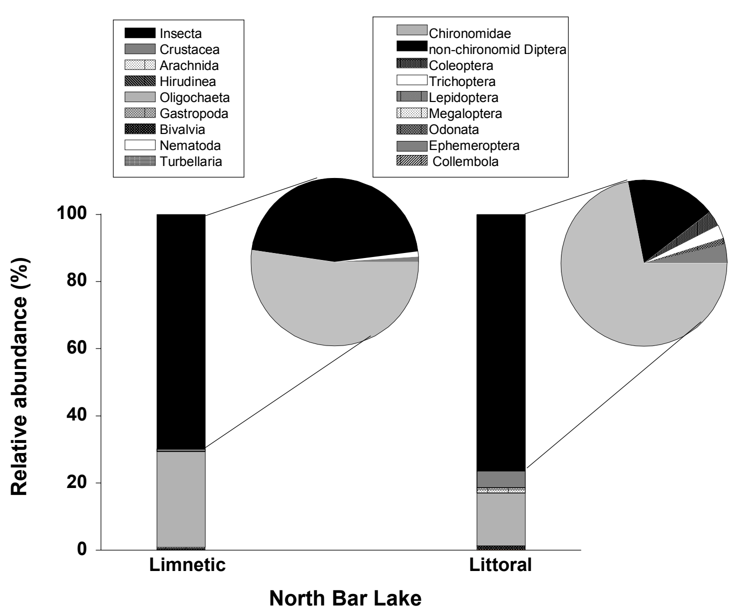


Figure 10-6. Macroinvertebrate densities for the sampled lakes in 1998. Shared letters (capital for limnetic and small case for littoral) indicate no significant differences. Limnetic and littoral samples are not compared.

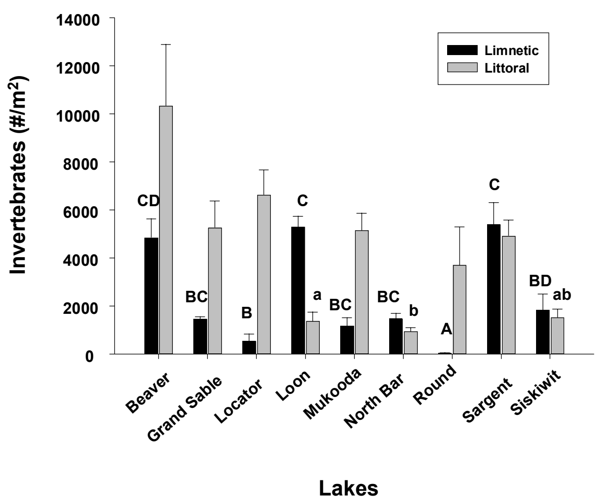


Figure 10-7. Margalef's index for the sampled lakes. This richness index was calculated by pooling all data from 1998.

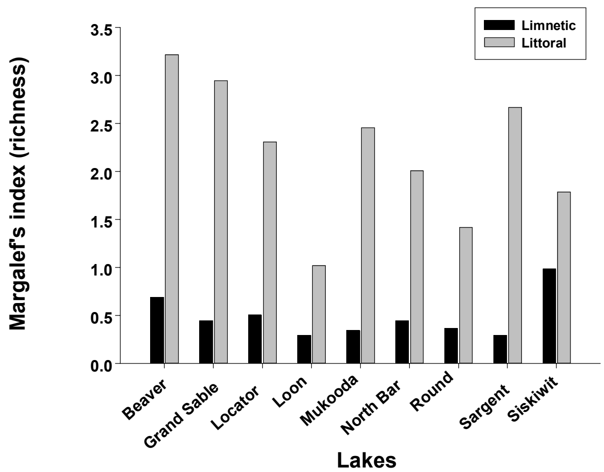


Figure 10-8. Shannon-Wiener index for sampled lakes. This index was calculated by pooling all data from 1998.

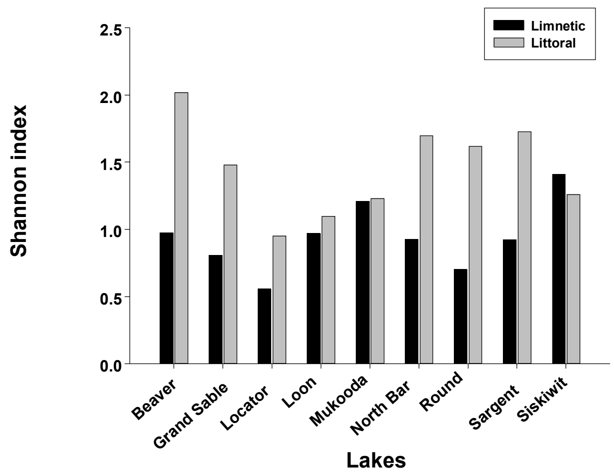


Figure 10-9. Relative abundance of macroinvertebrates from Loon Lake. Relative abundance was calculated by pooling all data from 1998.

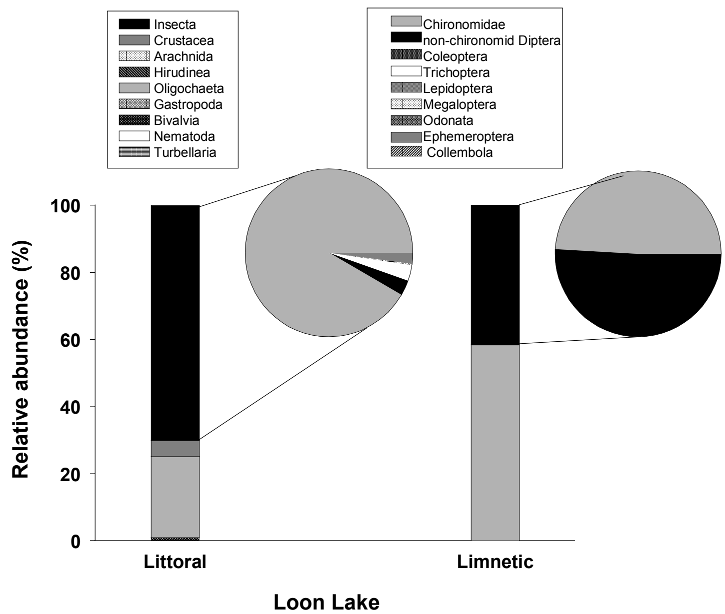


Figure 10-10. Relative abundance of macroinvertebrates from North Bar Lake. Relative abundance was calculated by pooling all data from 1998.

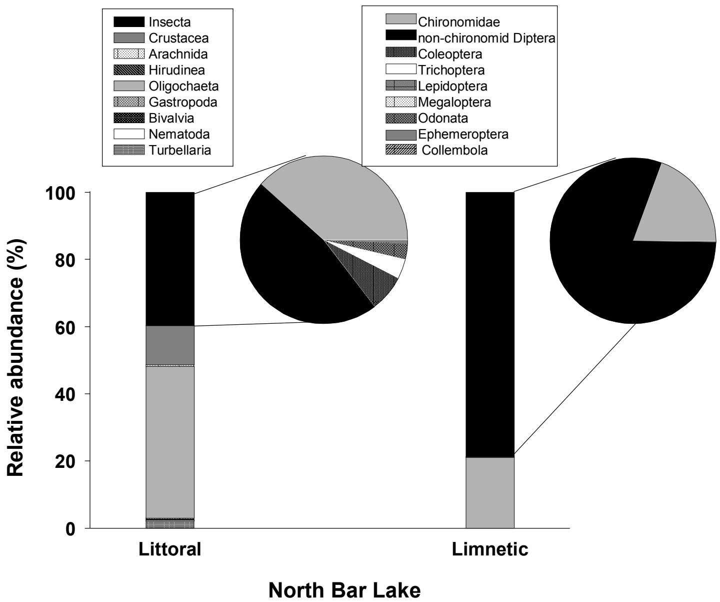
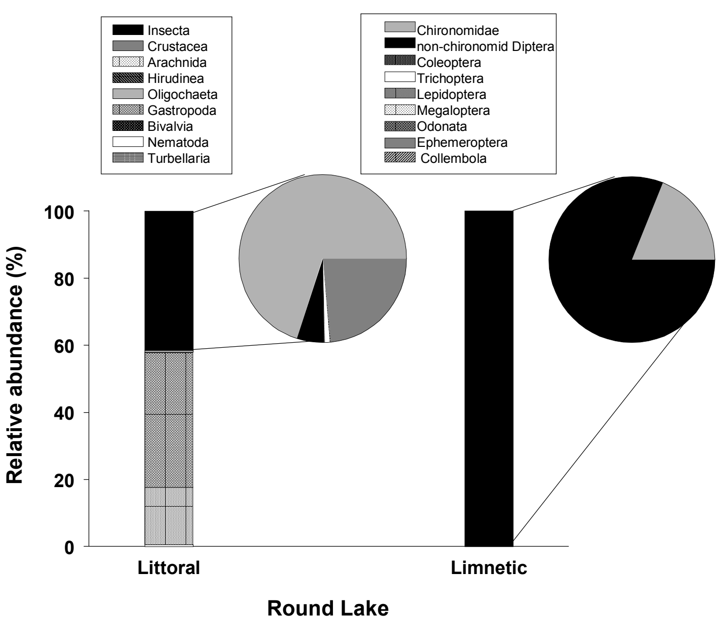


Figure 10-11. Relative abundance of macroinvertebrates from Round Lake. Relative abundance was calculated by pooling all data from 1998.



Pictured Rocks National Lakeshore

1997

Total abundance of invertebrates in the littoral zone of Beaver Lake was 5000 ± 395 individuals m^{-2}. This was significantly higher than the abundance in the limnetic zone ($p = 0.055$; marginally significant), which was 2500 ± 1395 individuals m^{-2} (Figure 10-1). Margalef's index was about two times higher in the littoral zone compared to the limnetic zone in both Beaver and Grand Sable lakes (Figure 10-2). Shannon-Wiener index also was higher in the littoral zones than in the limnetic zones (Figure 10-3), indicating higher diversity. Insects dominated the invertebrate community in the littoral zone of Beaver Lake (Figure 10-12). The high percent of amphipods detected was due to a single sample, taken in September, when a large number (4800 m^{-2}) of *Hyalella azteca* was collected. On the other dates sampled *H. azteca* was either present in low numbers or absent, whereas the limnetic zone samples had consistently high numbers of *H. azteca*. This difference may indicate a sampling anomaly, although no other taxa demonstrate an obvious difference. The insects were, in turn, composed mostly of Chironomidae (32% of the total community). *Chironomus* spp. and the predacious midge *Procladius* spp. were the most common chironomids encountered (see Appendix 5b).

In Beaver Lake's limnetic zone, insects were again the dominant class (Figure 10-12). Chironomids composed only 13% of the total sample. Surprisingly, *Dubiraphia* spp. beetles were present in high numbers (17% of total sample). *Dubiraphia* are riffle beetles and generally are found in cool, well-oxygenated waters such as streams or along wave-influenced lakeshores (Merritt and Cummins 1996). Their presence in the limnetic zone may be explained by the polymictic nature of Beaver Lake, which insured high dissolved oxygen concentrations in the deeper waters. This frequent mixing of the water column also may have allowed the high diversity of Trichoptera, such as *Molanna* spp., *Polycentropus* spp., *Trianodes* spp., and *Setodes* spp., which are generally intolerant of low-oxygen conditions.

Organism abundance in Grand Sable Lake was significantly higher in the littoral zone zone (3140 ± 280 individuals m^{-2}) than in the limnetic zone (1160 ± 200 individuals m^{-2}; Figure 10-1). The benthic community in Grand Sable Lake was mostly chironomid larvae, which composed over 70% of the total invertebrates from both littoral and limnetic samples (Figure 10-13). The littoral samples were over two times higher for Margalef's index than the limnetic samples (Figure 10-2) and the Shannon-Wiener index was also much higher in the littoral (Figure 10-3). For example, the Trichoptera contributed six genera in the littoral zone compared with no Trichoptera in the limnetic zone, and Ephemeroptera also were found only in the littoral zone (Appendix 5a).

1998

Total abundance of invertebrates in Beaver Lake was 10337 ± 2557 individuals m^{-2} in the littoral zone and 4850 ± 778 individuals m^{-2} in the limnetic zone (see Figure 10-6). Density was significantly higher in the littoral than in the limnetic zone ($t = 2.57$, $df = 7.5$, $p = 0.035$). Margalef's index was over five times higher in the littoral zone than in the limnetic zone (Figure 10-2), and Shannon-Wiener was twice as high in the littoral zone (Figure 10-8). In the littoral zone, insects and crustaceans dominated the invertebrate fauna (Figure 10-14). The crustaceans were composed primarily of the amphipod, *Hyalella azteca*. *H. azteca* density increased steadily from 1400 individuals m^{-2} in May to over 6800 m^{-2} in September. The insect fauna was quite diverse; we observed at least 19 genera, not including chironomid diversity (see Appendix 5b). The most abundant insect was the riffle beetle, *Dubiraphia* sp. *Molanna* sp. (Trichoptera) also was particularly abundant in the littoral sediments. The mollusks were well represented by amnicolid snails and fingernail clams (Sphaeriidae; mostly *Pisidium* sp.).

The limnetic zone invertebrate fauna, again, was much less diverse than the littoral zone, and was composed mostly of chironomid larvae (Figure 10-14). The only other organisms present in detectable numbers were the fingernail clams (*Pisidium*, *Musculium*, and *Sphaerium* spp.), oligochaete worms, and a few chaoborids.

Invertebrate abundance was significantly greater in the littoral zone, 5268 ± 1101 individuals m^{-2}, than in the limnetic zone, 1462 ± 93 individuals m^{-2} ($t = 5.12$, $df = 4.7$, $p = 0.004$; Figure 10-6). Margalef's index was nearly five times higher for the littoral samples than the limnetic samples (Figure 10-7), and Shannon-Wiener index also was notably higher for the littoral zone (Figure 10-8). Chironomid larvae dominated both the littoral and limnetic fauna (Figure 10-15). In the littoral zone, crustaceans were common and composed primarily of the amphipod *Hyalella azteca*. Oligochaetes also were present in appreciable numbers. Three families of mayfly (Ephemeroptera) were present, but none were consistent across the sampling dates. The caddisfly (Trichoptera) *Phylocentropus* sp. was found fairly consistently, and an additional five families were encountered. Only *Chaoborus* sp. and oligochaete worms were observed apart from the chironomids in the limnetic zone.

Figure 10-12. Relative abundance of macroinvertebrates from Beaver Lake. Relative abundance was calculated by pooling all data from 1997.

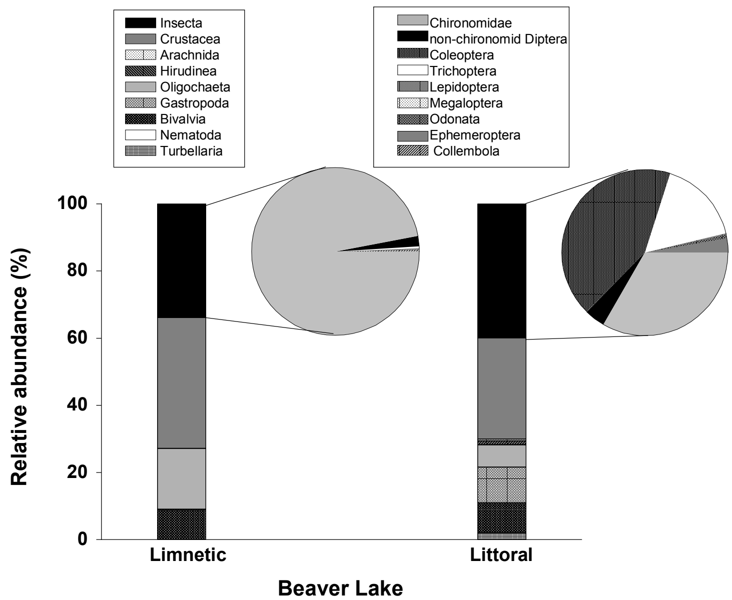


Figure 10-13. Relative abundance of macroinvertebrates from Grand Sable Lake. Relative abundance was calculated by pooling all data from 1997.

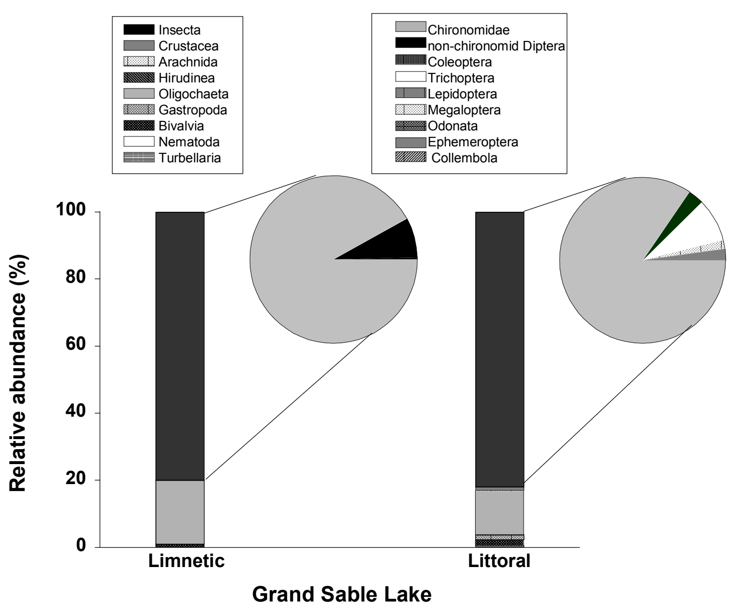


Figure 10-14. Relative abundance of macroinvertebrates from Beaver Lake. Relative abundance was calculated by pooling all data from 1998.

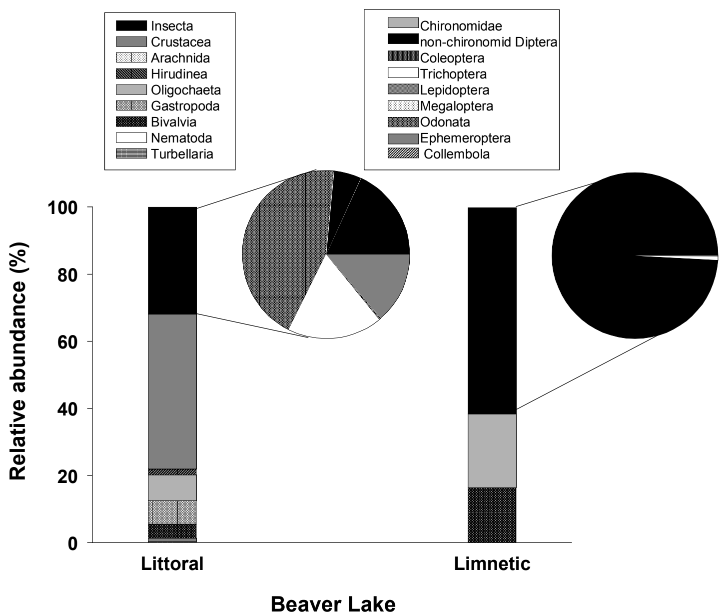
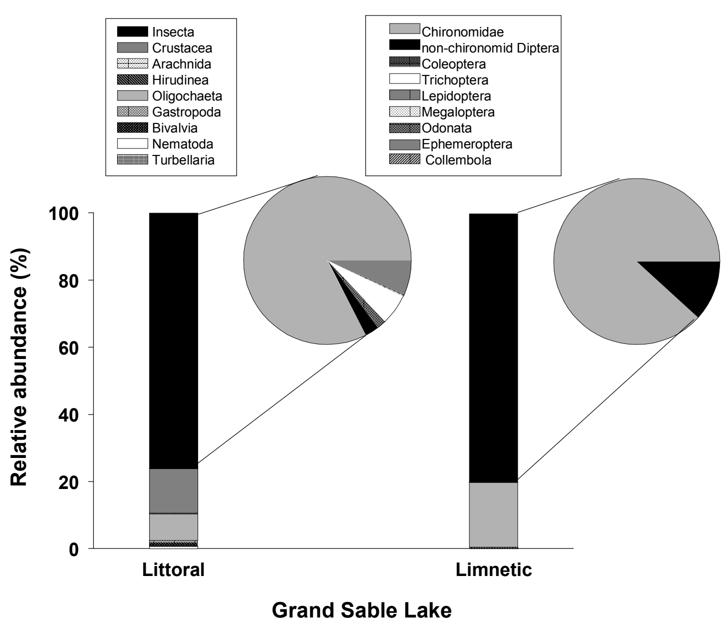


Figure 10-15. Relative abundance of macroinvertebrates from Grand Sable Lake. Relative abundance was calculated by pooling all data from 1998.



Isle Royale National Park

1997

Total invertebrate density in Sargent Lake was about 3000 m^{-2} in both the littoral and limnetic zones (no significant difference; $p = 0.779$; Figure 10-1). Margalef's index and the Shannon-Wiener index, however, were much higher in the littoral zone than in the limnetic zone (Figures 10-2 and 10-3). *Chironomus* and *Chaoborus* dominated the limnetic community (46% and 41%), whereas oligochaetes composed only 11% of the total community (Figure 10-16). In the taxa rich littoral zone, chironomids composed 47% of the total community, but other taxa were present. For example, *Sialis* spp. and five genera of Trichoptera were observed, and ephemerid mayflies were 20% of the total community. Sphaeriid clams and snails were also present.

Siskiwit Lake was much different than the other lakes in this study. For one, the density of total invertebrates was much lower (about 4-5 times) in the littoral zone than in the limnetic zone ($p = 0.015$; Figure 10-1). Although the Margalef's (Figure 10-2) and Shannon-Weiner indexes (Figure 10-3) were notably higher in the littoral than in the limnetic, as in other lakes, the community composition was unique. The deepwater amphipod, *Diporeia*, dominated the benthic community in the limnetic zone (64% of the community; Figure 10-17). Also present in high numbers were sphaeriid clams at 17% of the community. Insects, almost all *Rheotanytarsus* and *Chaoborus*, were only 8% of the total community. In contrast, insects dominated the littoral community (62% of the community), although only 37% of the community was chironomids. Other insect taxa—mayflies (1 genus), dragonflies (2 genera), and caddisflies (2 genera)—were observed, but they were present in only one or two samples, i.e., they were temporally patchy. Amnicolid snails were consistently present in densities near 50 m^{-2}.

1998

As in 1997, there was no significant difference ($t = 0.38$, $df = 7.8$, $p = 0.72$) in invertebrate density between the limnetic and littoral zones (Figure X.6) in Sargent Lake. Again, the two indexes (Margalef's and Shannon-Wiener) were much higher in the littoral zone than in the limnetic zone (Figures 10-7 and 10-8). Many different taxa were represented in the littoral sediments (Figure 10-18). Chironomids were the dominant insect collected, but the other aquatic orders were present, including the riffle beetles, *Dubiraphia* sp. and *Macronychus* sp., and *Hexagenia* sp., the burrowing mayfly. The bivalves were mostly *Pisidium* sp. (Sphaeriidae). The midges (Diptera: Chironomidae and Chaoboridae) were consistently very abundant in the limnetic sediments. The only other invertebrates collected were oligochaete worms (Figure 10-18).

Siskiwit Lake also had no significant difference ($t = 0.68$, $df = 4.3$, $p = 0.53$) in invertebrate density between the two zones (Figure 10-6). Margalef's index was notably higher in the littoral zone than in the limnetic zone (Figure 10-7), whereas the Shannon-Wiener index was slightly higher in the limnetic zone (Figure 10-8). Although dominated by chironomids, the littoral invertebrates included many different orders of insects, the dominant class in these sediments (Figure 10-19). As in 1997, the deep-water taxa included the amphipod *Diporeia* sp. and sphaeriid clams (mostly *Pisidium* sp.), and occasionally *Mysis relicta*. The taxa were, however, dominated by the midges, Chaoboridae and Chironomidae.

Figure 10-16. Relative abundance of macroinvertebrates from Sargent Lake. Relative abundance was calculated by pooling all data from 1997.

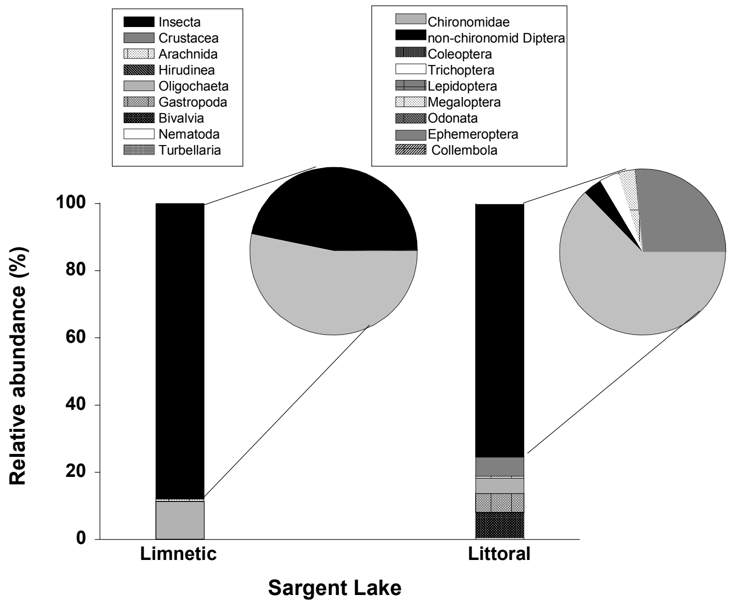


Figure 10-17. Relative abundance of macroinvertebrates from Siskiwit Lake. Relative abundance was calculated by pooling all data from 1997.

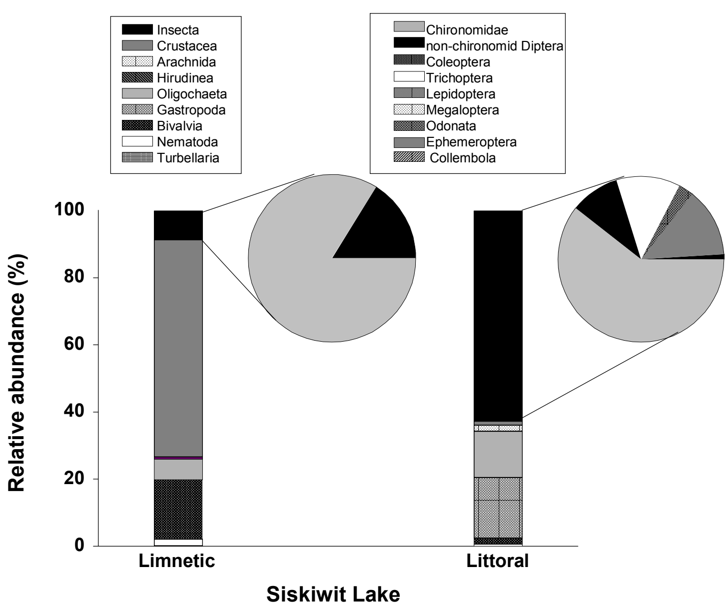


Figure 10-18. Relative abundance of macroinvertebrates from Sargent Lake. Relative abundance was calculated by pooling all data from 1998.

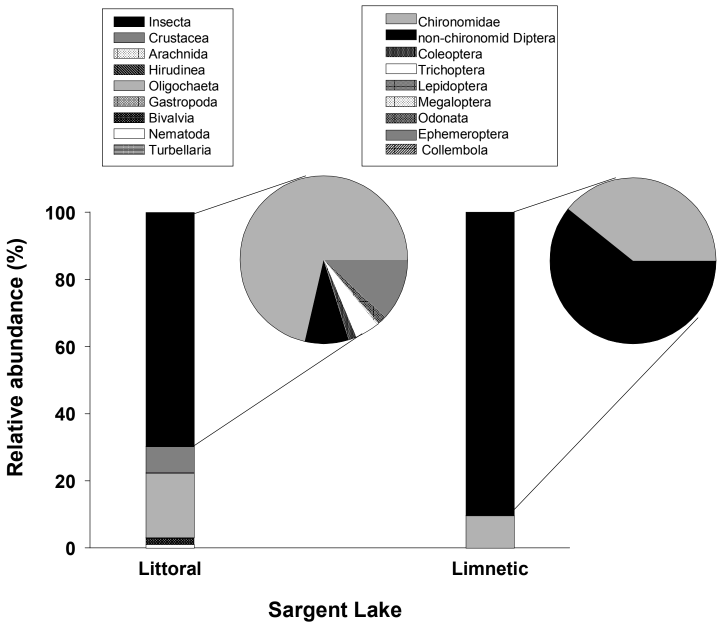
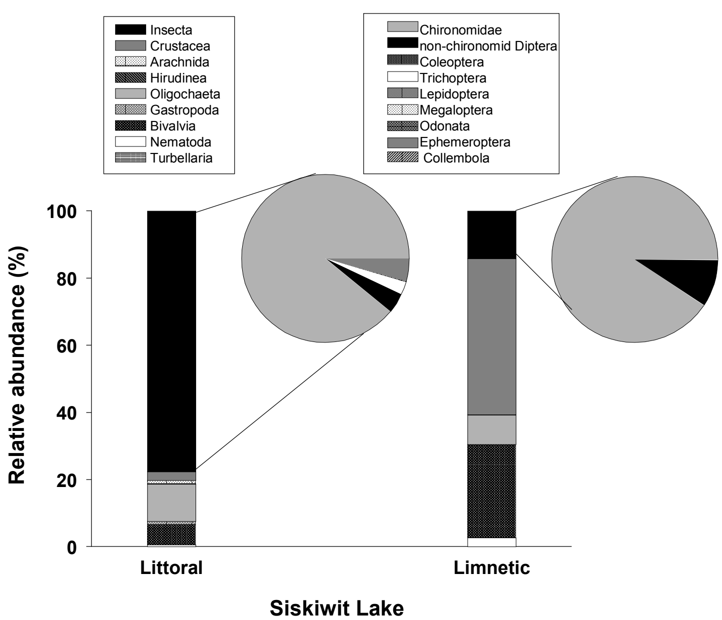


Figure 10-19. Relative abundance of macroinvertebrates from Siskiwit Lake. Relative abundance was calculated by pooling all data from 1998.



Voyageurs National Park

1997

Total invertebrate abundance in Locator Lake was significantly higher ($p = 0.007$) in the littoral zone (2712 ± 683 individuals m^{-2}) than in the limnetic zone (256 ± 27 individuals m^{-2}; Figure 10-1). The invertebrate community in the limnetic zone was dominated by *Chaoborus* spp. (42% of the total community), whereas the chironomids were the most abundant taxa in the littoral zone (69% of the total community; Figure 10-20). Margalef's index was slightly higher in the littoral zone than in the limnetic zone (Figure 10-2), whereas the Shannon-Wiener index was slightly lower in the littoral zone (Figure 10-3). Five genera of Trichoptera were found in the littoral zone, compared to zero found in the limnetic. *Hexagenia* sp. was found in both zones, but the abundance was higher in the littoral zone despite being nearly the same relative abundance. *Ephemera* sp. also occurred only in the littoral, as did the dragonfly larvae, *Epicordulia* sp. and *Tetragoneuria* sp.

In Mukooda Lake, invertebrate abundance was approximately six times higher in the littoral than in the limnetic zone (difference was significant, $p < 0.001$; Figure 10-1). The Shannon-Wiener index was slightly higher in the limnetic zone than in the littoral zone (Figure 10-3). Oligochaetes and chironomids together composed more than 80% of the invertebrates collected in the limnetic zone (Figure 10-21). Although five genera of chironomids were identified, the Margalef's index was lower than in the chironomid-dominated (71% of total community and 15 genera identified) littoral zone (Figure 10-2). The amphipod *Diporeia* was found in both the littoral and limnetic zone, although the density was much higher in the littoral.

1998

The invertebrate abundance in Locator Lake was significantly higher ($t = 5.62$, $df = 4.7$, $p = 0.003$) in the littoral zone (6627 ± 1040 individuals m^{-2}) than in the limnetic zone (543 ± 290 individuals m^{-2}; Figure 10-6). The Shannon-Wiener index was just under two times higher in the littoral zone than in the limnetic zone (Figure 10-8), whereas the Margalef's index was nearly four times higher in the littoral zone (Figure 10-7). Chironomids dominated the littoral taxa, and *Hyalella azteca* also contributed numerically (Figure 10-22). In contrast, the non-chironomid Diptera were most prevalent in the limnetic zone (Figure 10-22). This group was composed entirely of *Chaoborus* sp. larvae. Oligochaetes were numerous in only the September samples (see Appendix 5b).

In Mukooda Lake, the invertebrates were significantly more abundant ($t = 3.53$, $df = 4.7$, $p = 0.018$) in the littoral zone (5148 ± 718 individuals m^{-2}) than in the limnetic zone (1175 ± 335 individuals m^{-2}; Figure 10-6). Although the Margalef's index was much higher in the littoral zone than in the limnetic zone (Figure 10-7), the Shannon-Wiener indexes were nearly identical between the two zones (Figure 10-8). Chironomids were the dominant taxa in the littoral zone (Figure 10-23), despite there being at least 23 different taxa encountered (see Appendix 5b). The amphipod, *Hyalella azteca*, and oligochaetes were well represented in all the samples, but other taxa were not present consistently (see Appendix 5b). In the limnetic zone, the oligochaetes were numerically dominant (Figure 10-23), but chironomid and chaoborid-midge abundance also was consistently high.

Figure 10-20. Relative abundance of macroinvertebrates from Locator Lake. Relative abundance was calculated by pooling all data from 1997.

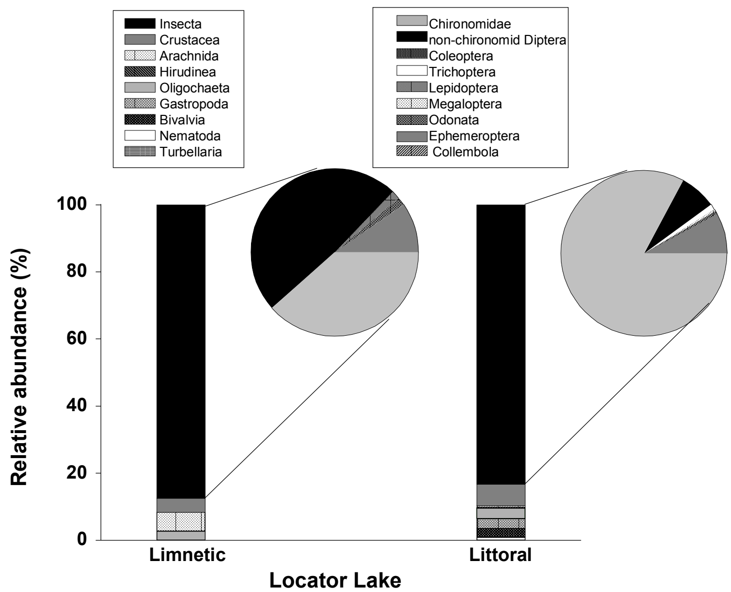


Figure 10-21. Relative abundance of macroinvertebrates from Mukooda Lake. Relative abundance was calculated by pooling all data from 1997.

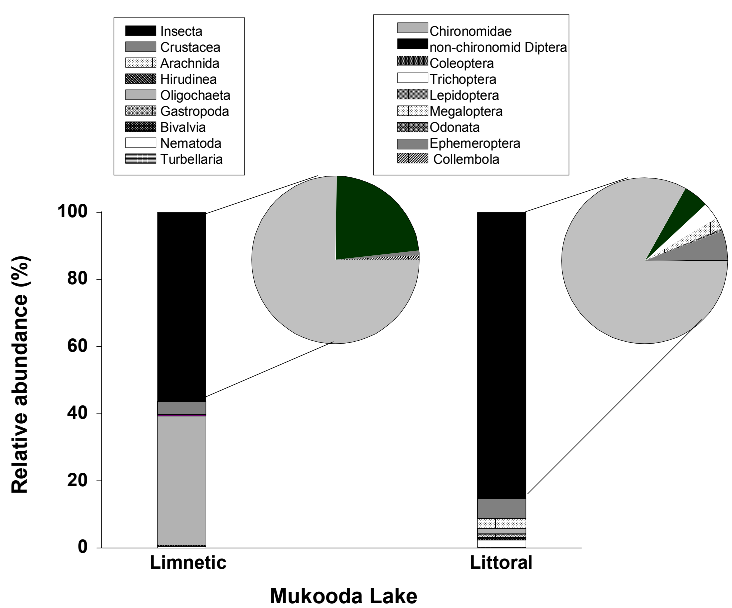


Figure 10-22. Relative abundance of macroinvertebrates from Locator Lake. Relative abundance was calculated by pooling all data from 1998.

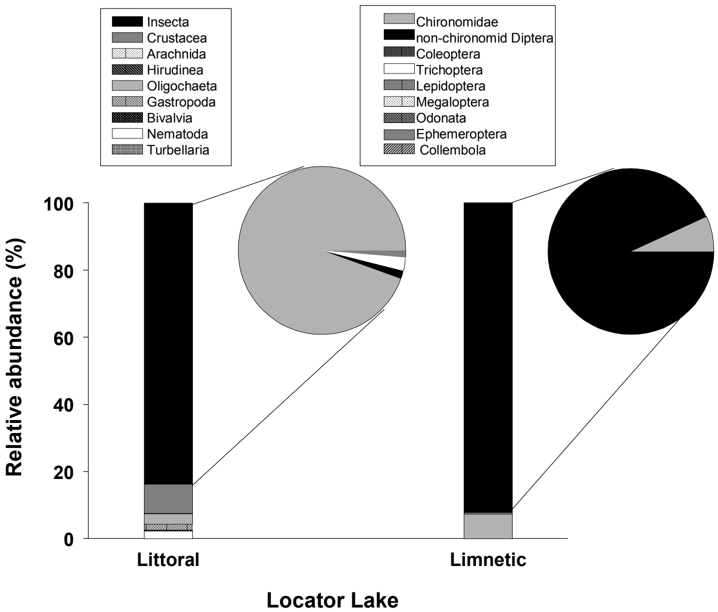
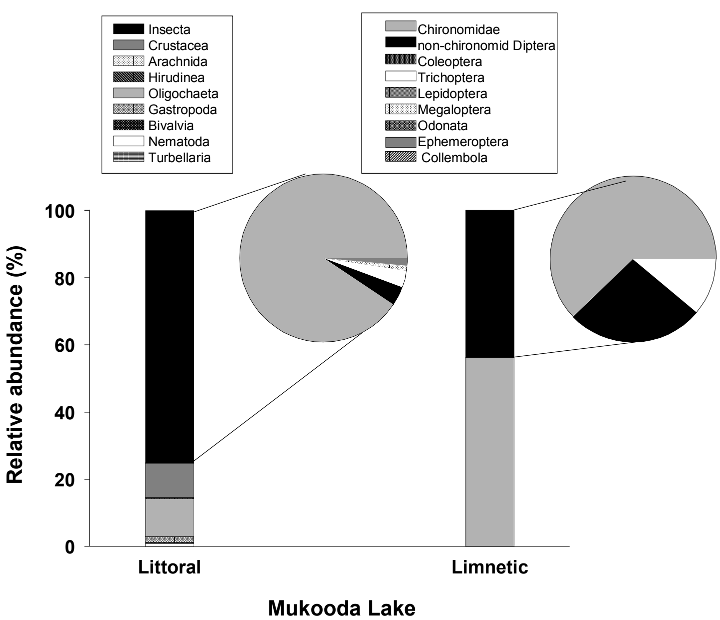


Figure 10-23. Relative abundance of macroinvertebrates from Mukooda Lake. Relative abundance was calculated by pooling all data from 1998.



Comparisons among lakes

Littoral comparisons *1997*

Density of invertebrates in the littoral zones was different among lakes ($F_{8,34} = 13.41$, $p < 0.001$). Densities in Long and Siskiwit lakes were consistently lower compared to the other lakes. Beaver Lake had the highest mean invertebrate density, but it was significantly higher to only a few of the lakes (Figure 10-1). Beaver Lake had the highest Margalef's index, although it was only slightly higher than Mukooda and Siskiwit lakes. Other lakes showing similarities with this index include Locator and Sargent lakes, and Grand Sable and North Bar lakes. Margalef's index in Loon and Long lakes was low compared to the other lakes (Figure 10-2). Beaver Lake also had the highest Shannon-Wiener index, albeit just slightly higher than Siskiwit Lake (Figure 10-3). This diversity index was quite similar among many of the other lakes. Evenness was similar among lakes, with the possible exception of high evenness in Long, Beaver, and Siskiwit lakes, indicating that the abundance of organisms was fairly evenly distributed among the present taxa (Table 10-1).

Littoral comparisons 1998

Density of invertebrates in the littoral zones was different among lakes ($F_{8,36} = 13.05$, $p < 0.001$). Densities in North Bar and Loon lakes were consistently lower compared to the other lakes. Beaver Lake had the highest mean invertebrate density, but it was significantly higher than only a few of the lakes (Figure 10-6). Beaver and Grand Sable lakes had the highest Margalef's indexes of all the lakes (Figure 10-7). Locator, Mukooda, and Sargent lakes were fairly similar, whereas Loon Lake was the lowest of the lakes. Beaver Lake also had the highest Shannon-Wiener index, albeit just slightly higher than North Bar and Sargent lakes (Figure 10-8). Locator Lake had the lowest Shannon-Wiener index. Evenness was similar among lakes, with the possible exception of low evenness in Locator Lake (Table 10-1).

Limnetic comparisons 1997

Density of invertebrates in the limnetic zones was different among lakes ($F_{7,30} = 7.73$, $p < 0.001$). Locator Lake was significantly lower than all other lakes except Mukooda Lake, which itself was significantly lower than Loon, Sargent, and Siskiwit lakes (Figure 10-1). Densities among the other lakes were fairly similar (Figure 10-1). Margalef's index in Loon Lake was lower than in the other lakes, although only slightly lower than in Grand Sable and Sargent lakes (Figure 10-2). Grand Sable Lake had the lowest Shannon-Wiener index, but in general this index was similar across the lakes (Figure 10-3). Evenness was similar across all the lakes (Table 10-1).

Limnetic comparisons 1998

Density of invertebrates in the limnetic zones was significantly different among lakes ($F_{8,36} = 12.42$, $p < 0.001$). Invertebrate density in Round Lake was significantly lower than all other lakes (Figure 10-1). Locator Lake also was significantly lower than most lakes. Densities among the other lakes were fairly similar, although with some exceptions (see Figure 10-1).

Margalef's index was low in all the lakes, generally being less than 1.0 (Figure 10-2). Locator Lake had the lowest Shannon-Wiener index, but it was only slightly lower than Round Lake. Siskiwit and Mukooda lakes had high Shannon-Wiener indexes relative to the other lakes. In general, this index was similar across the other lakes (Figure 10-3). Evenness was similar across all the lakes except in Loon and Mukooda, where it tended to be higher (Table 10-1).

Table 10-1. Benthic macroinvertebrate evenness values for littoral and limnetic zones in 1997 and 1998 for all lakes sampled.

		1997				1998			
		Littoral		Limnetic		Littoral		Limnetic	
Park	Lake	Evenness	SE	Evenness	SE	Evenness	SE	Evenness	SE
INDU	Long	0.907	0.027	-	-	-	-	-	-
SLBE	Loon	0.708	0.055	0.629	0.049	0.506	0.073	0.464	0.059
	North Bar	0.715	0.044	0.690	0.049	0.400	0.049	0.533	0.069
	Round	-	-	-	-	0.361	0.065	0.749	0.107
PIRO	Beaver	0.657	0.049	0.694	0.059	0.253	0.039	0.496	0.063
	Grand Sable	0.664	0.040	0.532	0.043	0.488	0.120	0.561	0.034
ISRO	Sargent	0.783	0.034	0.669	0.082	0.302	0.036	0.488	0.053
	Siskiwit	0.873	0.025	0.556	0.033	0.500	0.124	0.331	0.008
VOYA	Locator	0.746	0.047	0.805	0.083	0.642	0.070	0.700	0.101
	Mukooda	0.755	0.028	0.754	0.035	0.496	0.088	0.570	0.123

Similarity analyses

Based on similarity analyses of the taxa present and their densities, the limnetic samples tended to cluster together and the littoral samples tended to cluster together. This clustering pattern suggests that the limnetic zones were more similar to each other than they were to any of the littoral zones. The one exception was Siskiwit Lake, where the littoral samples were species poor and the total invertebrate density was low relative to the other lakes. Apart from this exception, the limnetic samples clustered likely because they tended to have lower Margalef's index and density than the littoral zones. Similarities among the littoral samples did not reveal any meaningful patterns, except that Locator Lake was not similar to any of the other lakes (Figure 10-24).

For the 1998 samples, the cluster analysis showed again that the limnetic and littoral samples clustered apart from each other, suggesting that the two zones are fundamentally different (Figure 10.25).

Figure 10-24. Cluster analysis based on the benthic invertebrate taxa present in samples from the study lakes in 1997.

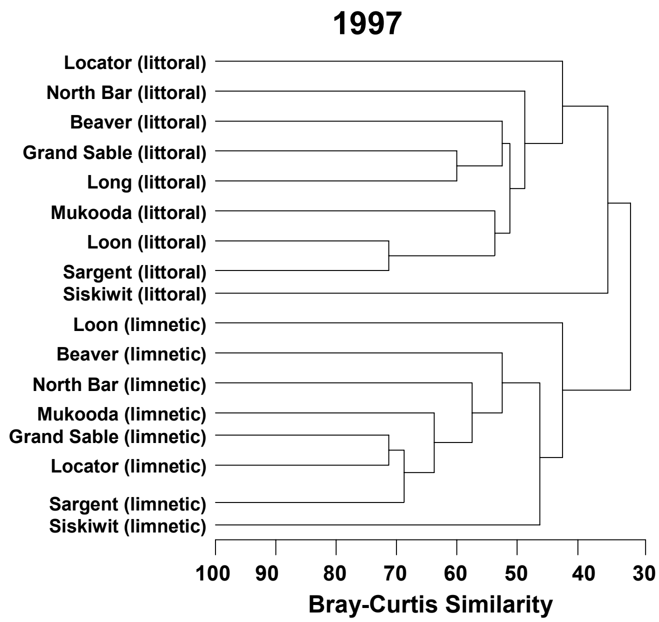
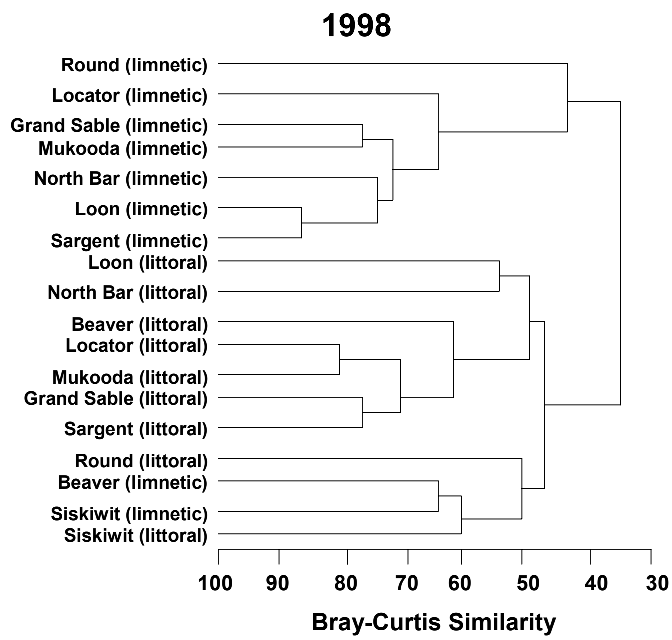


Figure 10-25. Cluster analysis based on the benthic invertebrate taxa present in samples from the study lakes in 1998.



Chapter 11 Macrophytes Results

Long Lake

A total of 12 macrophyte species were collected in Long Lake (Table 11-1). Macrophyte coverage was dense throughout the lake, including almost complete coverage by *Nuphar* sp. and *Nymphaea* sp. It was noted that although the area was not sampled, extensive stands of *Typha* sp. were visible on the southern side of the lake. Most of the species collected were found in both zones. *Eupatorium perfoliatum*, *Potamogeton natans*, and *Potamogeton pectinatus* were only present in the western zone, and *Potamogeton amplifolius* and *Potamogeton zosteriformes* were only present in the eastern zone.

Table 11-1. Macrophytes identified from Long Lake.

Species	Common Name
Brasenia schreberi	Water shield
Eupatorium perfoliatum	Common boneset
Lythrum salicaria	Purple loosestrife
Nuphar advena	Yellow pond lily
Nymphaea odorata	
Pontederia cordata	Pickerel weed
Potamogeton amplifolius	Large leaf pondweed
Potamogeton illinoensis	Pondweed
Potamogeton natans	Common pondweed
Potamogeton pectinatus	Sago pond weed
Potamogeton zosteriformes	Flat-stemmed pondweed
Utricularia purpurea	Purple bladderwort

Loon Lake

Fifteen macrophyte taxa were identified from Loon Lake (Table 11-2). Two species of *Potamogeton*—*Potamogeton angustifolius* and *Potamogeton crispus*—were only found in this lake at Sleeping Bear Dunes. The taxa found, however, were generally present in all Sleeping Bear Dunes lakes. *Decodon verticillatus* was collected only along the western shore, and *Myriophyllum* sp. and *Chara* sp. were the most dominant plants in transects.

Table 11-2. Macrophytes identified from Loon Lake.

Species	Common Name
Ceratophyllum demersum	Coontail
Chara sp.	Stonewort
Family Cyperaceae	Sedge family
Decodon verticillatus	Swamp loosestrife
Juncus sp. 1	Rush
Juncus sp. 2	Rush
Myriophyllum sp.	Water milfoil
Najas flexilis	Bushy pondweed
Nuphar variegata	Yellow water lily
Nymphaea odorata	White water lily
Potamogeton amplifolius	Large leaf pondweed
Potamogeton angustifolius	
Potamogeton crispus	Curly leaf pondweed
Potamogeton pectinatus	Sago pondweed
Potamogeton sp. 3	

North Bar Lake

Plants in North Bar Lake were scattered and sparse, but eight taxa were identified and an additional unknown was collected (Table 11-3). No macrophytes were found in the transect nearest the bar where the sediment is all sand. Only F. Cyperaceae, *Juncus* sp., and *Chara* sp. were found at the northwestern site near the boat launch. *Polygonum natans* had the greatest coverage at the southeastern transect but was not collected at the southwestern site. *Chara* sp. and *Myriophyllum* sp. dominated at that site.

Table 11-3. Macrophytes identified from North Bar Lake.

Species	Common Name
Chara sp.	Stonewort
Family Cyperaceae	Sedge family
Juncus sp. 1	Rush
Myriophyllum sp.	Water milfoil
Nuphar variegata	Yellow water lily
Polygonum natans	Amphibious *Polygonum*/ smartweed
Potamogeton pectinatus	Sago pondweed
Potamogeton sp. 4	
Unknown	

Round Lake

Ten macrophyte taxa were collected from Round Lake, including one unknown (Table 11-4). Macrophyte coverage was sparse, with the greatest coverage along the bog on the southern side. *Nymphaea odorata* and *Potamogeton pectinatus* were the dominant plants along the southern transect. Macrophytes were extremely sparse at the other three sites, with *Chara* sp. occurring most frequently.

Table 11-4. Macrophytes identified from Round Lake.

Species	Common Name
Chara sp.	Stonewort
Juncus sp. 1	Rush
Juncus sp. 2	Rush
Myriophyllum sp.	Water milfoil
Najas flexilis	Bushy pondweed
Nymphaea odorata	White water lily
Potamogeton amplifolius	Large leaf pondweed
Potamogeton pectinatus	Sago pondweed
Potamogeton sp. 4	
Unknown	

Beaver Lake

We identified 16 different taxa of macrophytes in Beaver Lake (Table 11-5). We did not record any species new to Michigan, nor did we observe any non-native macrophytes. *Potamogeton* spp. and the macroalga *Chara* sp. were the most common plants in the lake. Macrophyte cover, estimated for the entire lake's littoral zone, was about 35%. Macrophytes were patchy, and areas of 100% cover were recorded. Often, macrophyte communities were absent or minimal within the first 50 m of shore but more apparent toward the middle of the lake. For example, along the south shore of the section furthest east, the cover was << 1 % within the first 13 m of shore and 100% beyond 13 m. According to an inventory in 1984 (Michigan Natural Features Inventory 1984), *Myriophyllum alterniflorum* was collected in Beaver Lake, but it was not encountered in this study.

In addition to Beaver Lake, we qualitatively surveyed the macrophytes in Little Beaver Lake and the channel connecting the two lakes. A few species were found in Little Beaver Lake that were not found in Beaver Lake (see Table 11-6).

Table 11-5. Taxa list for Beaver Lake macrophytes. Transects T1, T2, T3, and T4 indicate where plants were found. Plants were collected/surveyed on 6 August 2000.

Species	Common Name	Transect
Carex sp.	Sedge	T1, T2, T3
Chara sp.	Muskgrass	T1, T2, T3, T4
Eleocharis compressa/elliptica Sulivan/Kunth	Spike rush	T1
Juncus gerardi Loisel	Rush	T1
Juncus pelocarpus f. *submersus* Fassett	Rush	T2
Myriophyllum exalbescens Fern.	Water milfoil	T3, T4
Najas flexilis (Willd.) Rostk. & Schmidt	Bushy pondweed	T3
Potamogeton amplifolius Tuckerman	Pondweed	T4
Potamogeton gramineus L.	Pondweed	T4
Potamogeton illinoensis Morong	Pondweed	T2
Potamogeton pectinatus L.	Pondweed	T1, T2, T4
Potamogeton perfoliatus L.	Pondweed	T1, T3, T4
Potamogeton praelongus Wulfen	Pondweed	T1, T2
Potamogeton zosteriformis Fern.	Pondweed	T4
Scirpus pungens Vahl.	Bulrush	T1, T3
Vallisneria americana Michaux.	Wild celery	T2, T4

Table 11-6. Taxa list for Little Beaver Lake and channel connecting Beaver Lake macrophytes. Plants were collected/surveyed on 6 August 2000.

Species	Common Name
Elodea canadaensis Michaux.	Waterweed
Equisetum fluviatile L.	Horsetail
Nuphar variegata Durand	Yellow water lily
Potamogeton amplifolius Tuckerman	Pondweed
Potamogeton zosteriformis Fern.	Pondweed
Ranunculus longirostris Godron	Buttercup
Sagittaria latifolia Willd.	Arrowhead
Typha sp.	Cattail
Utricularia purpurea Walter	Bladderwort
Vallisneria americana Michaux.	Wild celery

Grand Sable Lake

In Grand Sable Lake, we identified 21 taxa of macrophytes along four transects. As in Beaver Lake, *Potamogeton* spp. and *Chara* sp. were quite common, and no rare or non-native plants were identified (Table 11-7). The percent cover of macrophytes was about 30% for the entire lake's littoral zone. Again, there were areas entirely void of plants (e.g., shore near the road), and areas where the bottom was completely covered with plants (e.g., west end of lake).

Table 11-7. Taxa list for Grand Sable Lake macrophytes. Transects T1, T2, T3, and T4 indicate where plants were found, and NT indicates plant taxa recorded from areas other than the transects. Plants were collected/surveyed on 7 August 2000.

Species	Common Name	Transects
Chara sp.	Muskgrass	T1, T2, T4
Equisetum fluviatile L.	Horsetail	T4
Iris sp.	Iris	NT
Isoetes sp.	Quillwort	T1, T2, T4
Juncus balticus Willd.	Rush	T2
Lemna minor L.	Duckweed	NT
Lysimachia sp. (unconfirmed)	Loosestrife	T2
Myriophyllum exalbescens Fern.	Water milfoil	T2, T4
Najas flexilis (Willd.) Rostk. & Schmidt	Bushy pondweed	T1, T2, T4
Nymphaea odorata Aiton	Water lilly	NT
Polygonum amphibium var. *stipulaceum* Coleman	Smartweed	T4
Potamogeton amplifolius Tuckerman	Pondweed	NT
Potamogeton gramineus L.	Pondweed	NT
Potamogeton perfoliatus L.	Pondweed	NT
Potamogeton pectinatus L.	Pondweed	NT
Potamogeton praelongus Wulfen	Pondweed	T1, T3, T4
Potamogeton zosteriformis Fern.	Pondweed	T2, T3, T4
Ranunculus longirostris Godron	Buttercup	T3, T4
Sagittaria latifolia Willd.	Arrowhead	NT
Scirpus cyperinus (L.)Kunth	Bulrush	T4
Sparganium sp.	Bur reed	T2
Vallisneria americana Michaux.	Wild celery	T1, T4

Sargent Lake

In total, 20 taxa of macrophytes were found (Table 11-8). No species identified is considered a new record for the island. *Nasturtium officinale*, or watercress, was an interesting find, only because this plant is usually associated with groundwater seepage areas. However, it is commonly collected along shallow shores of lakes. Macrophyte cover was about 30% in the littoral zone, estimated for the whole lake. Percent cover ranged from the maximum of about 80% to a minimum of 0%. The steep-sided lake bottom, common in Sargent Lake, limited the effective area of the littoral zone. Coves and small inlets tended to harbor more diverse and abundant macrophytes communities (personal observation). Of special note was the occurrence of *Myriophyllum alterniflorum*, a species that has been reported from ISRO, although Voss (1985) believes earlier reports to be false. We did not have flowers or fruits, but the leaf and stem morphologies fit that of *M. alterniflorum*. The species is a Michigan special concern species (L. Loope, pers. Comm.).

Table 11-8. Taxa list for Sargent Lake macrophytes. Transects T1, T2, T3, and T4 indicate where plants were found. Plants were collected/surveyed on 4 August 2000.

Species	Common Name	Transects
Carex sp.	Sedge	T1, T3, T4
Chara sp.	Muskgrass	T1
Eleocharis sp. (likely *elliptica*) Kunth	Spike rush	T2
Equisetum fluviatile forma l*innaeanum* (Doell)Broun.	Horsetail	T1, T2, T3, T4
Fontinalis sp.	Moss	T1
Iris sp. (likely versicolor)	Iris	T1, T3, T4
Juncus brevicaudatus (Engelm.)Fern.	Rush	T1, T4
Myriophyllum alterniflorum DC	Water milfoil	T1, T2, T3, T4
Nasturtium officinale R. Br.	Water cress	T1, T4
Nuphar sp. (likely *variegata)*	Yellow water lily	T1
Potamogeton amplifolius Tuckerman	Pondweed	T1, T3
Potamogeton gramineus L.	Pondweed	T1, T4
Potamogeton praelongus Wulfen	Pondweed	T1, T3, T4
Potamogeton zosteriformis Fern.	Pondweed	T1, T4
Sagittaria latifolia Willd.	Arrowhead	T1, T2, T4
Scirpus cyperinus (L.)Kunth	Bulrush	T1, T2, T3, T4
Scirpus validus Vahl or *acutus* Bigelow	Bulrush	T1, T2
Sium suave Walter	Water parsnip	T1
Sparganium fluctuans (Morong)Robinson	Bur reed	T1, T2, T3, T4
Utricularia vulgaris L.	Bladderwort	T1

Siskiwit Lake

We identified 8 taxa of macrophytes from Siskiwit Lake (Table 11-9). No species identified is considered a new record for the island. In general, the plant community of Siskiwit Lake is sparse (<<1% cover) and not very diverse. Most of the lake bottom is rocky, and the littoral area is very limited because of the steep slopes. However, we were not able to collect or see plants below 10 m. Light penetration may exceed this depth, and macrophytes could be present at the unsampled depths. The west end of the lake contained the greatest macrophyte abundance. In addition, *Lobelia dortmanna*, or water lobelia, was found in the shallower waters of the west end.

Table 11-9. Taxa list for Siskiwit Lake macrophytes. Transects T1, T2, T3, and T4 indicate where plants were found. Plants were collected/surveyed on 2 August 2000.

Species	Common Name	Transects
Chara sp.	Muskgrass	T3
Potamogeton praelongus Wulfen	Pondweed	T4
Potamogeton richardsonii (Benn.)Rydb.	Pondweed	T4
Potamogeton amplifolius Tuckerman	Pondweed	T4
Potamogeton gramineus L.	Pondweed	T3, T4
Lobelia dortmanna L.	Water lobelia	T1
Sparganium sp. (likely *variegata)*	Bur reed	T1
Juncus pelocarpus f. *submersus* Fassett	Rush	T1
Eleocharis acicularis (L.)R.&S.	Spike rush	T3

Chapter 12 Summary and Conclusions

Weather and Design Considerations

Winter 1997-98 was unusually warm due to the *El Nino*. In fact, several lakes had the earliest 'ice out' in decades. Many of the studied lakes that are typically dimictic were actually monomictic during 1998. This had significant implications for the water quality and plankton communities. Spring phytoplankton and zooplankton pulses were expected to occur much earlier and be less sharp because of the unusually warm winter without sustained ice cover. Deoxygenation of the hypolimnion and nutrient replenishment would likely be moderated. Although we attempted to develop a representative baseline by monitoring for two years, the unusual 1998 meteorological conditions may have been fortuitous since it represents an 'extreme' against which we can reference future results. Wide fluctuations observed in meteorology and thus, lake seasonality in two successive years emphasize the need to have long-term monitoring programs (greater then decadal) so that trends can be drawn.

In designing the present monitoring program, the authors and many reviewers considered two approaches: should each lake be characterized in general or should baseline information be collected at a fixed reference location that can be used for future indications of lake health. Because a fixed point does not detect all changes occurring lake-wide, it acts as a surrogate indicator of lake status. Thus, caution should be exercised in extrapolating the findings presented in this study to the lake-wide ecosystem and in generalizing the results beyond the two years of observation.

Water Quality

Indiana Dunes National Lakeshore

Long Lake is a polymictic lake, mixing throughout the ice-free season. It is not unusual, however, for this lake to lack oxygen during late summer months and extended ice cover periods. The lake is extremely shallow with a maximum depth of just over one meter. Much of the lake is covered by macrophytes from late spring through fall. Blue-green blooms are common. Recurring winter fish kills have been documented. Phosphorus and nitrogen levels indicate that Long Lake is in a stage of advanced eutrophication with a limited capability to sustain a balanced, diverse aquatic fauna.

Sleeping Bear Dunes National Lakeshore

Loon Lake remained stratified from late-May to mid-October during the study. Vertical oxygen profiles do not suggest high primary productivity during much of this time. Ammonia and nitrate nitrogen was typically low as was total phosphorus. Loon Lake appears to be a hard water lake. Elevated chlorides and specific conductance are likely related to unidentified upstream riparian influences of Platte River and Platte Lake. Low nutrients may in part be related to uptake of nutrients by algae and plants of Platte Lake, which have historically been higher in phosphorus. Eutrophication was not strongly evidenced in Loon Lake, and it is likely classified as a mesotrophic, dimictic (winter and summer stratification) hard water open (flow through) lake. We would predict reasonably good fish productivity from the water quality information collected.

North Bar Lake apparently remains stratified from at least June (mid-May in 1998) through September. The oxygen profiles indicate that plant productivity is moderate to moderately high. Water quality was surprisingly consistent throughout the warm months sampled. There were no signs of eutrophication. North Bar is a hard water lake with moderately high ionic content. Its water chemistry is not unlike Lake Michigan, which is predictable since it shares obvious surface and groundwater connectivity. It is likely classified as a mesotrophic, dimictic lake with no obvious water quality concerns aside from potential threats posed by Lake Michigan contaminants.

Round Lake remained stratified from May until September. Water quality parameters taken in the limnetic (open water zone) and littoral zone were within ranges expected for this type of lake. Like North Bar and Loon Lake, Round Lake is an alkaline, hard water aquatic ecosystem, low in nutrients and moderately low in productivity. It would be classified as an oligo- mesotrophic lake with no apparent water quality concerns. Its near-pristine condition, uniqueness among SLBE lakes, isolation from cultural influences, and the presence of bogs (facing exotic plant threats) make it deserving of special attention by the park.

Pictured Rocks National Lakeshore

In 1997, Beaver Lake became stratified in mid-July and remained so until the beginning of September. First stratification occurred in early June but did not have an extended stratification period. Dissolved oxygen remained relatively high in the epilimnion and metalimnion. Water quality variation was minimal. The increase of chloride over the season is likely related to evapotranspiration. Nitrogen and phosphorus data from both the hypo- and epilimnion suggest that excessive nutrients in Beaver Lake is not a problem and that the lake could likely be classified as a mildly alkaline, hardwater oligotrophic, polymictic lake.

Grand Sable was clearly stratified throughout the summers of 1997 and 1998. Oxygen vertical profiles were rather typical in 1998 for the type of temperature stratification observed, but metalimnion oxygen maxima seemed slightly exaggerated during 1997 relative to accompanying oxygen saturation data. Regardless of the source of this inconsistency, Grand Sable appears to be a moderately productive lake that stratifies strongly during summers. Complete turnover was evident. Nutrients (nitrates, ammonia and phosphorus), cations and turbidity were low; hardness was moderate with very small variation between dates. Grand Sable would likely be classified as an oligotrophic, slightly alkaline, dimictic lake with no apparent signs of water quality degradation.

Isle Royale National Park

Sargent Lake strongly stratifies through much of the summer months. Metalimnetic oxygen maxima suggest increased algal production in this zone. Variation in water quality parameters tested was minimal. Waters are slightly alkaline, low in dissolved ions, low to moderate in hardness, moderate in sulfates, and very low in phosphorous, nitrates, ammonia, and turbidity. Productivity was low, and the lake would likely be classified as oligo- to slightly mesotrophic and dimictic. There was no evidence of water quality degradation.

Siskiwit Lake was stratified by June during both the 1997 and 1998 sampling periods. It generally remained stratified until September. High water clarity allowed algal production throughout much of the water column as evidenced by vertical oxygen saturation deep into the euphotic zone. Nitrogen, phosphorus, sulfates and dissolved ions were typically very low. Hardness was moderate and the lake was slightly alkaline. The thermocline was at 8-12 meters, but the lake remained well-oxygenated to 32 meters. Siskiwit can be characterized as a very deep, oligotrophic, dimictic lake with high water clarity and low nutrients. No conspicuous water quality threats were evident in chemical limnological investigations.

Voyageurs National Park

Locator Lake was stratified during the entire sampling period in both 1997 and 1998. There was evidence of substantial daytime mid-water algal photosynthesis depicted by dissolved oxygen maxima commonly occurring at 3-6 meters (orthograde profiles). Although not statistically significant, there is an indication that phosphorus and turbidity trended higher in 1997. Temporal water quality variation within a sampling season was small. Locator is a softwater lake, low in ionic content and alkalinity. Nutrients and silica were very low. Chemical limnological inspection indicates that Locator is a dimictic, oligotrophic dilute lake with no pressing water quality issues.

Like Locator Lake, Mukooda remained stratified throughout the 1997-1998 sampling seasons. Positive, orthograde oxygen curves show high algal production within the epi-metalimnion, generally between 6-12 meters. This is characteristic of an oligotrophic coldwater lake with good transparency. Chloride was higher in 1998 than 1997, but specific conductance only weakly correlated with these results. Nitrate nitrogen was slightly higher in 1997. Like Locator, Mukooda is a softwater lake. Ammonia nitrogen, phosphorus, and nitrates were typically very low, suggesting oligotrophy. No water quality problems were detected.

Water Quality Comparison Among Lakes

Discriminant analyses showed that with a few exceptions, the Great Lakes Cluster Parks lakes could be separated based on alkalinity, chloride, or silica. Siskiwit and Mukooda Lakes, and North Bar and Loon Lakes had substantial overlap for alkalinity and chloride. Round Lake clearly separated by alkalinity alone. Sleeping Bear Dunes National Lakeshore lakes were substantially different than other parks in terms of water quality characteristics. For littoral zones, alkalinity and chlorides were the important factors. Differences in alkalinity can best be explained by basin geology: Sleeping Bear Dunes limestone shale dolomite; Pictured Rocks mixed dolomite-sandstone; Isle Royale conglomerate; and Voyageurs sedimentary-igneous.

Phytoplankton

Indiana Dunes National Lakeshore

Long Lake phytoplankton was dominated by yellow-green algae (54% in June) and euglenoids (60% in July), and cryptomonads were common in August (23%). Blue greens were generally low throughout the sampling period. Overall phytoplankton counts decreased steadily from May to September. Highest counts were 1.2 X 10^6 cells or colonies per liter. Chlorophyll *a* remained steady and high throughout the season, ranging from 19.4 mg/l in May to 10 mg/l in June. This elevated chlorophyll *a* and the high numbers of euglenoids indicate eutrophy. It is important to note that much of Long Lake's productivity and biomass is tied up in macrophytes and periphyton not the phytoplankton.

Sleeping Bear Dunes National Lakeshore

Diatoms dominated the phytoplankton community of Loon Lake, composing as much as 95% of the algal density in 1997 and 91% of the overall community in 1998. The dominant species for Loon Lake was *Cyclotella*. *Chlorochromonas* and *Synedra* were subdominant. Besides diatoms, only yellow-green algae made up greater than 1% of the community. Green algae tended to increase with season, and cryptomonads were present but low and sporadic. Highest densities tended to be in mid-summer, reaching 10^7 cells or colonies per liter, which many limnologists consider bloom conditions. Chlorophyll *a* concentration patterns were different between years. In 1998, chlorophyll *a* was highest in the spring and generally decreased. In contrast, chlorophyll remained stable through spring-summer of 1997 but spiked in the fall. There was little agreement between plankton counts and chlorophyll *a* density, likely due to the low initial concentration vs. inherent variability of chlorophyll (i.e. signal to noise).

North Bar Lake was clearly dominated by diatoms. Diatom relative abundance varied from 85% in September 1997 to 99% in June 1997. Yellow-green algae made up the next most dominant group, and ranged from 0.4 to 10%. Relative taxa abundance was mostly steady in 1997 but showed some variation in spring 1998. Density tended to decrease seasonally until September. Highest counts were near 10^7 cells or colonies per liter. The dominant species was *Cyclotella* in 1997, but no single species was clearly dominant in 1998; *Synedra, Chlorochromonas,* and *Acanthes minutissima v. minutissima* were common. Chlorophyll *a* concentrations were comparable in 1997 and 1998, ranging from 3-8 mg/l in the first part of 1997 season to 18 mg/l in August 1998. Chlorophyll results indicate a mesotrophic condition.

Diatoms also dominated Round Lake, ranging from 62% in May to 89% in June 1998. Cryptomonads comprised 26% of the community in May and declined to less than 10% of the community density thereafter. Greatest phytoplankton density was in May with 3.5 X 10^6 cells or colonies per liter. Thirty different taxa were identified from Round Lake. Chlorophyll *a* ranged from 5-15 mg/l. Interestingly, there was no spring algal pulse with lake turnover.

Pictured Rocks National Lakeshore

Unlike Sleeping Bear Dunes National Lakeshore, the Beaver Lake algal community was dominated by yellow-green algae in 1997, followed by diatoms and green algae. Highest algal counts were 1.75 X 10^6 per liter. Although the 1998 season was similar to 1997, results from May reflect conditions unlike the rest of the seasons. In May 1998, blue-greens were almost equal to diatoms in abundance. Yellow-greens generally dominated Beaver Lake for the remainder of the season while diatoms and greens were the next most dominant groups. Highest density in 1998 was 1.1 X 10^6 cells or colonies per liter. A total of 42 taxa were identified. During 1997, chlorophyll *a* was generally low: 1-6 mg/l. Chlorophyll was generally higher in 1998 with greater ranges. Chlorophyll concentration and algal counts were roughly consistent.

Yellow-green algae clearly dominated Grand Sable Lake from June to September 1997. In June, yellow-greens made up 60% of the community. Blue-greens rose to 30% in August. Some 40 taxa were identified during 1997, and densities rose as high as 9.3 X 10^5 cells or colonies/liter. Patterns in 1998 were similar to 1997. Yellow-greens again dominated the community, followed by diatoms, cryptomonads, and blue-greens. Blue-greens increased to as high as 41% in August. Highest 1998 counts were in May, 1.4 X 10^6 cells or colonies per liter. 50 taxa were identified in August. Chlorophyll *a* concentrations tended to be high and erratic, following no specific patterns.

Isle Royale National Park

Species evenness of phytoplankton was much greater at Sargent Lake than at either Pictured Rocks or Sleeping Bear Dunes lakes. For instance, in June no single algal division made up more than 33% of the community, and green algae made up 23%. In August 1997, greens composed 39% of the community, one of the highest representations of the taxa among the project samplings. Less taxonomic evenness was observed in 1998 with yellow-greens clearly the dominant taxon. Diatoms were also strongly represented, followed by green algae. The highest concentration was 1.1 X 10^6 cells or colonies per liter in May 1998. Chlorophyll *a* was generally lower in 1997 than in 1998 but relatively low through both years.

Diatoms clearly dominated Siskiwit Lake. Relative abundance ranged from 60% to 89%. Yellow-greens were also abundant but never dominant. Greens, blue-greens, and cryptomonads were far fewer in abundance. Highest numbers were found in June 1998, 1.1 X 10^6 colonies or cells per liter. Overall density between years was similar. A total of 40 taxa were counted in 1997, and 49 taxa were counted in 1998. Chlorophyll *a* was higher in 1997 than in 1998 but was generally low in both years. Clear patterns and relationships to algal counts were not strongly apparent.

Voyageurs National Park

Algal divisions were nearly even early in 1997 but shifted to diatom dominance by August. Still, species evenness was among the highest of any lake sampled. No single group exceeded half the community representation during the study period. Dinoflagellates comprised 10% of the population in July 1998, one of the highest occurrences of this group in the study.

The even distribution and high species diversity attest to the health of this lake and its trophic status. Highest algal counts occurred in June 1997, 1.4 X 10^6 cells or colonies per liter. Species richness was less than most other lakes studied, with 39 taxa identified in 1997 and 38 in 1998. Patterns of chlorophyll *a* over the season was similar between years and generally low suggesting oligo- to mesotrophic lake status. There was no clear relationship between chlorophyll and algal density.

Mukooda Lake was similar to Locator Lake in species evenness. The even distribution was more consistent through the season with no group exceeding 50%, although blue-greens comprised 45% in July 1997, which is similar to the 34% in August 1998. Aside from this strong representation by the blue-greens, most other major divisions were well represented with notable exceptions of diatoms in September 1998 and cryptomonads in August 1997. The highest number of algae was in July 1997 with 1.2 X 10^6 cells or colonies per liter. Thirty-eight taxa were counted in 1997 and 40 taxa in 1998. Chlorophyll *a* tended to increase during the season but did not seem related to algal counts. Phytoplankton community reflected an oligo- to mesotrophic lake condition with an interesting tendency to late summer blue-green shifts.

Zooplankton

Indiana Dunes National Lakeshore

Zooplankton peaked in mid-summer 1997 at about 1100 organisms/liter In Long Lake. Abundance was higher at Long Lake than other lakes studied presumably because of the eutrophic littoral nature of the lake. Species diversity was comparable to other lakes studied, helped by the dense vegetation and microhabitats that offer refuge from predation. Dominant taxa included the rotifers *Conochilus unicornis* and *Keratella* and the waterflea, *Bosmina longirostris*. *Bosmina* longirostris is commonly associated with lakes that are undergoing eutrophication.

Sleeping Bear Dunes National Lakeshore

In Loon Lake, limnetic zooplankton ranged from 90 to 379 organisms/liter, peaking in July 1997. The dominant species during this mid-summer high was *Conochilus unicornis*. In 1998 *Keratella spp.* was more important in overall zooplankton dominance. Limnetic densities were about 2 times higher in 1997 than 1998, which is the opposite of the trends in phytoplankton. This may have been due to sustained harvesting pressure of the zooplankton. Thirty-five taxa were counted during the study, and the diversity index ranged from 2.0-2.3 (Shannon-Weiner). Rotifers comprised about three quarters of the numerical density of Loon Lake. The most common microcrustacea were *Bosmina longirostris* and *Diacyclops* spp. Diversity was moderately high in Loon Lake, suggesting that the ecosystem is healthy.

Mean number of zooplankton was similar between 1997 and 1998 in North Bar Lake. Nonetheless, the patterns of occurrence were different between years, with densities peaking in May of 1997 and September of 1998. This abnormality may be explained by a reproductive pulse in the copepods, as evidenced by the high number of nauplii in the fall 1998 sample. 38 taxa were identified during the study with the rotifers *Kellicottia longispinus*, *Bosmina longirostris* and *Polyarthra* and the copepods *Diacyclops thomasi* and *Tropocyclops prasinus* *mexicanus* well represented. Between years, similarity was only 54%, suggesting that there is a large variation between years in this lake. Of particular note was the presence of zebra mussel veligers in samples from the limnetic zone in August and September 1998, not reported in the 1997 samples.

In Round Lake, summer density of zooplankton ranged from 89 to 383 organisms/liter, peaking in May and several-fold lower during the remainder of the summer. Highest spring density was largely due to *Kellicottia* early in the year, which was replaced by *Keratella* and *Bosmina*. We recorded 31 taxa, and the Shannon-Wiener diversity index was 2.0-2.1 reflecting good lake conditions. In general, the zooplankton assemblage was fairly typical of this type of lake in the region.

Pictured Rocks National Lakeshore

While mean zooplankton densities were similar in 1997 and 1998 in Beaver Lake, ranges were different: 62-133 organisms/liter and 49-422 organisms/liter. Seasonal patterns remained relatively stable during the summer of 1997 but had two pulses (June and September) in 1998. May 1998 was interesting not only because of the high zooplankton counts but the unexpected blue-greens that accompanied them. This is consistent with the rotifer-dominated population at that time. The high temporal variation and differences between the years were unknown but may have been weather-driven. Rotifers far outnumbered all groups, followed by cyclopoid and calanoid copepods. We encountered 32 taxa during the study and derived a Shannon-Wiener index of 1.7 – 2.1. Lower diversity was likely due to the oligotrophy of the lake plus the lack of structural complexity. Of special interest was the presence of *Epischura lacustris* and *Skistodiaptomus oregonensis*, which are typical of deep northern oligotrophic lakes.

Grand Sable zooplankton density was relatively low compared to other lakes studied. This lake wasn't sampled until July 1997, which may have been too late for the spring/early summer plankton pulse. Good microcrustacea recruitment was evidenced by high numbers of nauplii in June 1998. Mean density for 1997 was 26 organisms/liter compared to 45 organisms/liter in 1998. Shannon-Weiner diversity index was high: 2.3-2.5, suggesting high species evenness and richness as well as good lake conditions. Again, rotifers dominated the zooplankton especially *Bosmina longirostris*. Waterfleas were also quite common (e.g. *Daphnia galeata mendotae*, *Diaphanosoma birgei* and *Holopedium gibberum)*. Thus, while density was low, diversity was high, indicating a healthy, oligotrophic lake condition.

Isle Royale National Park

Sargent Lake zooplankton density was similar between years. Populations were bimodal, as is typical in a northern temperate dimictic lake. Summer means were 82 and 104 organsims/liter for 1997 and 1998. Even though patterns were similar, the relative abundance of individual species was quite different seasonally, with *Keratella* and *Conochilus* dominating these fluctuations. Population densities were moderate, but diversity was relatively high. A total of 39 taxa were found in Sargent Lake, which was one of the highest species richness values among the lakes examined in this study. Waterfleas included *Bosmina*, *Leptodora* and *Holopedium.* The latter species is typical of a softwater lake like Sargent. *Leptodora* was not commonly collected at other lakes in this study. The zooplankton community reflects a northern, soft, dilute oligotrophic lake in excellent condition.

Consistent with low primary productivity, zooplankton density in Siskiwit Lake was among the lowest of any lake studied. Mean summer density was 22 and 17 organisms/liter for 1997 and 1998. Peak zooplankton density occurred in June. Even though density was low, species richness was high (30 taxa counted). Shannon-Weiner diversity index was 2.4 – 2.5, which was among the highest in the study. This lake differed from other lakes studied because rotifers did not clearly dominate the zooplankton. This is consistent with a northern oligotrophic lake, which is more balanced between higher taxa. The fauna were representative of cold, softwater dilute lakes in this region. Littoral diversity and species evenness were also among the highest encountered in the study. In all, zooplankton reflected a very healthy, albeit low productivity, lake with no apparent water quality concerns.

Voyageurs National Park

Limnetic zooplankton counts in Locator Lake were twice as high in 1998 as in 1997; mean summer density was 115 and 63 organisms/liter. Because of the early warm weather in 1998, the summer-spring zooplankton peak may have occurred prior to sampling. Changes in the year were largely due to *Conochilus unicornis,* which tended to dominate. We encountered 34 species and derived a Shannon-Weiner diversity index of 2.2-2.4, one of the higher recorded in this study. Taxa composition had a similarity of 67% between years, which is relatively good compared to other lakes examined. The rotifer *Conochilus* was clearly the dominant species for Locator Lake, although *Kellicottia* was quite common at times. Littoral and limnetic zooplankton were moderately dissimilar in composition. The data suggest that the conditions in Locator reflect a well-balanced, cold water, oligotrophic, soft, dilute lake.

Zooplankton densities in Mukooda Lake tended to decline rapidly in both years sampled. Summer means were higher in 1997 (187 organisms/liter) than 1998 (74 organisms/liter). Perhaps a warmer winter in 1998 moderated nutrient regeneration, which expressed itself as a longer but less punctuated growing season. Rotifers clearly dominated the community, especially *Asplanchna, Kellicottia* and *Keratella,* but with strong representation by *Conochilus*. We identified 37 different zooplankton and derived a Shannon-Weiner diversity index of 2.2 – 2.4. Limnetic and littoral zones were quite different, as low as only 24% community similarity. For instance, *Collotheca* was present in the littoral zone but was not plentiful in the limnetic. Interestingly, *Bosmina longirostris* was absent from Voyageurs lakes but present at all other parks. In general, the zooplankton community reflected good water quality conditions for a northern, soft water, dilute, oligotrophic lake.

Benthic Macroinvertebrates

Sleeping Bear Dunes National Lakeshore

In North Bar Lake, limnetic density of benthic macroinvertebrates was markedly higher than in the littoral zone. Despite higher density, diversity was lower in the limnetic zone, as is typical for this habitat. Chironomids were the most abundant taxon in both the limnetic and littoral zones, comprising over half the insect community. Non-biting midges tended to decline through the sampling season, presumably through emergence and predation. The mayfly, *Hexagenia*, was also quite common. Amphipods, *Chaoborus* and tubificids were generally abundant. Biting midges were occasionally collected.

Limnetic and littoral macroinvertebrate densities were similar. In spite of these similarities, diversity was much higher in the littoral zone of North Bar Lake. Insects made up almost three-quarters of the community density with non-biting midges dominating. Phantom midges and tubificid worms were also common in the limnetic zone. The dragonfly (Gomphidae and Corduliidae) and the beetle, *Berosus* were noteworthy. In general benthic density was low, probably owing to the sandy substrate, which is not especially conducive to macroinvertebrate development.

Density of benthic macroinvertebrates in Round Lake was significantly higher in the littoral zone, and the diversity index in the littoral was 2-3 times higher than in the limnetic. For this lake, mollusks took on an important numerical role as was evidenced by the high number of snails and bivalves. Of the insects collected, midges were the most prevalent although high numbers of mayflies were also present. Other open water aquatic Diptera included phantom midges, midges, and biting midges.

Pictured Rocks National Lakeshore

Littoral density of macroinvertebrates was very high, 5000 $\pm$ 395 organisms/m^2 in 1997 rising to10337 organisms/m^2 in 1998. The limnetic zone overall density was 2500 organsisms/m^2 in 1997 compared to 4850 organisms/m^2 in 1998. This was about twice as high as the limnetic zone. In both years diversity was also quite high in the littoral zone and compared favorably to Grand Sable species diversity. Major taxa included midges and amphipods. Amphipods were quite common in the littoral zone and tended to increase during the season. We observed 19 genera in the littoral zone exclusive of the midges. The limnetic community was composed mostly of midges, but there were also sporadic occurrences of fingernail clams, oligochaete worms, and phantom midges.

Grand Sable Lake macroinvertebrate density was somewhat lower than Beaver Lake. The littoral zone macroinvertebrate density in 1997 was 3140 organisms/m^2 compared to 5269 organisms/m^2 in 1998. The benthic community of Grand Sable Lake was mostly midge larvae, which composed 70% of overall invertebrate relative abundance. Nonetheless the Shannon-Weiner diversity index was much higher in the littoral zone in large part due to amphipods and several species of caddisflies and mayflies. *Phylocentropus* sp. was the most common caddisfly found. Midges, oligochaetes, and phantom midges dominated the limnetic zone.

Isle Royale National Park

The benthic macroinvertebrate density of Sargent Lake was about 3000 organisms/m^2 in both the littoral and limnetic zones in 1997. There was no significant difference in density in 1998. Diversity was much higher in the littoral zone. Non-biting midges and phantom midges dominated the limnetic station along with oligochaetes. Whereas midges dominated the littoral zone, other species such as *Sialis*, five genera of caddisflies, and ephereid mayflies were well represented. The riffle beetle, *Dubiraphia* and *Macronychus* and the burrowing mayfly, *Hexagenia* along with the bivalve, *Pisidium* appeared in appreciable numbers.

Siskiwit Lake had macroinvertebrate communities notably different from others lakes studied. Overall density was several times lower in the littoral zone than in the limnetic zone although diversity was still higher. This is likely due to the selection of a sand-dominated littoral station, which typically does not have high species richness. The deepwater amphipod, *Diporeia*, dominated the benthic community in the limnetic zone with 64% of the relative abundance, and there were also spheriid clams (17%) and occasionally *Mysis relicta*. The littoral zone in 1997 was made up of midges (37%), one genus of mayfly, two genera of dragonflies, two genera of caddisflies, and amnicolid snails. In 1998, the diversity was slightly higher in the limnetic zone, likely due to the sandy nature of the littoral site selected.

Voyageurs National Park

Invertebrate abundance was significantly higher in the littoral zone in both 1997 and 1998 in Locator Lake. The littoral zone contained five genera of caddisflies, and none were found in the limnetic zone. *Hexagenia* was found in both zones, but abundance was higher in the littoral zone despite being nearly the same in relative abundance. The mayfly, *Ephemera*, the dragonfly larvae, *Epicordulia,* and *Tetragoneuria* were restricted to the littoral zone. Amphipods were also well represented. Midges dominated the littoral zone and phantom midges dominated the limnetic zone. Oligochaetes were most common late in the sampling year.

In Mukooda Lake, benthic macroinvertebrate density was about six times higher in the littoral than in the limnetic zone. Interestingly, Shannon-Weiner diversity was only slightly higher in the limnetic zone in 1997 and almost equal to the littoral in 1998. Oligochaetes and midges together comprised about 80% of the invertebrate community in the limnetic zone in 1997. Phantom midges were also common. Midges dominated the littoral zone despite at least 23 genera taken there. The amphipod, *Diporeia*, was found in both the limnetic and littoral zone although it was more common in the littoral zone.

Comparisons among lakes

Littoral zones. Density of macroinvertebrates was significantly different among lakes, with Long Lake and Siskiwit Lake consistently lower. Beaver Lake had the highest mean invertebrate population and the highest Margalef diversity index. There was a high species evenness at Long, Beaver, and Siskiwit Lakes. Density of macroinvertebrates in the littoral areas selected for study during 1998 was significantly different with North Bar and Loon Lakes consistently lower compared to other lakes. Beaver Lake again had the highest diversity in 1998, although it was only slightly higher than North Bar and Sargent lakes. Locator Lake had the lowest Shannon-Wiener index during the 1998 littoral samplings.

Limnetic zones. In 1997, Locator Lake was significantly lower than all lakes but Mukooda in benthic density, and Mukooda was significantly lower than Loon, Sargent and Siskiwit lakes. Grand Sable Lake had the lowest Shannon-Wiener diversity index, but in general this index was similar across the lakes studied. Species evenness was also similar among the lakes studied. In 1998, macroinvertebrate populations were significantly lower in density than all other lakes. Locator Lake was also significantly lower than most other lakes in density. Locator Lake had the lowest Shannon-Wiener index, but it was only slightly lower than Round Lake. Siskiwit and Mukooda Lakes had the highest Shannon-Weiner indexes among the lakes. Evenness was similar across all lakes except Loon and Mukooda, where it tended to be higher.

Similarity analysis

Based on similarity analysis of the taxa present and their densities, the limnetic stations selected for this study tended to cluster together, and the littoral tended to form a second cluster. The largest exception was Siskiwit Lake, where the littoral samples were species poor and the total invertebrate density was low relative to other lakes. This is likely due to the sandy nature of this site. Littoral clusters suggest that the Locator Lake station was not similar to the other lake stations selected. Both 1997 and 1998 had similar clustering patterns.

Chapter 13 Monitoring Plan Recommendations and Options

The table below outlines a possible matrix of monitoring options and sampling frequency for selected inland lakes of the Great Lakes Cluster Parks.

Plan Tier	Limnology	Water Quality	Sediment Quality	Phytoplankton	Zooplankton	Benthos	Cost/sampling point[1]
1	biweekly	monthly	5 years				$2K
2	biweekly	monthly	5 years	monthly	monthly (optional)		$4K to 6K
3	biweekly	monthly	5 years	monthly	monthly	monthly	$8K
4	biweekly	monthly	3 years	biweekly	biweekly	monthly	$12K

[1]includes only epi- or hypolimnetic chemistry when sampling site is limnetic

The annualized costs listed above do not include sediment chemistry. Sediment chemistry would include pesticides, semivolatiles, halogenated hydrocarbons, PAHs, PCBs and heavy metals. Sediment costs are roughly estimated at $1.5-2.5K per sample. Five year sampling intervals are likely adequate for most sampling programs for the lakes studied since there is little reason to believe that the sediment contaminant characteristics of these lakes are changing rapidly or that there is acute, immediate potential for organic and metal contamination. If lake status changes, threats become more likely, or more thorough information is needed for legal or remedial action, a more intensive regime could be implemented.

Limnological parameters listed in the above table include vertical oxygen, specific conductance, temperature profiles, and pH using the YSI multiprobes owned by each park. Each park can titrate chloride, alkalinity and hardness reasonably well with a minimum investment (less than $500 startup cost) and less then a few hundred dollars annually. Secchi disk transparency could be routinely done in the field. Total annual costs including YSI multiprobe costs would be about $1000. That mostly consists of normal instrument upkeep, chemicals and average supplies. This component constitutes the core of the monitoring program.

Water quality includes nitrate/nitrite-nitrogen, ammonia-nitrogen, total phosphorus, sulfate and perhaps, silica. Total cost is estimated at $1000 if done by a local commercial laboratory that meets EPA criteria.

Costs are rough estimates from the author's experience and may be revised after consultation with representative commercial vendors. Cost estimates are generally derived from private sector laboratories that comply with EPA guidelines. Government laboratory costs are generally higher but have rigorous quality assurance programs. Per-unit analysis cost would be far less expensive if done in-house (exclusive of contaminants), but personnel cost, QA/QC, OSHA, NPS, EPA and local state compliance become important issues. Once these additional costs are considered, in-house programs may be even more expensive than contracted services.

Phytoplankton, zooplankton or benthos costs include digitized counts to the nearest practical taxon (usually genus) and confirmation, but does not include interpretation or delivery of a reference collection. Professional costs by an established laboratory are usually $100-200 per unit, but less expensive services are often available through graduate students. Consistency is an important issue in biological analysis. Species identification of plankton and benthos are especially vulnerable to differing taxonomic nomenclature, personal expertise and identification consistency. For instance, some laboratories may be efficient with one or more taxa but poorly acquainted with other aquatic groups. Use of graduate students as contract technicians is temporary in nature and all too often the park is paying (and unknowingly suffering from) a student's learning curve. Chemical determinations have well-established protocols and are less vulnerable to this experimental variation. Thus, parks should seek long-term relationships with biological support services. Cost for biological analysis is estimated at $900 to 1800 per group per sampling location.

This matrix is for one sampling point taken in triplicate. Costs do not include contract sampling, travel, mailing or salaries and do not reflect costs of expert consultations and reports preparation. Prices will vary widely both in biological and chemical analysis. The manager is strongly advised to weigh cost, objectives, budget and quality of product before making a choice as to contract laboratory. Of these, the objective of the monitoring program will have the strongest influence. If the manager suspects that these data might be used legally, for instance, quality assurance and regulatory approval of the methods are critical. If the manager wants only an indicator and will not use their data for external comparisons, a more lenient approach might suffice. This is rarely the case in the long run. It is far better to err on the conservative side since few managers know just how these data will be used in the far future, and without strict guidelines the information is much more vulnerable to experimental error. For suggested protocols refer to the *Lake Monitoring Field Manual* (volume 2 of these documents) and the methods section of this report.

The manager or a designer should visit the contract laboratory or at least have a confident referral. Quality of the product is the most important attribute of contracted services, but ability to build a long term working relationship with the lab, timeliness of reporting, availability, technical support, data format, and logistics are important factors as well. Documented quality control must be insisted upon regardless. This could include control charts, reference tests, inter-laboratory comparisons and laboratory audits. It might be best for the Great Lakes Cluster Parks to use a common contract laboratory. This would assure better comparability in data sets and perhaps enhance economy of scale in the analysis and interpretation. Another approach would be to have one master park servicing other parks for specific parameters (e.g. phytoplankton, zooplankton, water quality). If the contractual route is desired, there are many experienced people in the National Park Service and USGS that can provide guidance and recommendations for services.

The table above illustrates a menu of monitoring plans for a single sampling location in a lake. In the present study we did general water chemistry for three locations for each lake: epilimnion, hypolimnion, and littoral zone. Thus, to cover each zone a park would need to take 3 replicates X 3 months (minimum) X 3 zones X 2 lakes (minimum) or 54 samples for water chemistry and

32 samples for benthos, phytoplankton and zooplankton per lake. Thus, the annualized cost per lake, exclusive of personnel, for a Tier 2 plan would be about $10K plus $2 – 3k for contaminants every 5 years. While the authors strongly recommend sampling the hypo- and epilimnion and the littoral zone of their subject lakes, we recognize that many parks simply cannot afford to monitor several lakes in this manner. Thus, the manager must choose between monitoring fewer lakes or fewer zones in each lake. While good arguments can be made on either side, we believe that it is better to monitor as many lakes as possible without compromising the objectives of the monitoring program: to provide good indicator information on the status and trends of selected lakes. Assuming such programmatic optimization is required, the best chemistry sampling location may be the epilimnion. The present study demonstrated that the water quality of the epilimnion and the littoral zone were not greatly different. Thus, knowing the epilimnion nutrients and limnological characteristics might be useful surrogates for the littoral area. Sampling for limnetic vertical profiles, Secchi disk, phytoplankton, zooplankton, and benthos would remain unmodified.

By concentrating sampling in the epilimnion, hypolimnetic water quality and littoral biotic information are compromised. Much of species diversity and lake productivity may occur in these zones; however, monitoring limnetic biology may be more efficient for long-term tracking. Habitat heterogeneity and species diversity make shifts in the littoral zone harder to identify on a short-term basis without many samples. Finally, the greatest ecological stress as well as the most critical lake metabolism takes place in the hypolimnion. Again, we strongly recommend monitoring the littoral zone and hypolimnetic chemistry and only present these considerations when priorities must be imposed.

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