Paleolimnological Reconstructions for the White Iron Chain of Lakes

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ABSTRACT

To quantify the environmental history of the White Iron Chain of Lakes (Lake and St. Louis Counties, Minnesota), five lakes were selected for retrospective analyses. Primary goals were to determine pre-European settlement conditions and track the timing and extent of anthropogenic impacts and remediation. Sediment cores were collected from each lake and sediment intervals were dated using isotopic analyses. Fossil remains, in concord with other stratigraphic indicators (organic and inorganic materials, sedimentation rates, other biological entities), were used to reconstruct the ~200-year history of each lake. Pollen analyses allowed for reconstruction of local and regional terrestrial conditions. Geochemical analyses provided data on historical flux of elemental trace metals to the sediments. Diatom assemblages were assessed from sediment intervals and inferred trophic conditions in the profiles were derived using a regional diatom-based model for Minnesota lakes. Eutrophication apparently occurred following settlement, particularly in White Iron Lake, but reconstructed phosphorus trends indicate more recent nutrient reductions. Pollen data track the decrease in pine abundance in the region and the rise of birch. Sedimentary metals largely reflect physical changes in the system, such as a change in sediment deposition regimes resulting from damming. Recent increases in metals are probably a result of increasing accumulation of soil and bedrock materials, a trend that is supported by increasing accumulation rates of overall organic and inorganic material. These recent increases in the last 30-40 years, which include increased algal deposition in Birch, Farm and Fall lakes, are not well explained at this time, but may be due to shifting water quality unrelated to phosphorus and possibly hydrological changes.

BACKGROUND AND INTRODUCTION

The White Iron Chain of Lakes (WICOL) comprises the lower portion of the Kawishiwi Watershed, draining an area of 1,200 square miles of northern Minnesota’s Rainy River Basin (Fig. 1). The primary source of inflow to the WICOL is from Birch Lake, the outflow of which is controlled by a dam. Other sources are the Bear Island River which drains into White Iron Lake, and the North Kawishiwi River which drains into Farm Lake. The WICOL system as a whole comprises several rivers and lakes which eventually flow from Garden Lake to Fall Lake, the last in the chain of lakes at the Winton hydroelectric dam.

Water quality in the WICOL has been subject to human-induced environmental changes since European settlement of the region approximately 140 years ago. Anecdotal and measured evidence has indicated that several stressors are having detrimental impacts, or have the potential for negative effects, on the quality of this system. Like other lakes in northern Minnesota, such as Shagawa Lake (Larsen et al. 1975, 1979, 1981), nutrient enrichment has undoubtedly occurred in the WICOL and may have contributed to algal growth. Humans have added excessive amounts of nutrients (primarily phosphorus, nitrogen, and organic carbon) to many Minnesota lakes in various ways including treated and untreated domestic wastewater, agricultural and urban runoff. Sewage was a particularly significant source of phosphorus to lakes when detergents contained large amounts of phosphate. Awareness of the potential for deleterious environmental damage has resulted in widespread action to reduce anthropogenic pollutant flux, such as that outlined in the Clean Water Act, which was enacted in 1972 (Litke 1999). Despite the success of nationwide remedial efforts, there are remaining pollution issues in Minnesota lakes caused by failing septic systems, contaminated sediments, abandoned waste sites, failing landfills, airborne deposition, industrial discharges, diffuse agricultural areas and other surface runoff (e.g. Cross and Jacobson 2013). Little is known whether these problems are resulting in lake-wide environmental problems in the WICOL.

In addition to pollutants, recent species invasions have impacted ecological conditions in many Minnesota lakes. For instance, large populations of the non-native zebra and quagga mussels, rusty crayfish and other species have developed within the last two decades in Minnesota lakes and rivers (Minnesota DNR

2008). In most cases, the impacts of these invasions on water quality and native species productivity are not well understood.

Another stressor of concern to the WICOL is erosion. As much of the watershed was deforested in the late 1800s through the early 1900s, and the WICOL was flooded starting in the late 1800s following dam construction (Mulholland 1996), erosion of the catchment became an important factor for bank stability, water quality and sediment accumulation rate issues. Shoreline erosion may still be a concern today as development of residential property expands and may be exacerbated by wakes from recreational motor boating, an activity that has increased over the last several decades.

The sensitivity of the lakes in the WICOL to environmental insults is not well understood. It is likely that there have been improvements in water quality following nationwide remedial mandates, but the long-term effects of both negative and preservative activities are rarely described in sufficient detail in aquatic systems. As a result, the sensitivity of these systems to future impacts is not known. In the case of the WICOL, shoreline development is likely to continue in the forms of housing, resorts and mining. By describing in detail impacts caused by past environmental insults, we can define the sensitivity of these lakes to stress and potentially predict the impacts of future development scenarios.

Long-term environmental data are critical to proper management of aquatic resources, but as is generally the case, long-term monitoring data are usually unavailable or incomplete. In cases where long-term data are available, these data tend to only cover recent time periods, generally after an environmental problem has been identified. Even with an exemplary, ongoing monitoring program for the WICOL lakes (e.g. http://kawishiwiwatershed.com), there is little evidence to adequately describe the long-term impacts human activities have had on their environmental quality, and the sorts of impacts that may result from future developments are unknown. Hence, management recommendations are lacking because a time-integrated assessment of the overall environmental status of the WICOL is needed. Paleolimnology offers a means to fill these data gaps in ecological history. Paleolimnology is the historical study of inland aquatic systems. In most cases, a core is taken from the sediments of a limnological system, and the fossils, geological and chemical signals that are preserved in the core are investigated to reveal the ecological history of the aquatic system and the surrounding landscape (Smol 1992). The evidence preserved in the sediments can be used to provide quantitative and qualitative reconstructions of important physical, chemical and biological trends that have resulted from natural and anthropogenic factors.

An incredible amount of information is stored in the biological fossil record of sediments. Algae are the most popular of paleolimnological indicators because they respond to stressors associated with major “pressure” indicators, including nutrient and salinity loading, siltation, and factors affecting water transparency (such as erosion and exotic species) (Smol and Stoermer 2010). The most commonly used biological indicators in paleolimnological analyses are the diatom algae (class Bacillariophyceae). They are ubiquitous, diverse, have a short turnover rate and have narrow tolerances to environmental conditions (Dixit et al. 1992). Diatoms are especially useful because their siliceous cell walls (frustules) leave diagnostic fossil remains that allow past species assemblages to be identified from the sedimentary record. There has been a rapid increase in the use of indices based on diatoms in aquatic ecosystems. A recent study (Reavie et al. 2006) indicated that diatoms can provide robust indicator models for several water quality variables. Moreover, the diatom assemblages better reflected impacts from watershed stressors (e.g., agriculture, urban development, point sources) than measured water quality parameters such as nutrients, water clarity and chloride, demonstrating that the diatoms were better suited to integrating environmental conditions than snapshot water quality measurements. The fossil diatom species present in sedimentary profiles from the WICOL can provide a detailed archive of past environmental information that would otherwise be unavailable. A paleolimnological assessment offers pre-settlement baselines, environmental trends, and the timing and magnitude of changes related to human activities. By reconstructing the long-term degradation and subsequent rehabilitation (if evident) of environmental quality, several important questions could be answered:

- What has been the extent of past ecological change, and how close have rehabilitation efforts come to returning water quality to pre-human settlement conditions?
- Since the initiation of remedial activities, what are the continued impacts on chemical variables and primary producers resulting from persistent stressors such as failing septic systems, erosion and invasive species?
- What is the trajectory of the inferred environmental trends, and is there evidence that additional remedial action might be needed?
- What is the sensitivity of the lakes to human activities, and how are they likely to respond to future development within the WICOL catchment?

This investigation developed quantitative information for aquatic management regarding environmental insults and remedial measures and their impacts on water quality in the WICOL. A paleolimnological analysis of the WICOL was undertaken based on sediment cores collected from five lakes. Physical, chemical and biological remains in the sedimentary profiles were used to develop lake environmental histories and trajectories. Data were related to known historical impacts and remediation efforts. The paleolimnological information obtained in this study will provide baseline data for sound management decisions and remediation goals for future efforts to maintain and restore the quality of ecological systems in the WICOL.

Region History

The following historical data were compiled from five main sources: Brownell (1981), Ely-Winton Historical Society (1982), Porthan (2008), Lamppa (2004) and Anderson and Baratono (2011).

Prior to 1800, the watersheds of these lakes were an area of virgin timber and wetlands. Indigenous people were in the area in low density. The WICOL area was originally populated by the Sioux and Chippewa Indians in the 1700s and 1800s. During the half-century from 1800–1850 Europeans are known to have harvested the area for beaver and other furs, but established no major settlements in the watersheds. The first apparent gold ore discovery occurred near White Iron Lake in 1884, but it turned out to be false. Confirmation of iron ore finds soon followed and deposits around the city of Ely were developed for mining starting in 1886 (Ely officially became a city in 1891).

Deforestation around the immediate vicinity of Ely began in 1888 and in many cases the slash areas burned fiercely for a few years. In 1892 the first logging camps and sawmills (particularly Knox's Mill) were built on the peninsula that is now Winton (southwest end of Fall Lake). Winton's population rapidly grew from 600 to 1900 due to sawmill employment. Logging activities from Knox's Mill peaked during the 1890s and Shagawa, Garden, White Iron, Fall and Birch lakes were often covered in floating logs. By 1910 it was realized that log drives in the Shagawa River (connecting Shagawa Lake to Fall Lake) were polluting the water being used by the city of Ely. Around that time most sawmills were purchased by the St. Croix Lumber Co. and more limited log booming began.

Beginning in the late 1890s Birch Lake Dam was progressively developed created to control the water for floating logs down the river to White Iron Lake. By 1900 lumbering in the area was common, and Highway #1 was constructed for the movement of product. As the city of Winton on Fall Lake became a prosperous sawmill location, major deforestation occurred from 1895 through ~1920 around Fall Lake, White Iron Lake and the Kawishiwi River. Between 1898 and 1920 the town of White Iron became bustling due to mining and timber activities. The first major Mill on White Iron Lake was built in 1921, shortly after Babbitt was officially established as a major mining town in 1920.

In 1909 Theodore Roosevelt set aside 900,000 acres of Superior National Forest for conservation after determining that increasing demand for lumber was going to exhaust supplies. Before this, however, most of the catchments of the WICOL had been logged to varying degrees. By 1911 it is estimated that 325,000 feet/day of lumber were being produced. This lumbering occurred simultaneously with the Swallow & Hopkins logging camps (which started closing in 1915 – fully closed in 1922); unfortunately records of their production are incomplete.

1900 to 1917 saw most of the logging activity on the Kawishiwi River. Dams were built to impound water for moving logs at: 1) the site of the current Winton Hydroelectric dam at the outlet of Garden Lake, 2) the site of the current Birch Lake Hydroelectric reservoir dam (raising the water level about five feet), and 3) the first narrows on the South Kawishiwi River below the point where the north and south forks divide. The purpose of the last dam was to divert flow to the North Kawishiwi River while logging occurred there. There are few remnants of this dam but other flow restrictions remain.

Between 1900 and 1921 there was also a fishing boom, but mass fishing was largely halted in the 1920s due to new regulations that prohibited commercialization of game fish. In 1922, water power work began on the falls on Kawishiwi and Fall Lake (between garden and Fall lakes), representing 4,000 acres of flowage (Fall Lake Boom Co., and Minnesota Utilities Co.). By that time the logging boom was largely over and much of the former forest was barren stumpland. Most camps and mills closed and were transformed or dismantled. By 1923 the Winton Hydroelectric facility was completed, resulting in a lowering of Garden Lake to about 1.5 feet lower than White Iron Lake. Logging dams were improved at the Birch Lake outlet and on the North Kawishiwi River (Murphy Rapids) to divert and impound water.

As result of these various industries the population of Ely grew rapidly, peaking in 1930 at 6151 people. Mining activity peaked in the 1930s following the logging decline. Iron mining activities, particularly processing ore for distribution, were prevalent around the shores of Shagawa Lake. In the immediate WICOL catchment, mining for taconite occurred near the southwest shores of Birch Lake. The Dunka Mine is an open-pit taconite mine where sulfide-containing waste material from the mine was stockpiled adjacent to wetlands that ultimately drain into the lake. Operations at the Dunka Mine occurred from 1964 through 1994.

Hydrologic modifications were periodically necessary after construction of the hydroelectric dam. In the 1950s there were record floods on White Iron Lake and other area lakes; White Iron Lake's depth increased approximately eight feet and the west bridge at Silver Rapids (at the White Iron outlet) was washed out. A 10-foot culvert was subsequently added to alleviate future flooding but was found to be ineffective. Eventually a longer steel bridge was placed over the rapids, and the culvert and additional rock and gravel (added for damming in 1927) were removed.

After 1950, roads were improved and more use was made of the lakes as places for seasonal, summertime recreation. In the 1970s many complaints were received about spring flooding on White Iron Lake with water level rises up to six feet. There were concerns by the Department of Natural Resources that spring draw-downs of water of up to three feet in the Garden Lake Reservoir to alleviate flooding in White Iron Lake during the spring snow melt were negatively affecting walleye reproduction by exposing spawning areas. As a result, the present bridge at Silver Rapids was built to minimize flooding impacts and the channel was dredged of old debris and abutments to provide additional drainage during high-water periods.

Shagawa Lake, which flows into Fall Lake via the Shagawa River, is well-known to have undergone cultural eutrophication and subsequent rehabilitation (Larsen et al. 1975). Wastewater from Ely initially flowed into Shagawa Lake untreated, began receiving primary treatment in 1911, secondary treatment in 1952, and tertiary treatment to remove phosphorus in 1973 (Larsen et al. 1981). Phosphorus load reductions had a significant positive effect on Shagawa Lake's water quality (Larsen et al. 1979). Although the period of catchment development around the WICOL (particularly White Iron and Fall lakes) is similar to that for Shagawa Lake, there are no clear data suggesting that eutrophication occurred on the WICOL.

Although accurate settlement data are rare, some statistics for White Iron Lake indicate the rate of development. Based on a 1982 inventory shoreline development on White Iron Lake included 135 homes and cottages and six resorts with 52 cabins. In 2001 a comparable inventory on White Iron Lake included 197 homes and cottages and four resorts with 42 cabins and 11 motel units.

Current Lake Conditions

The five study lakes are located in northern Minnesota. The WICOL intersects St. Louis County and Lake County. The lakes have a variety of physical conditions (Table 1), but are all considered mesotrophic, some periodically eutrophic, based on averaged physicochemical data from the last decade. In general these lakes are considered to have good water quality.

White Iron Lake is one of 24 "sentinel lakes" located around the state for the Sustaining Lakes in a Changing Environment (SLICE) project. SLICE is a cooperative lake monitoring program between the DNR Fisheries and the Minnesota Pollution Control Agency. The project is designed to help scientists understand, predict, and respond to outcomes of major drivers of change (e.g. development, agriculture, invasive species and climate) on lake habitats and fish populations. White Iron is considered mesotrophic and since 2003 is known to contain notable populations of the non-native rusty crayfish.

Fall Lake ranks as mesotrophic to slightly eutrophic. Much of nutrient load to the lake likely originates in Shagawa Lake. Fall Lake does not always stratify in the summers and water levels often vary as much as four feet annually due in part to the operation of the hydroelectric plant. Fall Lake contains a high abundance of the non-native rusty crayfish, which was confirmed in the lake in 1986.

Birch Lake's large, irregular shape is due to the fact that it is an impoundment of old river beds. There is a dam at the lake outlet (to White Iron Lake) and dam operation results in winter drawdown of about 4 ft. Birch Lake ranks as mesotrophic-to-eutrophic. Rusty crayfish were captured in the lake in 2003 and have since grown in number.

With the exception of fisheries data, the conditions of Garden and Farm lakes are not as well documented. It is probable that rusty crayfish have been present in Garden and Farm lakes for over 20 years. Based on nutrient and water clarity data, Farm Lake and Garden Lake are considered mesotrophic.

METHODS

Sediment Sampling

Sediment cores from each lake were collected in 2011 from a boat or through a hole cut in the ice surface. Sediments were collected using a piston corer operated from the lake surface by rigid drive-rods that pushed the core tube into the surface sediments. A corer equipped with a 7-cm diameter polycarbonate core barrel was used to collect a continuous section of the upper sediments at all coring sites that was at least 100 cm long to ensure the oldest sediment in the tube predated human impacts on the lakes (Wright 1991). The cores were incrementally extruded into 1-cm vertical sections which were placed into polypropylene collection jars. All core material was stored at 4 °C until further processing.

Inorganic and organic content, sediment dating, metals accumulation

Loss on ignition. Organic and inorganic loss-on-ignition (LOI) analyses followed Dean (1974). Sediment water content was determined from weight lost following oven drying of sediments at 100ºC for 24 hours. Weight loss after placing in a muffle furnace at 550ºC for two hours was used as an estimate of organic content. Weight loss after placing the remaining material in a muffle furnace for 2 hours at 1000ºC provided an estimate of carbonate content that is used to reflect inorganic carbonates (largely calcium carbonate) (Boyle 2001).

Sediment dating. Cores of lake sediments were analyzed for excess lead-210 (^{210}Pb) activity to determine age and sediment accumulation rates for the past 100-150 years. ^{210}Pb was measured at 15-20 depth intervals in each core through its granddaughter product ^{210}Po, with ^{209}Po added as an internal yield tracer. The polonium isotopes were distilled from 0.5-3.0 g dry sediment at 550 °C following pretreatment with concentrated hydrochloric acid (HCl) and plated directly onto silver planchets from a 0.5 N HCl solution (modified from Eakins and Morrison 1978). Activity was measured for 1-6 x 10^5 s with ion-implanted surface barrier detectors and an Ortec® alpha spectroscopy system. Unsupported ^{210}Pb was calculated by subtracting supported activity from the total activity measured at each level; supported ^{210}Pb was estimated from the asymptotic activity at depth (the mean of the lowermost samples in a core). Dates and sedimentation rates were determined according to the constant rate of supply (CRS) model (Appleby and Oldfield 1978) with confidence intervals calculated by first-order error analysis of counting uncertainty (Binford 1990). Dating analyses were performed by scientists at the Science Museum of Minnesota's St. Croix Watershed Research Station.

Metals. Analysis for trace metals was performed to provide stratigraphic surrogates for natural deposition due to erosion of soils and bedrock and human activities such as mining and tailings disposal. For each sample, sediment subsamples were freeze-dried and 0.25 ± 0.02 g of dry sediment were added to a 50-mL centrifuge tube. To this, 25 mL 0.5 N HCl was added and samples were heated at 80-85 °C in a hot-water bath for 30 minutes. Vials were transferred to an ice-water bath and allowed to cool for 5 minutes. Samples were centrifuged at 2000 rpm for 10 minutes, and then 10.0 mL of the supernatant was moved to 125-mL acid-washed poly-bottles. Each sample was diluted with 40 ± 0.5 g deionized water. Samples were assessed using inductively coupled plasma mass spectrometry (ICP-MS), which is capable of the determination of a range of metals and several non-metals (B'Hymer et al. 2000, Jarvis et al. 1992). These analyses were performed by personnel at the University of Minnesota Department of Earth Sciences, Analytical Geochemistry Laboratory.

Pollen

Pollen analysis (Maher 1977) was used to estimate changes in (mostly) terrestrial botany and to verify ^{210}Pb results and the timing of European settlement. Pollen grains are dispersed by plants for reproductive processes, and their analysis from sediments of freshwater lakes can provide information about vegetation dynamics through time. Pollen assessment methods were similar to Fægeri and Iverson (1989) with modifications by Heck (2010). After sediment dating, 20 samples per core were chosen to characterize the pre- and post-European time periods. *Ambrosia* (ragweed) pollen is well-known to increase in sediments following European settlement and deforestation activities in North America (Maher 1977), so we used *Ambrosia* as an additional confirmation of the dating profile achieved by using ^{210}Pb.

For each sample, ~1 g wet sediment was diluted with 30 mL high purity water in 40 mL glass centrifuge tubes, centrifuged at 2000 rpm and decanted. Then, 30 mL of 10 % HCl was added to each sample and tubes were placed in a 90 ºC hot water bath for 30 minutes. Decanting and washing with 30 mL water and centrifuging were repeated until the sample pH was greater than 6. Thirty mL of 10% KOH was added to each sample followed by another 30 minutes in the water bath. Centrifuging, decanting and washing were repeated until the samples were clear. Samples were poured through an 80-μm sieve and the > 80 μm fraction was transferred to a 0.5 dram glass vial. The < 80 μm filtrate was poured through a 20 μm sieve and the > 20 μm material was transferred to a 40-mL test tube. These samples were treated with 15 mL 2.5 % sodium hypochlorite and placed in the water bath for ~3 minutes. These samples were centrifuged and decanted, 6 mL of 10% HCL was added and samples were then placed in a hot water bath for 10 minutes. As above, samples were centrifuged, decanted and washed until the pH was greater than 6. Pollen material in these samples was recombined with the > 80 μm fraction in 0.5 dram vial. Enough silicone oil was added to each vial to cover the sample. A known stock of 20-μm polyvinyl beads in silicone oil was added to each sample for quantitative assessment.

For pollen counting, a glass stirring rod was used to remove a small amount of oiled pollen from a vial and make a "smear" slide for analysis. A coverslip covered and was adhered to the subsample. Pollen and other entities (beads and spores) were counted along transects until a total of 300 pollen and/or spore entities were encountered. Preparation and analyses were performed by Andrea Nurse at the University of Maine.

Diatoms and other microfossils

Diatom frustules were cleaned of organic material by digestion to allow identification of diatom species. Weighed sediment subsamples (~0.5 wet g) were diluted with 130 mL deionized water and treated with 20 mL concentrated nitric acid on a hot plate. Samples were heated at approximately 100 °C until 20 mL remained. Then, 25 mL hydrogen peroxide (30% solution) was added using a catalyst of potassium dichromate and again heated until 10-15 mL remained. Samples were rinsed eight times in centrifuge tubes by diluting and spinning down the samples at 2000 rpm for 10 minutes. Coverslips were prepared using the Battarbee (1986) method, pouring a known subsample diluted in water into dishes holding coverslips, preparing two slides per sample interval by adhesion with Naphrax® mountant. Naphrax® has a high refraction index and allows better distinction of the morphological features of the diatoms. In this instance, the Battarbee method allowed, for the quantitative assessment of diatom accumulation rates and productivity.

Diatoms were identified and enumerated using an upright light microscope at 1000-1250 x magnification with oil immersion. At least 400 diatom valves were counted per slide. To allow for discussion and taxonomic refinement, representative specimens were photographed with a Lumenera Infinity 2 digital camera. Diatom taxonomy and enumeration techniques were practiced and an initial taxonomic photo database was created to aid in consistent diatom identification. Diatoms were identified to the species level or higher using standard floras and iconographs, including Hustedt (1927-1966), Patrick and Reimer (1966, 1975), Camburn et al. (1984-1986), Krammer and Lange-Bertalot (1986, 1988, 1991a, b), Cumming et al. (1995), Reavie and Smol (1998), Camburn and Charles (2000) and Fallu et al. (2000). Diatoms were counted when more than 50% of the valve was present or when a distinct valve fragment was present (e.g., central area of *Amphora libyca* or valve end in *Asterionella formosa*).

To aid in temporally grouping historical trends in the diatom assemblages, cluster analysis was applied to the diatom assemblages in each fossil profile using the "chclust" function in R (statistical package version 2.15.1, R Development Core Team 2010) using the rioja package (version 0.7-3, Juggins 2012). The CONISS algorithm (Grimm 1987) was used to perform clustering constrained to vertical stratigraphy, providing a dendrogram for each diatom profile. Major clusters were used as guidelines for "periods" in each lake's history.

Although diatoms were the primary indicators used, siliceous remains including chrysophyte stomatocysts, phytoliths, plates of testate amoebae and sponge spicules were also enumerated to provide additional ecological information. Most species of the chrysophyte algae (Chrysophyceae) endogenously form siliceous resting stages called stomatocysts (Duff et al. 1995). In general, the presence of stomatocysts in the sedimentary record characterizes oligotrophic conditions (Smol 1985), and cysts have been applied to paleolimnological issues such as eutrophication (e.g. Carney and Sandgren 1983) and acidification (e.g. Duff and Smol 1991). Testate amoebae (Rhizopoda, also known as testaceans and thecamoebans) are freshwater protozoans that are common in soils, streams, lakes and wetland environments containing dense vegetation such as macrophytes and mosses (Warner 1990; Tolonen 1985; Loeblich and Tappan 1964). Phytoliths are microscopic siliceous bodies that form in stems, leaves, roots, and inflorescences of plants. Following the disintegration of organic tissues, the phytolith microfossils are preserved in sedimentary records (Rovner 1971). Spicules are the microscopic skeletal elements of sponges, and these typically siliceous bodies are preserved in sedimentary records following the disintegration of organic tissues.

Diatom-Inferred Environmental Conditions. Diatom calibration and training sets have become powerful tools for paleoecological reconstruction. The development of weighted averaging regression and calibration introduced a method of quantitative reconstruction of historical environmental variables (Birks et al. 1990a,b). The method uses a transfer function developed from a training set of modern diatom assemblages and their relationships to environmental gradients. The transfer function can be applied to historical diatom assemblages in sediment cores to mathematically reconstruct specific environmental variables. The weighted averaging method is statistically robust and based on ecologically sound organismal responses (ter Braak and Prentice 1988, Birks et al. 1990b), and the approach has been used successfully in reconstructing a wide variety of environmental parameters including pH, total phosphorus (TP), dissolved organic carbon (DOC) and salinity (e.g. Anderson 1989; Fritz et al. 1991, 1999; Dixit et al. 1992; Reavie et al. 1995).

To develop a set of useful diatom indicators, a diatom transfer function is derived by relating diatom species assemblages in a training set of samples (e.g., from lakes, river reaches, coastal locales) to an environmental variable of interest (e.g., total phosphorus or nitrogen, pH, chloride, suspended solids) from a particular region (Charles 1990, Juggins and Birks 2012). The transfer function consists of species coefficients (e.g., environmental optima and tolerances for each species) that can be used to infer quantitative information about the variable of interest, based on the relative abundance of each species in a sample assemblage. Past total phosphorus (TP) concentrations were inferred from the fossil diatom assemblages using the diatom-based reconstructive model that has been developed (Ramstack et al. 2003) and progressively updated (Heiskary and Swain 2002, Edlund and Kingston 2004, Reavie et al. 2005) for Minnesota Lakes. Weighted averaging (WA) calibration and regression were used to derive a diatom-based model using R version 2.15.1 (R Development Core Team 2010) and the package rioja version 0.7-3 (version 0.7-3, Juggins 2012). Rioja quantitatively reconstructs environmental variables using weighted averaging of the diatom species and their associated environmental optima and tolerances. The diatom transfer function was derived by relating diatom species assemblages in the training set of phytoplankton samples to measured TP. Past TP concentrations were inferred from the fossil diatom assemblages using the transfer function, which estimates TP based on the TP optima of the fossil taxa, weighted by their relative abundance. The full model is known to have good performance statistics (Reavie and Juggins 2011), and details of diatom-based transfer function development and application are provided by Juggins and Birks (2012).

Two methods, analog and fit-to-TP analyses, were applied to ensure the diatom-based TP training set model was appropriate for application to the diatom assemblages in the WICOL cores. Analog analysis was performed using the R package analogue (Simpson and Oksanen 2011). Analog analysis implemented matching of assemblages among modern samples from the training set and fossil (downcore) diatom assemblages from this study, following Flower et al. (1997) and Simpson et al. (2005). The method identifies the closest analogs of fossil samples from the modern training set to assess the reliability of diatom-inferred data. The R script generates a pairwise dissimilarity matrix for the modern training set, and a second matrix containing the pairwise dissimilarities between each fossil sample and each sample in the training set. Analogs were determined using Bray-Curtis dissimilarity (Bray and Curtis 1957). Dissimilarities of the ten "closest" (i.e. most similar) modern assemblages to each fossil assemblage were compiled to assess how well each fossil assemblage was represented in modern collections.

A canonical correspondence analysis (CCA) constrained to TP was used to evaluate relationships between fossil samples and the modern TP gradient. The residual distance of fossil diatom assemblages to the TP axis provided a measure to assess "lack of fit" to TP. First, using the R package vegan (Oksanen et al. 2011), CCA determined extreme residual distances from the TP axis (i.e. axis 1) by ordinating modern diatom sample assemblages constrained to TP. TP effectively becomes the first CCA axis and the distance of each modern sample score from this axis reflects its "fit" to TP. Fossil samples were then run passively in CCA using the "timetrack" function in the R package analogue (Simpson and Oksanen 2011). Fossil samples were positioned, by means of transition formulae, with respect to the TP axis. Fossil samples with residual distances greater than the 95% confidence limits of the training set sample scores were considered to have "poor fit" to TP, and so inferred TP from those assemblages would be considered unreliable.

RESULTS AND DISCUSSION

Dating

Total ^{210}Pb in recent (core-top) sediments ranged from ~10 pCi/g in Birch Lake to 25 pCi/g in White Iron Lake, and down-core declines were largely smooth and monotonic (Fig. 2, left panel). The systems had fairly typical decay curves, exhibiting exponential declines in ^{210}Pb concentrations. Supported ^{210}Pb was well defined in all lakes and allowed for accurate calculation of dates according the CRS model (Fig. 2, center panel). Overall, these profiles would be expected in lakes with consistent sediment accumulation regimes, and we are confident that relatively undisturbed sedimentary profiles were obtained. Date-depth relationships were similar for Birch, Farm and Fall lakes, whereas higher average accumulation rates in Garden and White Iron resulted in deeper intervals representing the pre-European period. Sediment accumulation rates varied considerably among the five cores (Fig. 2, right panel): Birch and Farm Lake exhibited a recent increase in accumulation; White Iron Lake had an earlier increase in accumulation around the time of first Euro-American settlement but has since been relatively stable; Garden Lake's accumulation has fluctuated erratically for the last 200 years; Fall Lake had a substantial, temporary increase in accumulation just before the turn of the 20th century, but has since had accumulation rates similar to or lower than pre-European rates.

Sediment content

Loss-on-ignition (LOI) analysis (Fig. 3) provided retrospectives of the various sediment components. Water content was typically in excess of 70%, although Fall Lake sediments had very low water content until the last decade, likely owing to higher inorganic content and less porewater space in the sediments. A substantially lower proportion of organic content in Fall Lake's record supports this, further indicating that the lake has consistently received a large amount of inorganic input, and organics increased since human development, most markedly so in the past decade. Aside from the most recent decade which shows an increase in all but Garden Lake, little trend was observed for the proportion of carbonates in the sediments, although White Iron Lake had a naturally higher carbonate load than the other lakes. Accumulation rates of organic material indicate in increase in accumulation in the last ~60 years in Birch and Fall, in the last ~30 years in Farm and no consistent long-term trend in Garden. Accumulation rates in organic and carbonate material in White Iron Lake increased at the period of initial settlement (prior to the 1900s) but the lake has since largely maintained higher, steady rates. The high carbonate accumulation rates in the uppermost intervals of Farm, Fall and Birch lakes suggest recent, higher inputs of carbonate material, but this may be due to incomplete integration of new materials in the surface interval with the sedimentary record. Additional discussion of accumulation rates is provided in the context of their microfossil profiles.

Metals

Profiles of accumulation rates of trace metals, including several basic, transition, and alkali metals (Fig. 4), provide a history of metal supplies to the sediment in the five lakes. Fall Lake had an especially high pre-settlement metal load, indicating that relatively high amounts of erosional material (e.g. from soil and bedrock) containing these elements were accumulating in the basin before damming and transition to a more lacustrine system. In general the other lakes had similar pre-settlement rates of metals accumulation, although Farm Lake overall had the lowest rates. The initial phase of settlement just prior to the 20th century apparently caused a sudden decline in accumulation of these elements in Fall Lake, likely as a result of a revised sedimentary regime that reduced erosional supplies to lake sediments. The settlement period initiated the start of gradual increasing trend in White Iron Lake. Many elements (e.g. Al, Ba, Ca, Na, Si, Sr) increased in White Iron Lake until approximately 1950, after which concentrations remained steady. Iron (Fe) loads to White Iron Lake may be continuing to increase. Since approximately the 1980s, accumulation rates of all recorded elements have been increasing more rapidly in all of the lakes, particularly in the Birch Lake core. While elevated metals in Birch Lake, and secondarily in White Iron Lake, may be associated with elevated mining-related metals that have been detected in wetlands associated with the Dunka Mine (Eger 2013), it is worth noting that there is a small concomitant increase in metals in Farm Lake which has no upstream mining activity in its history.

Pollen

Although the profiles of pollen proportions (Fig. 5) and accumulation rates (Fig. 6) are somewhat chaotic, owing to multiple short-term fluctuations in pollen deposition and inter-lake variations in accumulation, several characteristics of historical terrestrial condition are apparent from these plots.

Pollen subfossils were overwhelmingly dominated by pine species, particularly white (*Pinus strobus*) and red (*Pinus resinosa*) pine, as well as several pine pollen grains that could not be differentiated into species. The period of damming and log booming at the turn of the 20th century had varying pollen trends. Accumulations of softwoods (mainly pine) increased in Garden, Fall and White Iron lakes, possibly due to lumber and log running activities. The accumulation of pine species pollen has decreased since then in Garden and Fall Lakes while hardwoods including birch (*Betula*) increased in all lakes except Fall Lake. White Iron Lake has maintained a higher accumulation of softwood and hardwood pollen since ~1900, indicating a unique contribution compared to lower concentrations in the other lakes. There was little overall trend in aquatic taxa (including emergent macrophytes and the floating species *Lemna*), although there may be increasing accumulations within the last decade, suggesting higher contributions from littoral areas (Fig. 6).

The proportions of *Ambrosia* (ragweed) pollen increased starting in the late 1800s (Figs. 5,6), associated with initial deforestation of the region. As an opportunistic taxon that establishes in newly-cleared forests (Swain 1973), the appearance of the *Ambrosia* horizon confirms our dating profile for the timing of the post-settlement period.

Charcoal particle counts were also included to estimate relationships to forest fires, but no trends were observed with the exception of very high charcoal accumulation at the time of European settlement (Fig. 5), concomitant with very low pine pollen abundance. Whether this reflects a significant fire influence is unknown. Accumulation rates in general (Fig. 6) reveal that pollen accumulation tended to be low in Fall Lake.

Diatoms and other entities

We encountered more than 830 diatom taxa and several additional entities in the five lake cores. Additional entities included the stomatocysts (resting stages) of chrysophyte algae, plant phytoliths, sponge spicules and the plates of testate amoebae.

The accumulation of diatoms in the sedimentary records varied among the five lakes, suggesting variations among historical lake productivity histories (Fig. 7). After settlement, White Iron Lake experienced a ~50-year period of increased algal accumulation which largely returned to pre-settlement levels by the 1970s. Diatom accumulation in Fall Lake increased in the early 1900s and has maintained a similarly higher rate of accumulation to the present day. The Farm Lake profile indicates a continuously increasing accumulation rate, whereas the accumulation rate in Birch Lake only began increasing in the last ~25 years. Garden Lake exhibited a complicated history of diatom accumulation, rising temporarily at the turn of the 20th century and subsequently experiencing periodic peaks and valleys. In the last ~50 years, the relative proportions of pennate diatoms has been increasing while centric forms necessarily decreased, suggesting increasing inputs from near-shore and wetland benthic taxa. A temporary increase in centric diatoms in Farm and Birch lakes at ~1920 reflects a change to more planktonic diatoms, likely as a result of flooding and increased planktonic growth in those lakes.

The general long-term decline in the ratio of stomatocysts to diatoms suggests a slight shift to higher productivity as chrysophytes tend to be more abundant in oligotrophic conditions (Smol 1985). However, there was no up-core decline in stomatocyst accumulation rates with the exception of White Iron Lake which declined in the late 20th century in line with diatom declines. Stomatocysts may be increasing in the last few decades in Birch and Farm lakes. Combined with diatom trends, chrysophyte fossil trends weakly suggest long-term eutrophication in the system.

Accumulations of phytoliths, sponge spicules and amoebae plates reflect inputs from littoral, wetland and riparian habitats. The mid-20th century increase in phytoliths and plates in White Iron Lake suggests a period of flooding, expansion of the littoral zone and increased erosion of soils which stabilized after approximately 1970. Recent increases in these entities in the last ~30 years in Birch and Farm Lake indicate increasing inputs from littoral areas and possibly increasing erosion from soils and littoral areas.

Because of variations in diatom taxa encountered among cores, diatom profiles and diatom-inferred total phosphorus (DI-TP) trends are presented separately for each lake in upstream-to-downstream order (Figs. 8-22). Large amounts of data are presented for each lake; not all profile data presented are addressed in this discussion, but instead we consider each lake's data in context and focus on key trends for interpretation. Cluster analyses on diatom profiles showed remarkably similar results between relative abundance and accumulation rate data, so discussion of clustered periods (below) reflects overall diatom trends.

Birch Lake. Figures 8 and 9 present the more common taxa observed in the Birch Lake core. Cluster analyses of relative abundance (Fig. 8) and accumulation rates (Fig. 9) indicate unique groups in the stratigraphy. There is a clear pre-impact period prior to the turn of the 20th century, characterized by several phytoplanktonic *Aulacoseira* species. From ~1900 through ~1970 planktonic, pennate diatoms such as *Asterionella formosa* and *Tabellaria flocculosa* became more abundant, reflecting more lacustrine conditions due to the occurrence of these lighter taxa that are better adapted to slow sinking. In the phytoplankton, *Aulacoseira ambigua* was partly replaced by *Aulacoseira subarctica*, a taxon with a lower phosphorus optimum (Reavie et al. 2005), suggesting some dilution of the standing load of nutrients may have occurred due to flooding. *Fragilaria crotonensis*, a mesotrophic taxon, and the oligotrophic *Aulacoseira alpigena* became more abundant in the recent sediments since ~1980. During this recent period accumulation rates of diatoms have increased dramatically, suggesting greater productivity, although species autecologies do not suggest increases in nutrient flux. The physical and/or chemical reasons for this recent change require further exploration.

Analog comparisons of the Birch Lake diatom assemblages with modern assemblages indicate that deeper assemblages tended to have poorer similarities to model data (Fig. 10a). This reveals that older fossil assemblages are poorly represented in the diatom-based TP model, which is perhaps not surprising as the physical condition of the lake was much more riverine during pre-settlement times, and so the assemblages from that time would be less representative than modern conditions in Minnesota lakes. Nonetheless, even at the bottom of the core there were at least two modern samples that were significantly similar to fossil samples (Fig. 10b). Further, all fossil sample scores were within the 95% distance from the TP axis, indicating good fit to TP for all fossil samples (Fig. 10c). The inferred TP profile (Fig. 10d) reflects that suggested by general observations of the diatoms, above; i.e., a decline in TP occurred following European settlement and conversion of the system to be more lacustrine. There may have been temporary nutrient enrichments since the major physical change at the turn of the 20th century, but modern inferred TP concentrations are comparable to those from pre-settlement.

White Iron Lake. White Iron Lake had a similar pre-settlement assemblage as Birch Lake (Fig. 11), which is not surprising considering their connectivity. However, accumulation rates in White Iron were generally higher, resulting in a larger number of "common" species presented in Figure 12. As for Birch Lake, the post-settlement transition and flooding of the lake, marked by a ~1920 transition between lower and mid-core clusters, coincided with an increase in *A. formosa*, indicating more lacustrine conditions. The most notable change from the early 1900s through ~1970 was an increase in *Aulacoseira granulata*, a taxon with a relatively high phosphorus optimum (Reavie et al. 2005), indicating nutrient enrichment. Accumulation rates of several diatom taxa increased during that period (Fig. 12) suggesting greater overall algal productivity. The most recent assemblage cluster representing the 1980s through 2010 indicates a recovery of several taxa to approximately pre-settlement levels. Persisting higher relative abundances of certain taxa (e.g. *Fragilaria capucina* var. *mesolepta* and *Achnanthes minutissima*) in the recent sediments suggest that the physicochemical conditions in the lake are continuing to change.

Analog and fit-to-TP analyses indicate good representation of the fossil data in modern (model) assemblages, so DI-TP data should be reliable (Fig. 13a,b,c). In general, modern-fossil sample similarity was better in upper sediment samples. The DI-TP profile indicates that periods of higher phosphorus concentrations occurred after ~1900 (Fig. 13d). The profile suggests that short-term fluctuations in nutrient loads have occurred, and that trend of erratic shifts persists to the present day.

Farm Lake. As for Birch and White Iron lakes, the earliest major diatom assemblage cluster occurs at and before the turn of the 20th century (Fig. 14). The whole core contains a high relative abundance of *Aulacoseira ambigua*, an often meso-eutrophic taxon (Reavie et al. 2005). The transition to the 20th century was marked by an increase in *A. subarctica*, a mesotrophic taxon, and *A. granulata*, suggesting some nutrient enrichment. Since the early 1900s there has been little consistent shift in the relative abundances of the diatoms, however the accumulation rate profiles (Fig. 15) record a significant long-term increase in phytoplankton, especially *A. ambigua* and *A. subarctica*. A post-1980 cluster in Figure 15 demarcates the uniquely high accumulation rates during the most recent period.

There was little apparent temporal trend in analog comparisons between modern and fossil diatom assemblages, and similarities were good for all core samples (Fig. 16a,b). Fit-to-TP analysis also indicates DI-TP reconstructions should be reliable for Farm Lake (Fig. 16c). Aside from a temporary drop in DI-TP at the time of initial flooding and probable chemical dilution of the Lake, there is little overall indication that nutrient enrichment occurred in the lake (Fig. 16d). The monotonous DI-TP trend suggests that the increase in algal supply to the sediments (Fig. 15) is a result of factors other than phosphorus.

Garden Lake. The historical dominance of mesotrophic (e.g. *A. ambigua, A. subarctica*) and probable epiphytic (e.g. *A. minutissima*) species in the pre-1900 cluster in the Garden Lake core (Fig. 17) indicates that Garden Lake is naturally productive. The post-1900 cluster generally tracks a higher accumulation rate of several diatom taxa (Fig. 18), indicating a more productive system, although the taxa that increased in relative abundance after ~1900 (e.g. *A. subarctica*, *A. formosa*, *F. crotonensis*) do not suggest a significant long-term increase in nutrient supply. Just prior to ~1940 there was a brief period of lower accumulation rates. The most recent diatom cluster (largely post-1960) reflects a relative increase in select species such as *F. crotonensis*.

Comparisons of modern and fossil assemblages indicate somewhat better analogs in upper sediment samples, but all fossil samples appeared to have good modern analogs and good performance in fit-to-TP analyses (Fig. 19a,b,c). With the exception of a brief period of increased DI-TP for ~25 years after ~1900, little consistent trend in nutrients was observed (Fig. 19d).

Fall Lake. The pre-impact diatom cluster for Fall Lake represents a largely phytoplanktonic assemblage dominated by *A. ambigua*, *A. subarctica*, *A. granulata*, *A. alpigena* and an unknown taxon tentatively named *Aulacoseira* "sp. 3a WIC" (Fig. 20). After ~1900 the assemblages were substantially changed to be more dominated by *A. subarctica*, *A. formosa* and *T. flocculosa*, indicating the transition to a more lacustrine system and possibly a slight increase in nutrients as indicated by slightly elevated *A. granulata* (Reavie et al. 2005). Greater algal supplies to the sediments were indicated by increases in accumulation rates of several taxa (Fig. 21). A post-1970 diatom cluster exhibited continuing higher deposition of *A. formosa* and *F. crotonensis*, although there was lower accumulation and relative abundance of *A. granulata*, suggesting a lower nutrient load in the last ~30 years.

Analog and fit-to-TP analyses indicated good overall representation of Fall Lake's fossil assemblages in the TP model (Fig. 22a,b,c). As suggested by the changes in dominant diatom species, the early 1900s was apparently a period of higher nutrient loads, but since the ~1950s these loads have dropped to pre-European settlement levels (Fig. 22d).

GENERAL DISCUSSION

Although there were variations in the paleolimnology of the five lakes, several general statements can be made about the WICOL system as a whole. A significant disruption that is evident in the paleoecological record was the transition from a fluvial to a more lacustrine system to facilitate movement of lumber and eventually to produce hydro-energy. Even without significant increases in nutrient flux to the lakes, and perhaps even a nutrient dilution effect, slowing the flow, deepening and creation of new littoral areas resulted in a more productive system. Conversion to a more lacustrine system likely manifested greater lake productivity due to increasing water replacement time which allowed for establishment of primary producer communities (algae and macrophytes). Phytoplankton abundance increased and species composition changed to suit the new physical environment. Organic accumulation increased and in the case of Fall Lake, which apparently experienced the most dramatic physical change at the core location, accumulation of inorganic materials including metals dropped as a probable result of reduced fluvial supply of suspended solids following gradual damming of the system. Increased sediment supplies may be related to increased shoreline erosion resulting from boat traffic and/or climatological factors, but greater study is needed to better quantify such drivers.

The most notable invasive taxon in the WICOL is the rusty crayfish, but we do not infer strong influence on overall lake conditions due to spread of the crayfish throughout littoral areas. If crayfish were having a major impact on macrophyte beds one might have expected declines in epiphytic diatom taxa in upper intervals of the sediment cores, but this does not appear to be the case. Nearshore impacts in heavily invaded areas likely need better study to evaluate localized impacts of the rusty crayfish.

Pollen data tracked terrestrial trends for the region and around specific lakes. Overall the post-19th century decline in softwood species is apparent, as is the subtle increase in dominance of birch. Surprisingly, lake-specific accumulations of pollen remains track significant variations in localized deposition that is relevant to each lake's catchment.

While there is some evidence of eutrophication in the 20th century, it has been more pronounced in certain lakes, and apparently phosphorus as a driver of increased productivity was only temporary. It is likely that increased nutrient loads in the early 20th century resulted from direct anthropogenic inputs such as sewage and from increased runoff resulting from deforestation. It also appears that better measures to control nutrient loads (e.g. sewage treatment at Ely) and allowing regrowth of forests has resulted in some remediation of the system. Very recent increases in organic and inorganic supplies and algal remains indicate that biological and sediment loads are on the rise in certain lakes, a trend that may continue. Reasons for this recent shift need further exploration. Lake-specific summaries are as follows.

Birch Lake has experienced the greatest long-term increase in organic and inorganic sediment accumulation as well as a very recent increase in accumulation of algal and other biological remains. The causes of these shifts appear to be unrelated to nutrient flux, so determinants such as changing hydrology and chemistry need to be studied. Birch Lake has also experienced the greatest recent increase in sediment metals.

White Iron Lake was the most productive lake overall in terms of diatom abundance in the sediments and accumulation rates of organic materials. Its high sediment accumulation rate is reflected in a greater supply of elemental metals in the sedimentary record. The lake experienced a period of increased productivity following initial settlement, an event that may have been partly attributed to increased nutrient loads. While sensitive to anthropogenic drivers, the resilience of the lake to environmental impacts is evident by the recent reversion to pre-settlement levels of algal supplies. Pollen records suggest that White Iron Lake has the highest relative proportion of forest cover in its catchment.

Farm Lake exhibited the most notable long-term, persistent increase in diatoms over the last ~100 years. Again, this apparent change in productivity does not appear to be related to increased nutrients. As for White Iron Lake, accumulation rates of organic and inorganic materials are increasing, but the rate of increase has risen in the last ~30 years.

Garden Lake had the most complicated diatom history, but it maintained the increase in productivity resulting from initial flooding as observed for the system as a whole. The lake has experienced a series of events that are not yet fully explained: long up-and-down periods of organic accumulation; a brief period of nutrient enrichment following hydrologic modification; a brief drop in diatom accumulation just before ~1940; and a strong peak in pine pollen accumulation at the turn of the 20th century. Despite its small size the lake has clearly undergone several short-term changes, but overall it has been fairly resilient to human effects.

Fall Lake exhibited the greatest evidence of physical changes, most notably the shift from mainly inorganic accumulation prior to European settlement to a more lacustrine, organically supplied lake, particularly since the 1950s. In terms of algae the lake developed a much greater abundance after settlement, and inferred phosphorus concentrations reflect a broad, ~60-year period of nutrient enrichment. The more recent sedimentary record suggests recovery of nutrient levels, but the algal supply to the sediments remains similar to that following initial damming.

FUTURE RECOMMENDATIONS

Based on inter-lake comparisons, Birch Lake is changing most rapidly and very recently. Because Birch Lake is a relatively large, complex system, additional sediment cores are warranted to better characterize long-term condition and trajectory. Specifically, a sequence of cores from Kramer Bay through Dunka Bay and Klobuchar Bay would be informative to better capture the changes taking place in the lake. Additional work near suspected “hotspots” may also help evaluate environmental trajectories and legacy effects from past mining activities.

Two more paleolimnology study sites are warranted in Fall Lake to better describe the history of this complex lake. It is not known whether our findings near the outlet apply to the whole lake because Fall Lake comprises several basins that may have unique histories.

An update to Shagawa Lake’s environmental history is needed. Thorough paleolimnological study of Shagawa Lake has not taken place since treatment system post-installation studies of the 1970s (Larsen et al. 1979). Fall Lake may have a gradually increasing algal load, and this may be partly due to source water from Shagawa Lake. Shagawa Lake is unquestionably the most nutrient-impacted lake in the region, and further study of the lake would clarify nutrient-biology relationships and help better define changes that have taken place in the WICOL.

It may be worthwhile analyzing a sediment core from White Iron’s southern basin to provide a better picture of inter- and intra-lake variability.

From this study it is obvious that, despite the hydrologic connectivity of the lakes in the WICOL, there are significant variations in ecological characteristics and responses among lakes. Hence, other lakes of concern should be subject to study. In particular, lake systems in the boundary waters would provide reference or “background” sites that would provide historical interpretations of lakes with no present-day catchment development. For instance, if a similar increasing productivity trend like that seen in Birch Lake is also observed in undeveloped lakes it would suggest that the recent change in Birch Lake is due to larger-scale issues (e.g. climate, precipitation).

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Table 1. General characteristics of the study lakes based on Minnesota Department of Natural Resources (DNR) collections. Numbers in parentheses represent the number of samples collected from 2003 through 2012 that were averaged to produce the value.

Lake	**Birch**	**White Iron**	**Farm**	**Garden**	**Fall**
DNR number	69000300	69000400	38077900	38078200	3808110
Core latitude (N)	47º46.85	47º53.26	47º53.77	47º55.46	47º57.83
Core longitude (W)	91º45.95	91º46.80	91º43.61	91º45.10	91º43.32
Surface area (acres)	5628	3238	1292	653	2258
Littoral area (acres)	1060	1603	459	239	1178
Maximum depth (ft)	25	47	56	55	32
pH	6.53 (159)	6.92 (483)	7.03 (582)	6.99 (327)	6.72 (73)
Specific conductance (µS/cm)	76 (158)	56 (250)	49 (46)	64 (70)	45 (82)
Mean measured total phosphorus (ppb)	24 (19)	20 (45)	17 (29)	19 (32)	21 (17)
Chlorophyll *a* (µg/L)	6.7 (39)	4.6 (72)	4.5 (155)	5.3 (63)	6.5 (34)
Secchi depth (m)	1.36 (67)	1.76 (276)	1.93 (394)	1.57 (315)	1.68 (37)
Minnesota Department of Natural Resources trophic status	mesotrophic to slightly eutrophic	mesotrophic	mesotrophic	mesotrophic	mesotrophic to slightly eutrophic

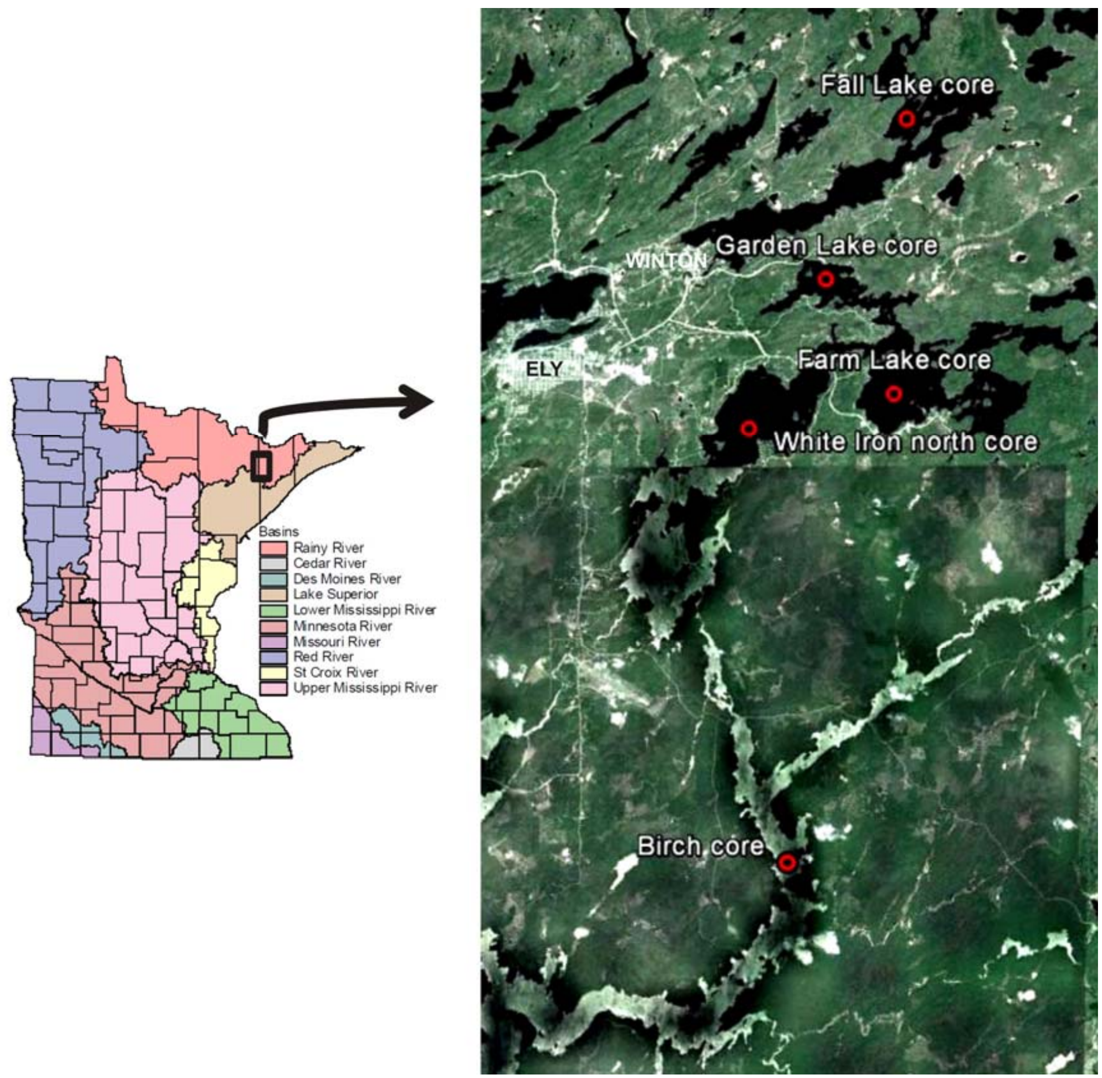

Fig. 1. Locations of the five sediment cores in the White Iron Chain of Lakes. The map was provided by Google Earth.

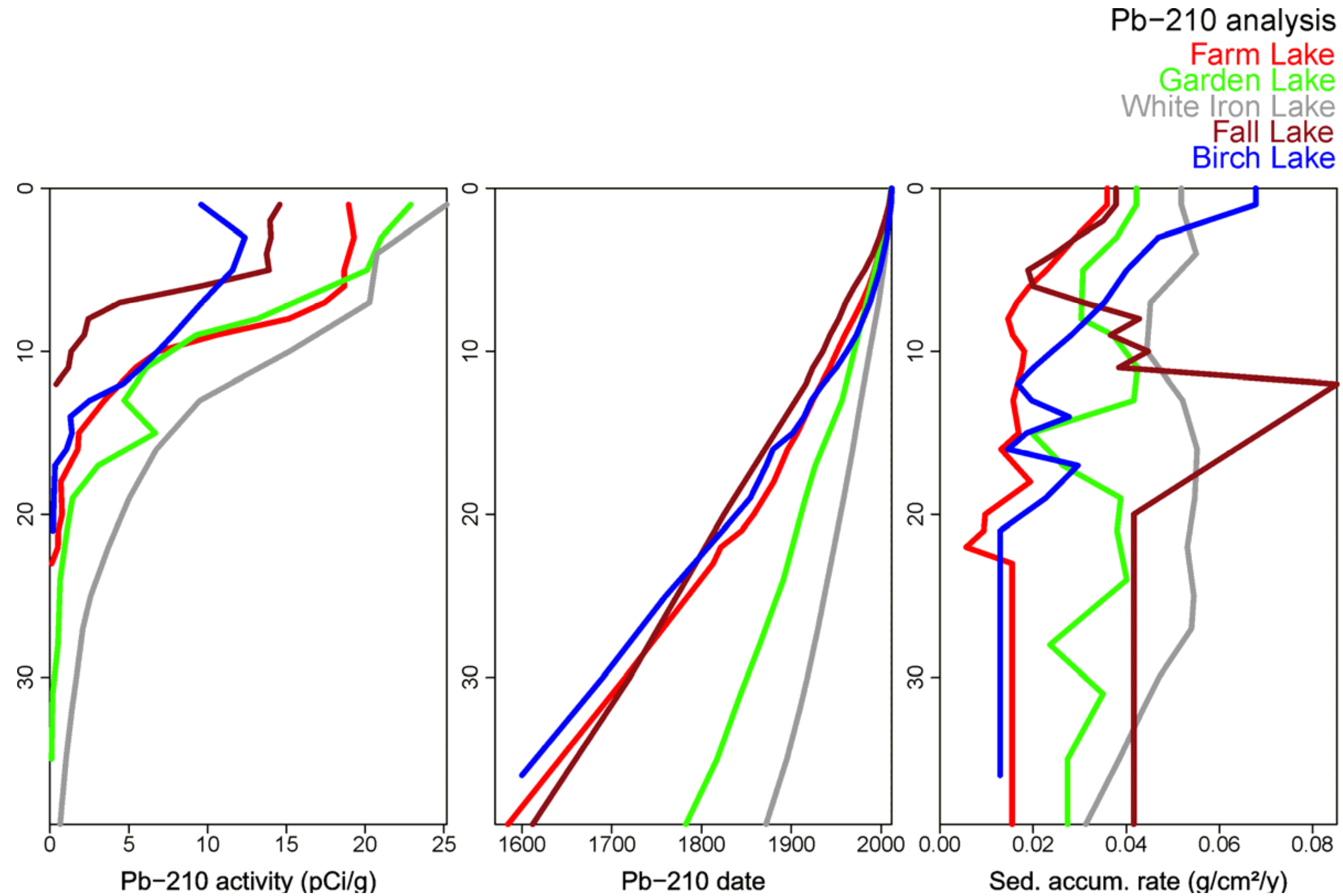

Fig. 2. Lead-210 activities (left panel), date-depth relationships (center panel) and sediment accumulation rates with depth (cm) in sediment cores collected from the five study lakes. Lake-specific line plots are identified by their corresponding color in the heading.

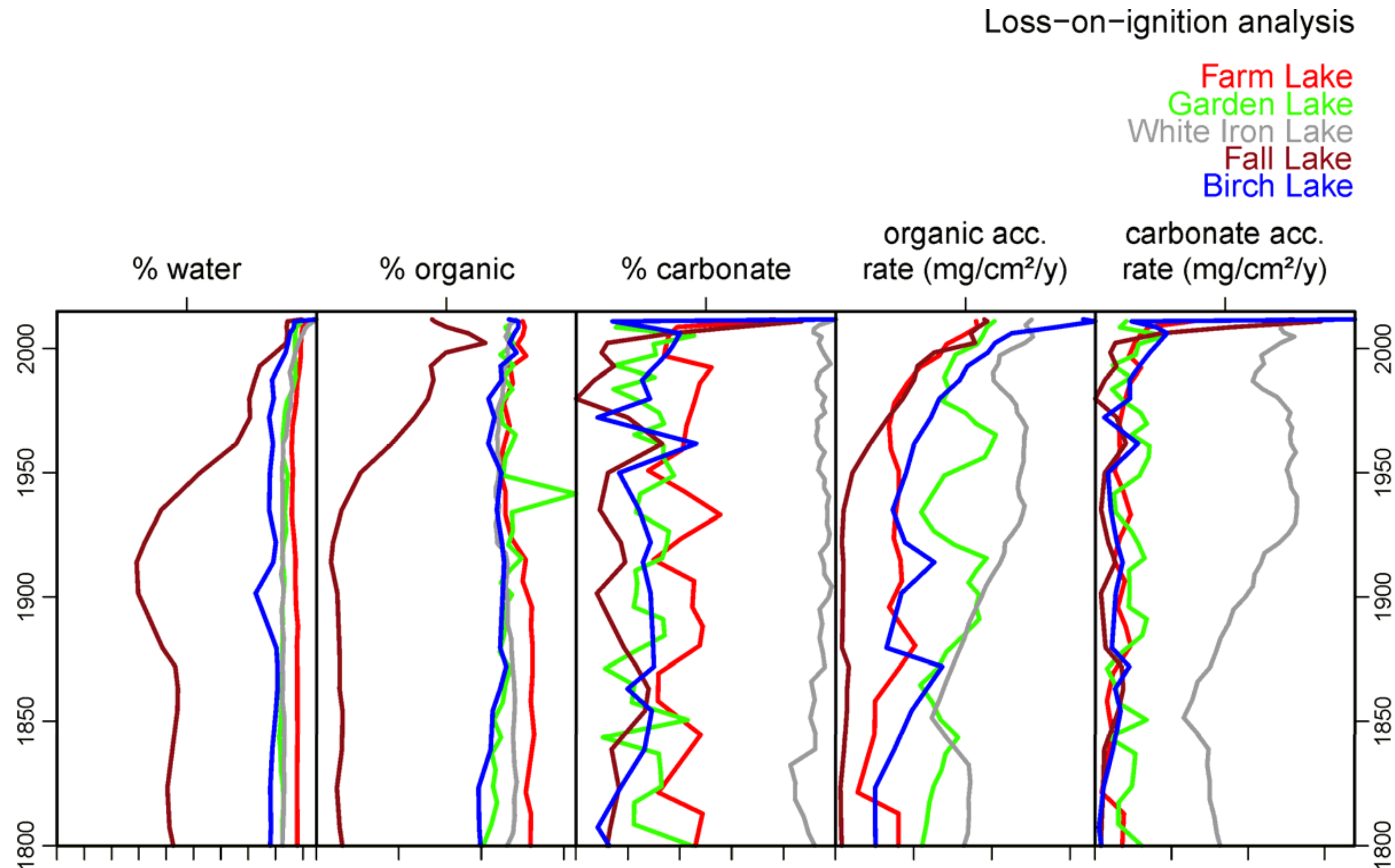

Fig. 3. Downcore results based on loss-on-ignition (LOI) analyses for the five sediment cores. Water, organic and carbonate contents are provided as proportions in each sediment sample, and the two right-hand panels present accumulation rates for organic and carbonate materials.

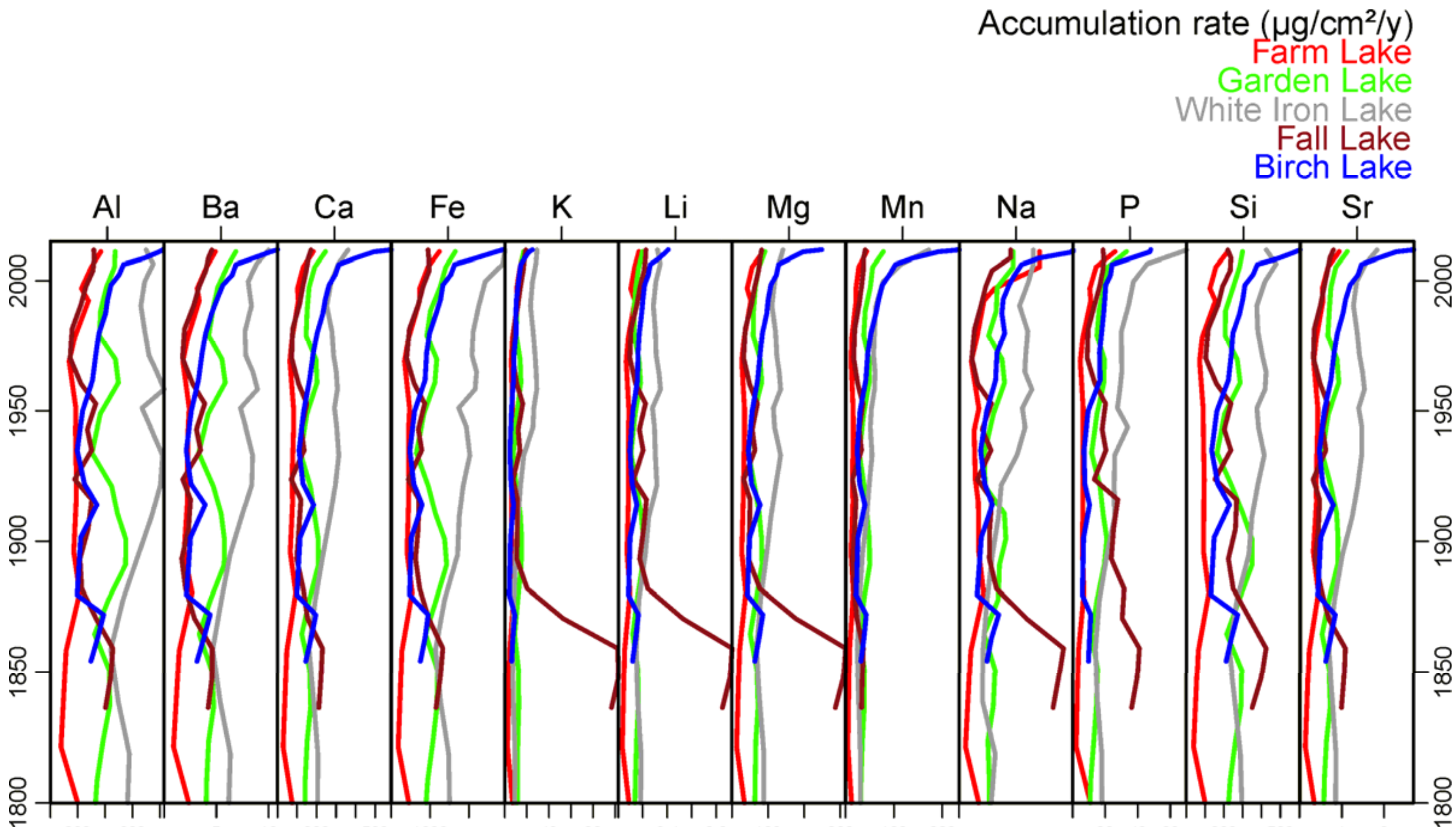

Fig. 4. Sediment trace metal data from the White Iron Chain of Lakes. Data are presented as accumulation rates versus date, and lake-specific line plots are identified by their corresponding color in the heading.

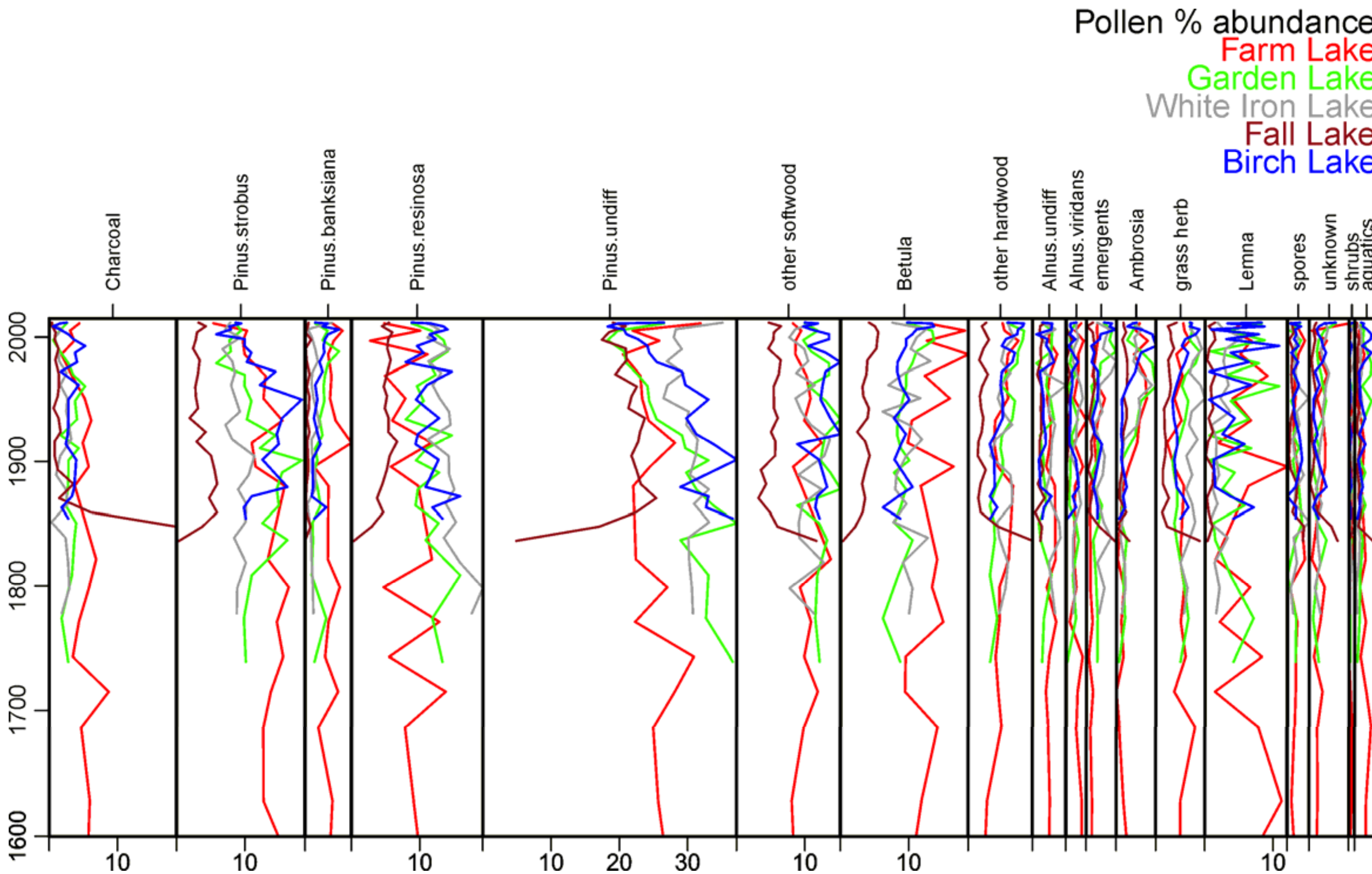

Fig. 5. Sediment pollen data from the White Iron Chain of Lakes. Data are presented as relative proportions in each sample versus date, and lake-specific line plots are identified by their corresponding color in the heading.

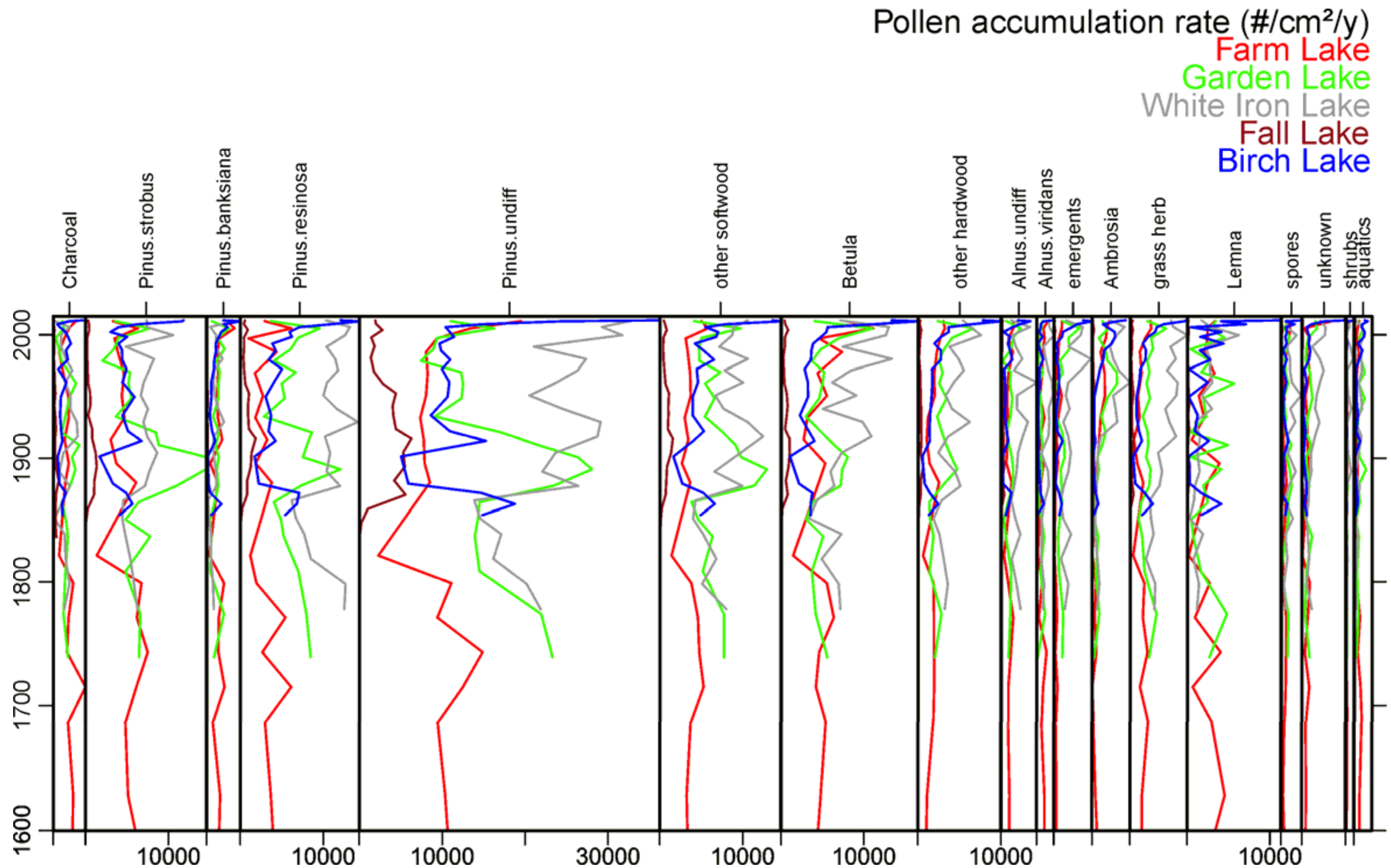

Fig. 6. Sediment pollen data from the White Iron Chain of Lakes. Data are presented as pollen accumulation rates in each sample versus date, and lake-specific line plots are identified by their corresponding color in the heading.

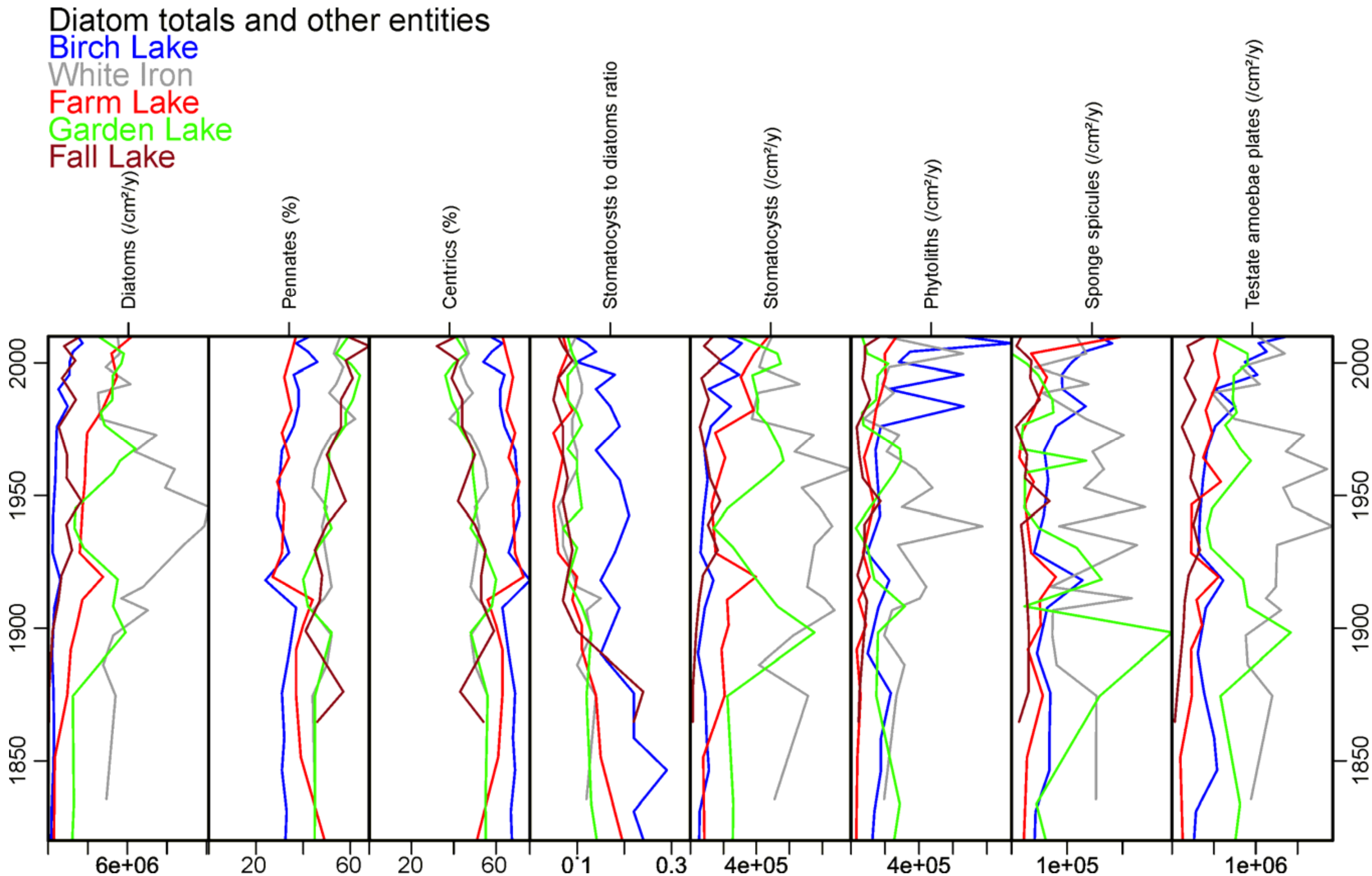

Fig. 7. Downcore sediment trends for diatoms, chrysophyte stomatocysts, phytoliths, sponge spicules and testate amoebae plates from the White Iron Chain of Lakes. Data are presented as accumulation rates or relative abundance (%) to illustrate trends. The ratio of stomatocysts to diatoms is presented as a unitless ratio. Lake-specific line plots are identified by their corresponding color in the heading.

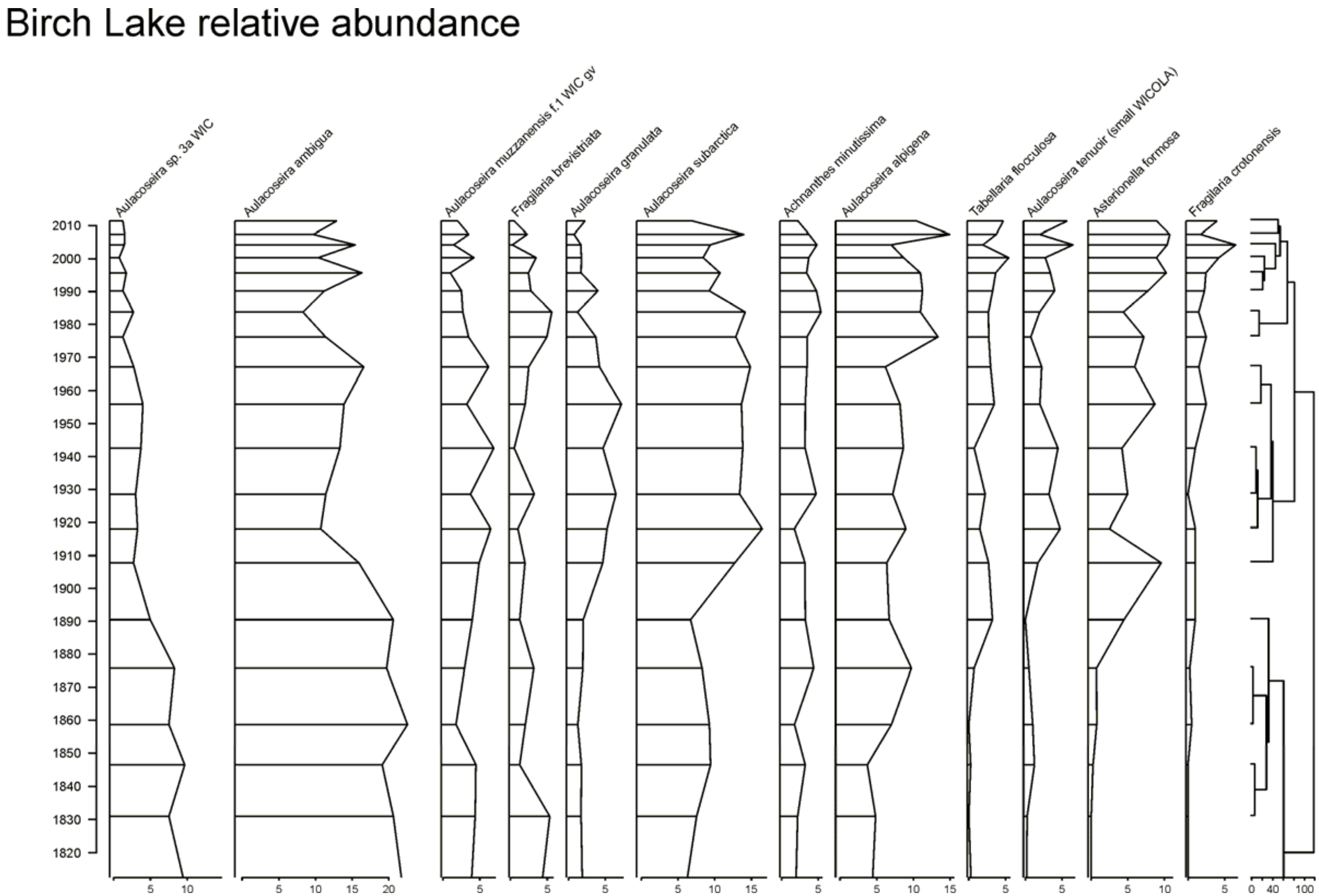

Fig. 8. Downcore relative abundances (%) of diatom sediment-fossil remains for Birch Lake. Taxa shown in this plot occurred at a relative abundance of at least 5 % in at least one sample. A dendrogram illustrating depth-constrained cluster analysis of the diatom assemblages is plotted on the right.

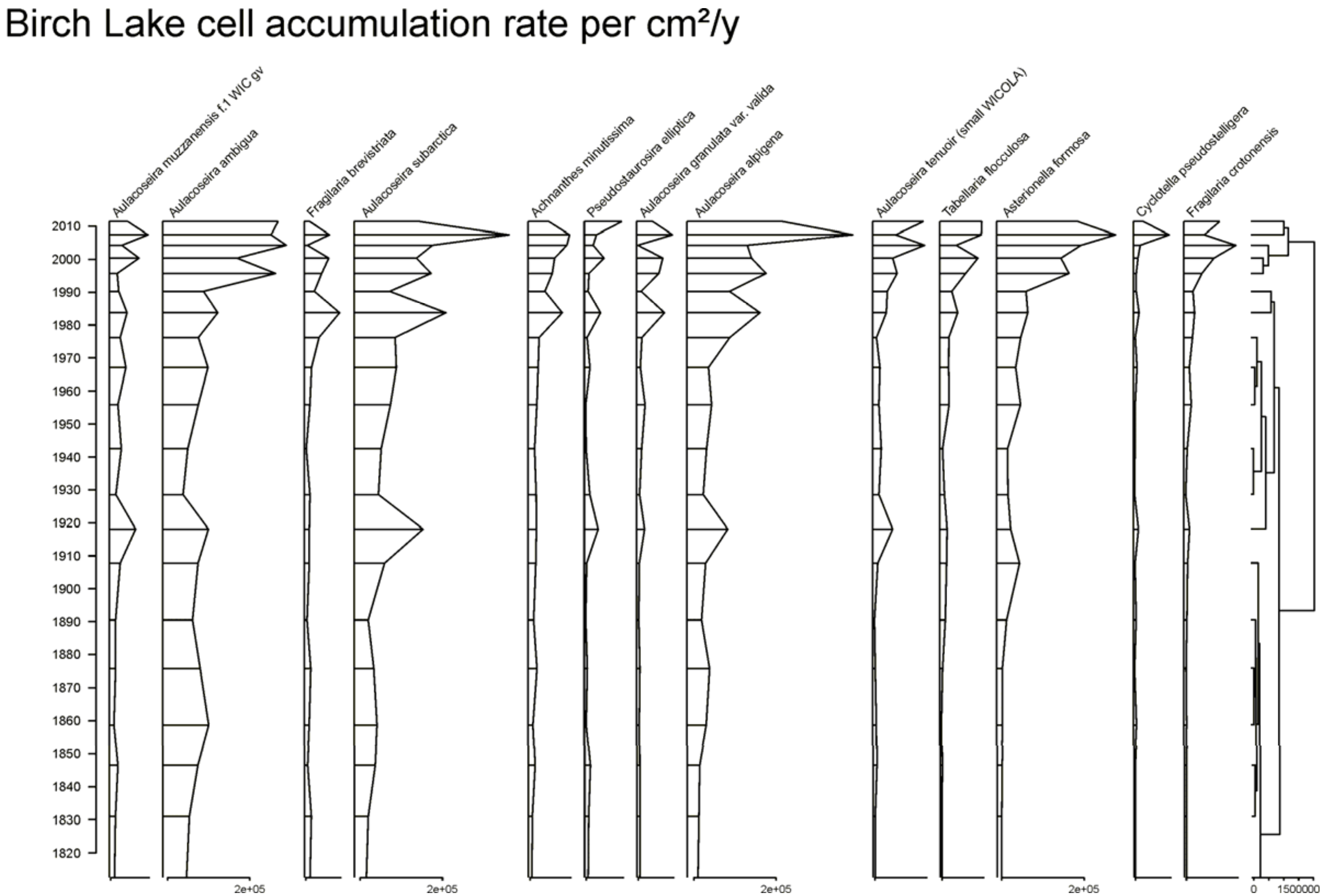

Fig. 9. Downcore accumulation rates of diatom sediment-fossil remains for Birch Lake. Taxa shown in this plot occurred at an accumulation rate of 80,000 cells/cm^2/y in at least one sample. A dendrogram illustrating depth-constrained cluster analysis of the diatom assemblages is plotted on the right.

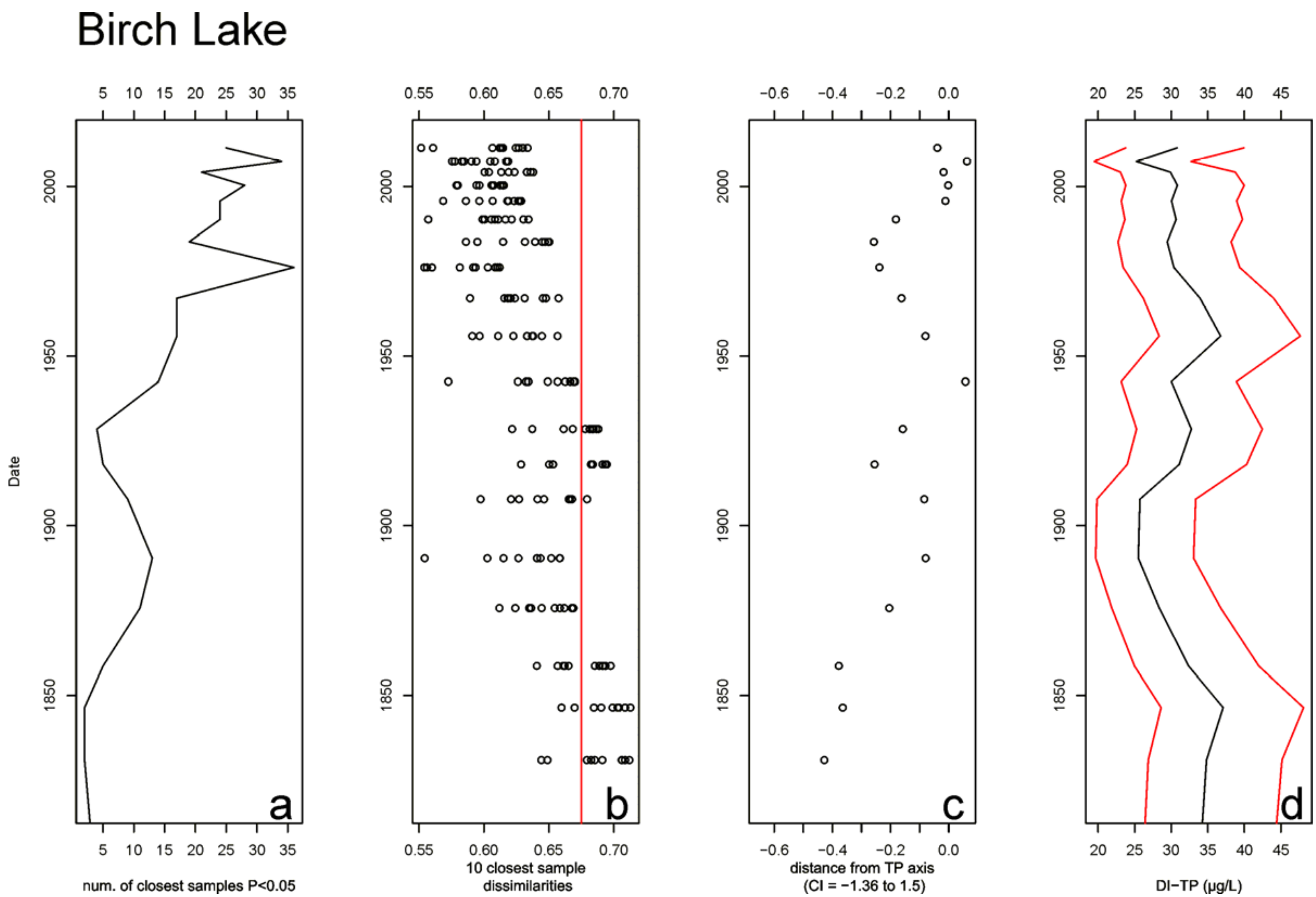

Fig. 10. Diatom assemblage analog, fit-to-TP and DI-TP results for Birch Lake. From left to right stratigraphic plots indicate: (a) the number of significantly close modern phytoplankton sample assemblages to each fossil assemblage based on analog analysis; (b) dissimilarity values for the 10 modern assemblages closest to each fossil assemblage (red line indicates 95th percentile based on all modern samples); (c) distance of each fossil sample from the TP axis in a CCA constrained to TP (lower and upper extremes of the 95% confidence interval shown in axis label); (d) DI-TP for fossil assemblages (black line indicates inferred TP and red lines indicate the range of model error).

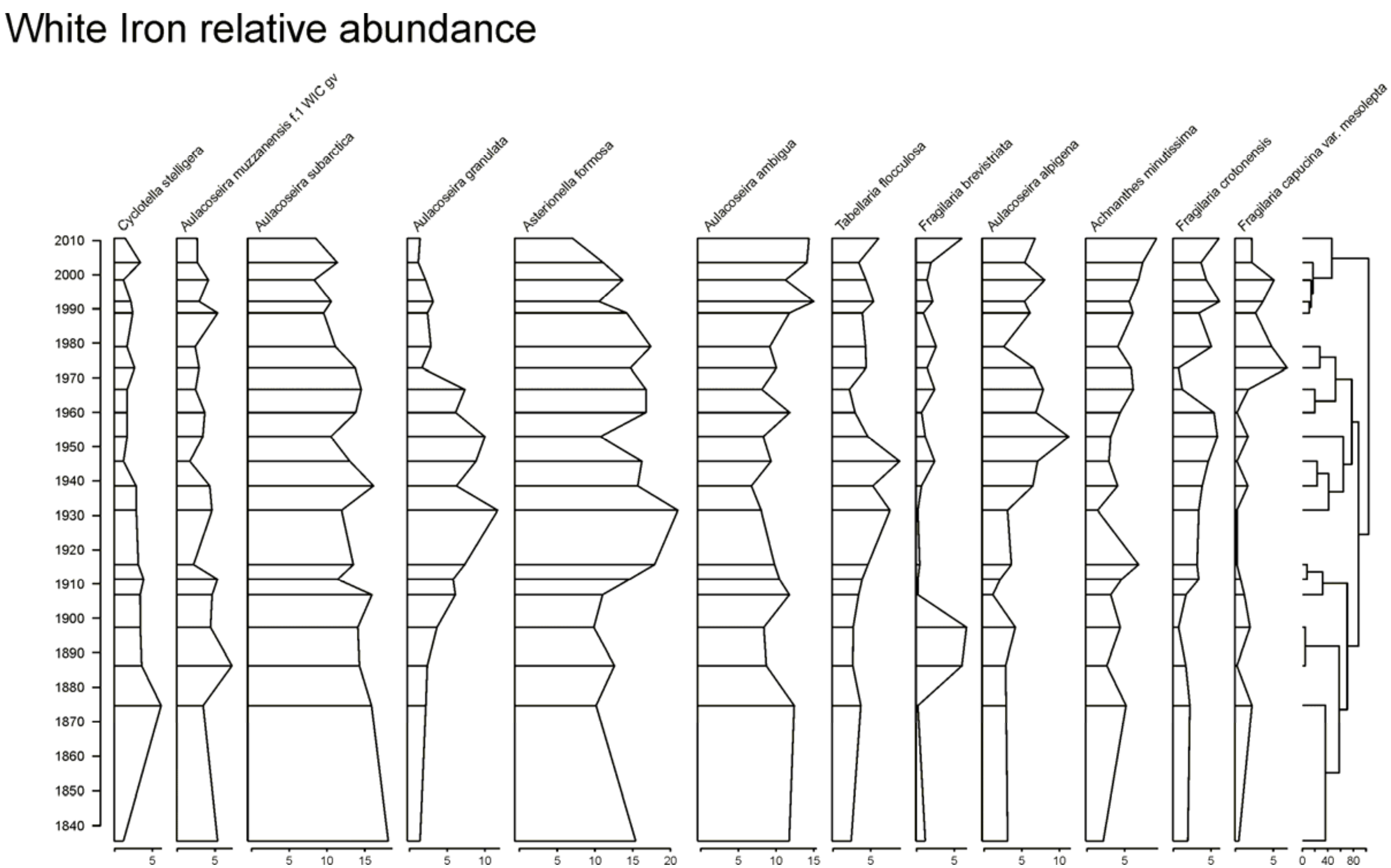

Fig. 11. Downcore relative abundances (%) of diatom sediment-fossil remains for White Iron Lake. Taxa shown in this plot occurred at a relative abundance of at least 5 % in at least one sample. A dendrogram illustrating depth-constrained cluster analysis of the diatom assemblages is plotted on the right.

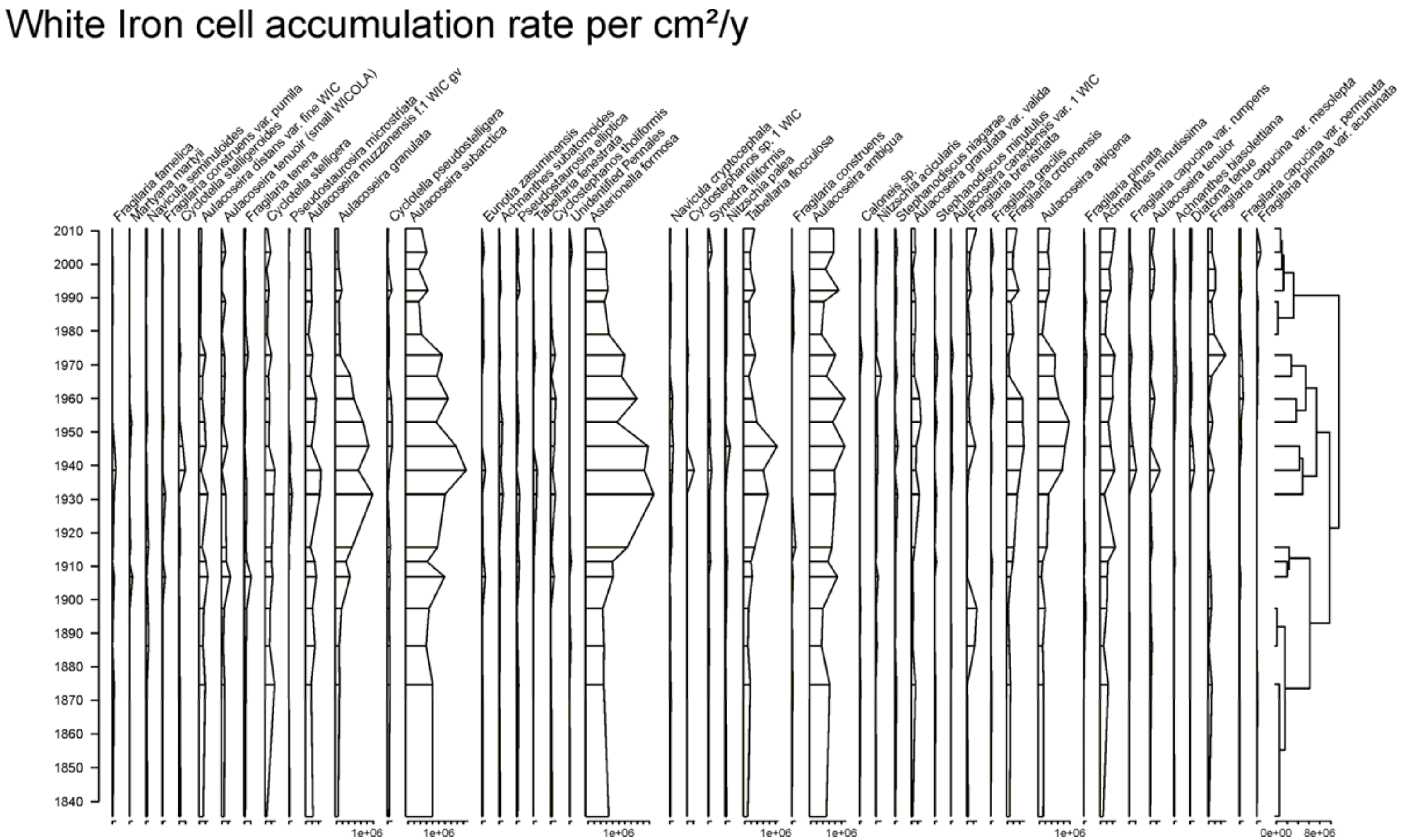

Fig. 12. Downcore accumulation rates of diatom sediment-fossil remains for White Iron Lake. Taxa shown in this plot occurred at an accumulation rate of 80,000 cells/cm^2/y in at least one sample. A dendrogram illustrating depth-constrained cluster analysis of the diatom assemblages is plotted on the right.

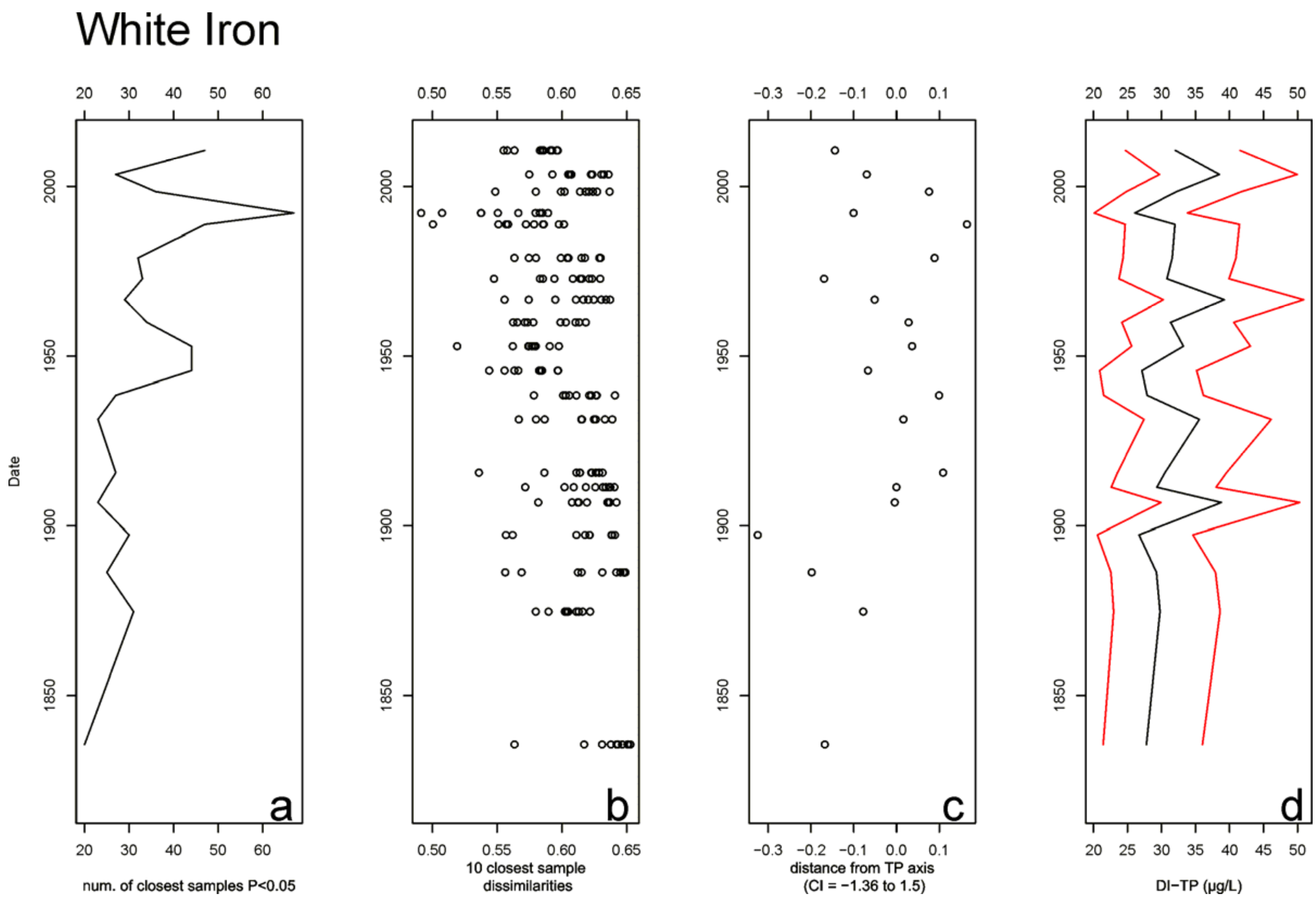

Fig. 13. Diatom assemblage analog, fit-to-TP and DI-TP results for White Iron Lake. From left to right stratigraphic plots indicate: (a) the number of significantly close modern phytoplankton sample assemblages to each fossil assemblage based on analog analysis; (b) dissimilarity values for the 10 modern assemblages closest to each fossil assemblage (all values were within the 95th percentile of modern samples); (c) distance of each fossil sample from the TP axis in a CCA constrained to TP (lower and upper extremes of the 95% confidence interval shown in axis label); (d) DI-TP for fossil assemblages (black line indicates inferred TP and red lines indicate the range of model error).

Farm Lake relative abundance

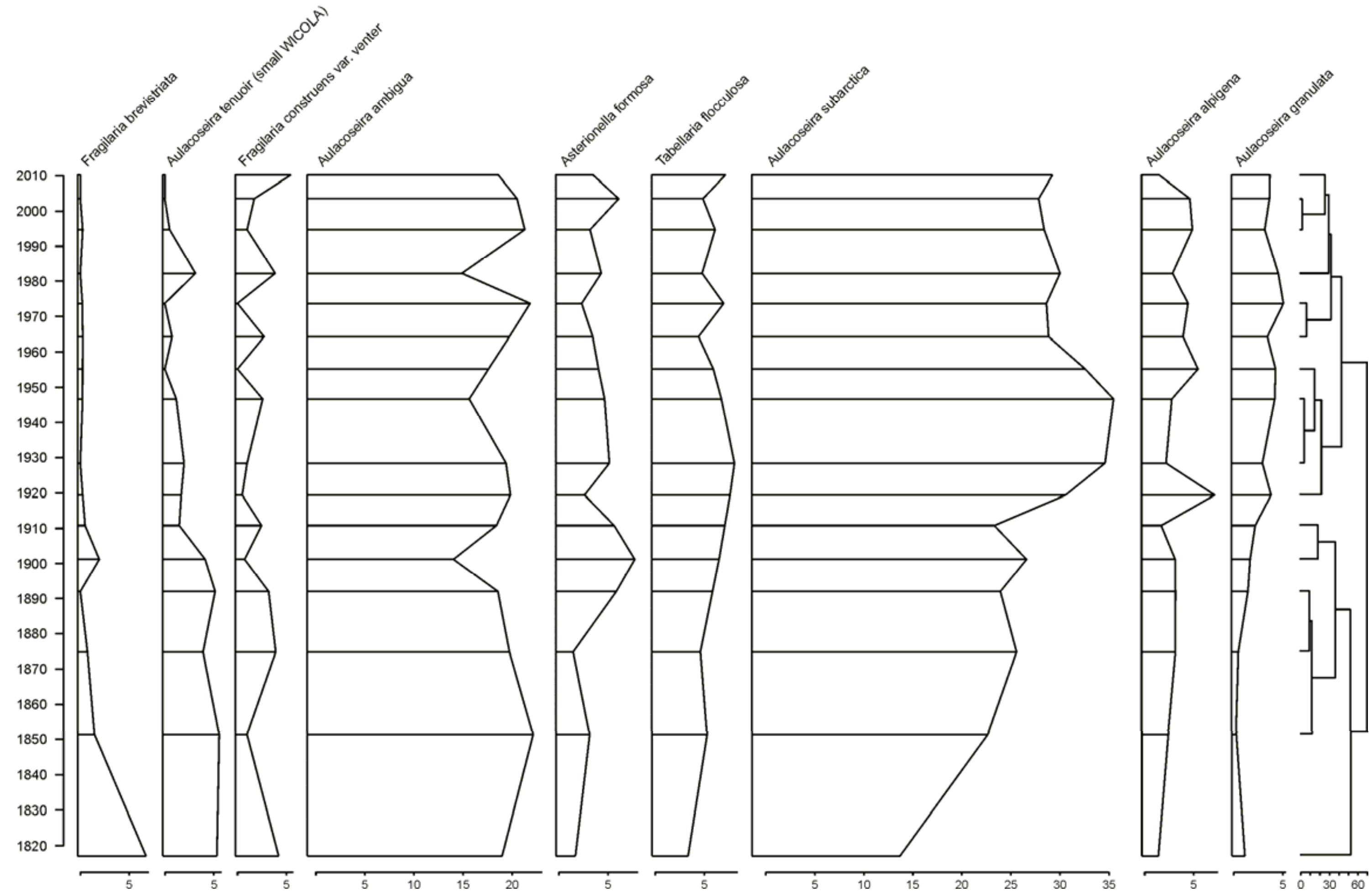

Fig. 14. Downcore relative abundances (%) of diatom sediment-fossil remains for Farm Lake. Taxa shown in this plot occurred at a relative abundance of at least 5 % in at least one sample. A dendrogram illustrating depth-constrained cluster analysis of the diatom assemblages is plotted on the right.

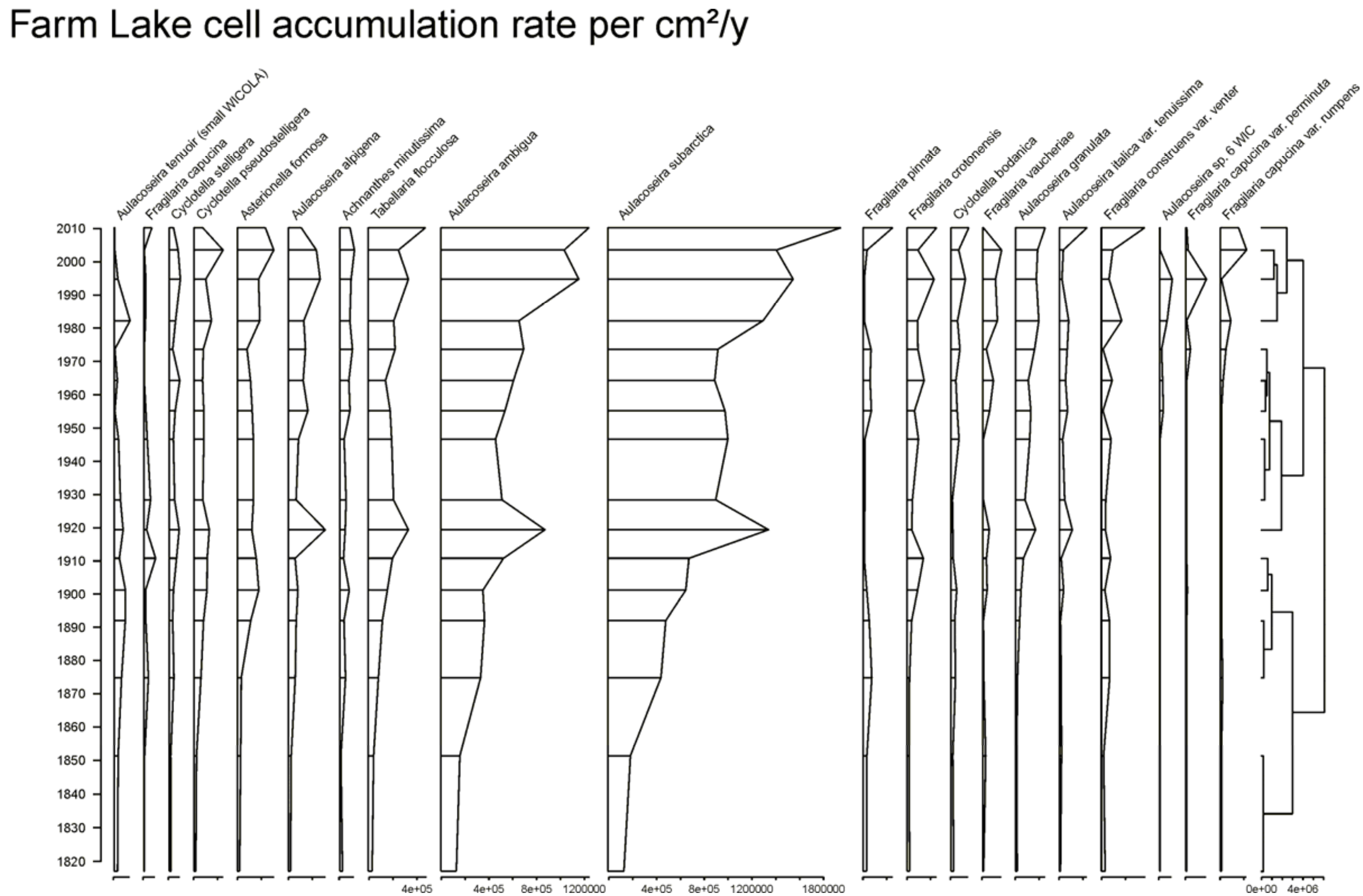

Fig. 15. Downcore accumulation rates of diatom sediment-fossil remains for Farm Lake. Taxa shown in this plot occurred at an accumulation rate of 80,000 cells/cm^2/y in at least one sample. A dendrogram illustrating depth-constrained cluster analysis of the diatom assemblages is plotted on the right.

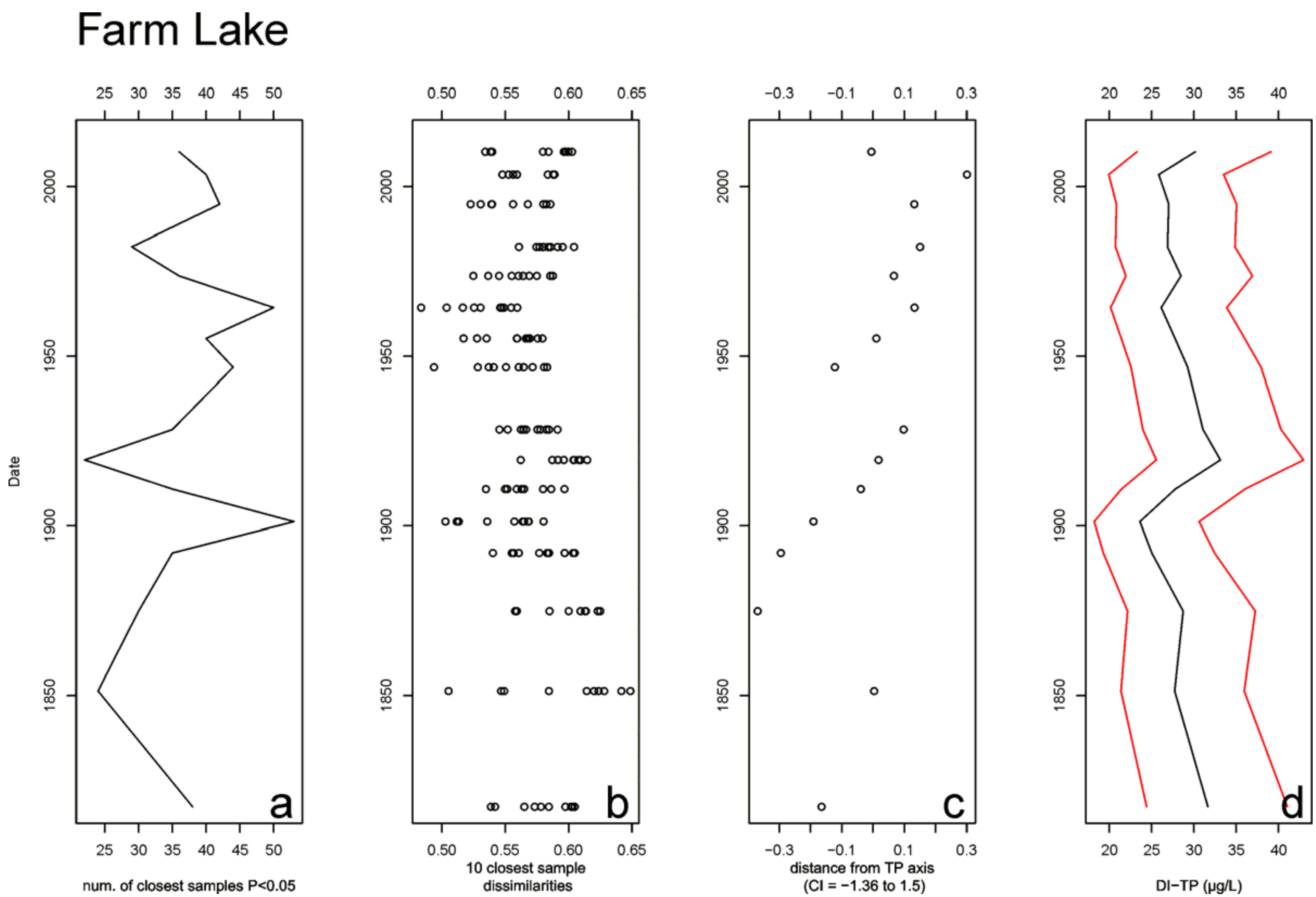

Fig. 16. Diatom assemblage analog, fit-to-TP and DI-TP results for Farm Lake. From left to right stratigraphic plots indicate: (a) the number of significantly close modern phytoplankton sample assemblages to each fossil assemblage based on analog analysis; (b) dissimilarity values for the 10 modern assemblages closest to each fossil assemblage (all values were within the 95th percentile of modern samples); (c) distance of each fossil sample from the TP axis in a CCA constrained to TP (lower and upper extremes of the 95% confidence interval shown in axis label); (d) DI-TP for fossil assemblages (black line indicates inferred TP and red lines indicate the range of model error).

Garden Lake relative abundance

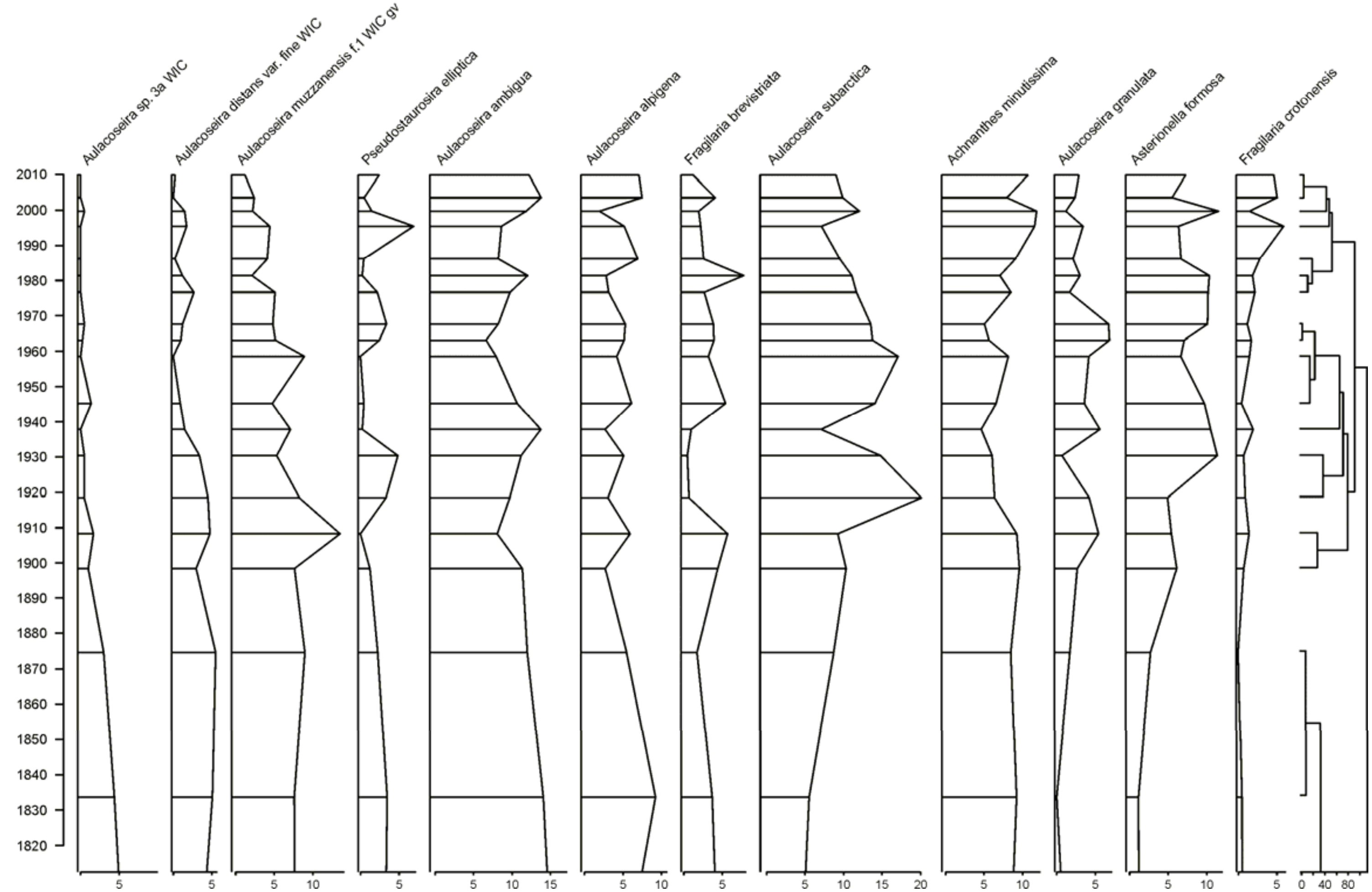

Fig. 17. Downcore relative abundances (%) of diatom sediment-fossil remains for Garden Lake. Taxa shown in this plot occurred at a relative abundance of at least 5 % in at least one sample. A dendrogram illustrating depth-constrained cluster analysis of the diatom assemblages is plotted on the right.

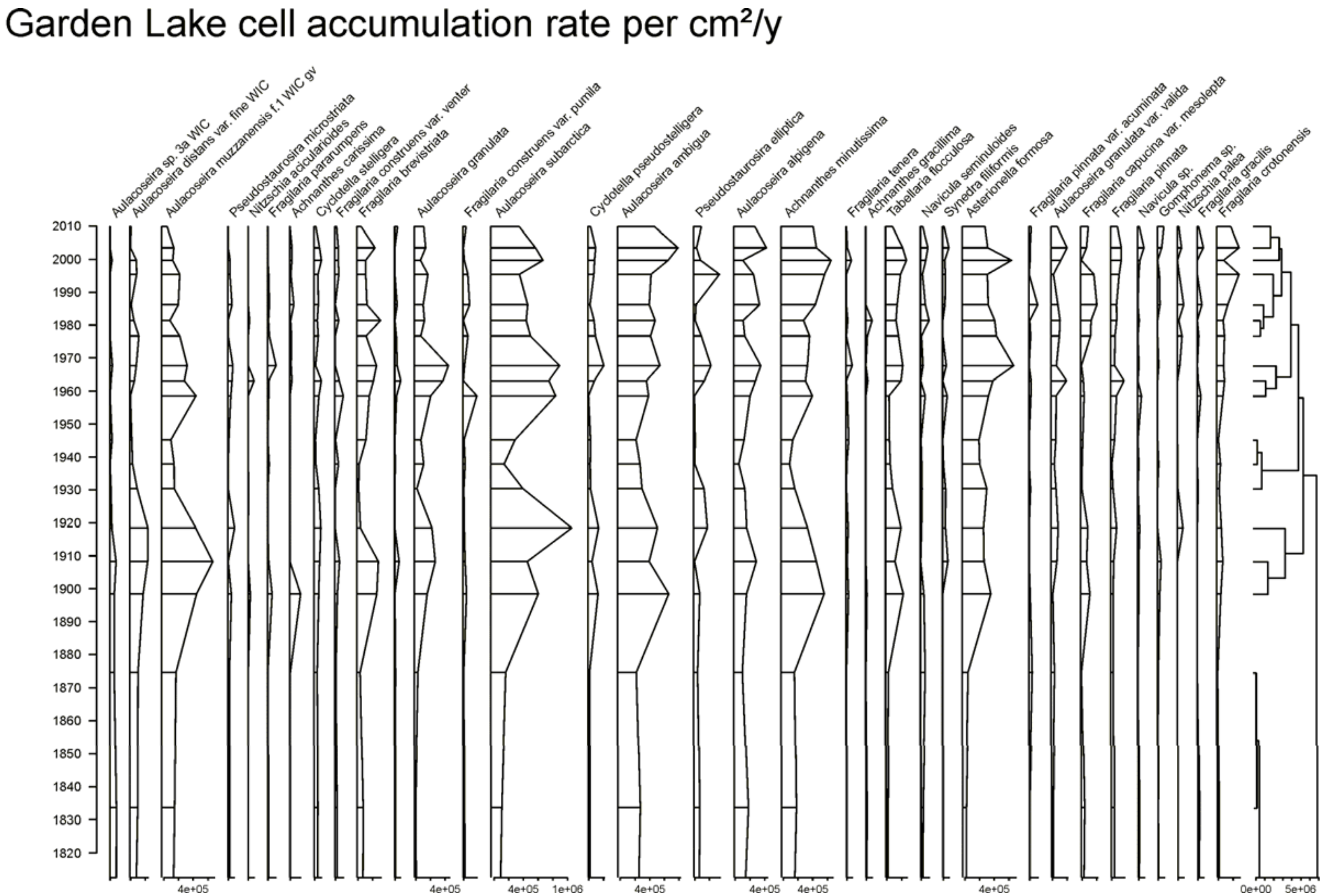

Fig. 18. Downcore accumulation rates of diatom sediment-fossil remains for Garden Lake. Taxa shown in this plot occurred at an accumulation rate of 80,000 cells/cm^2/y in at least one sample. A dendrogram illustrating depth-constrained cluster analysis of the diatom assemblages is plotted on the right.

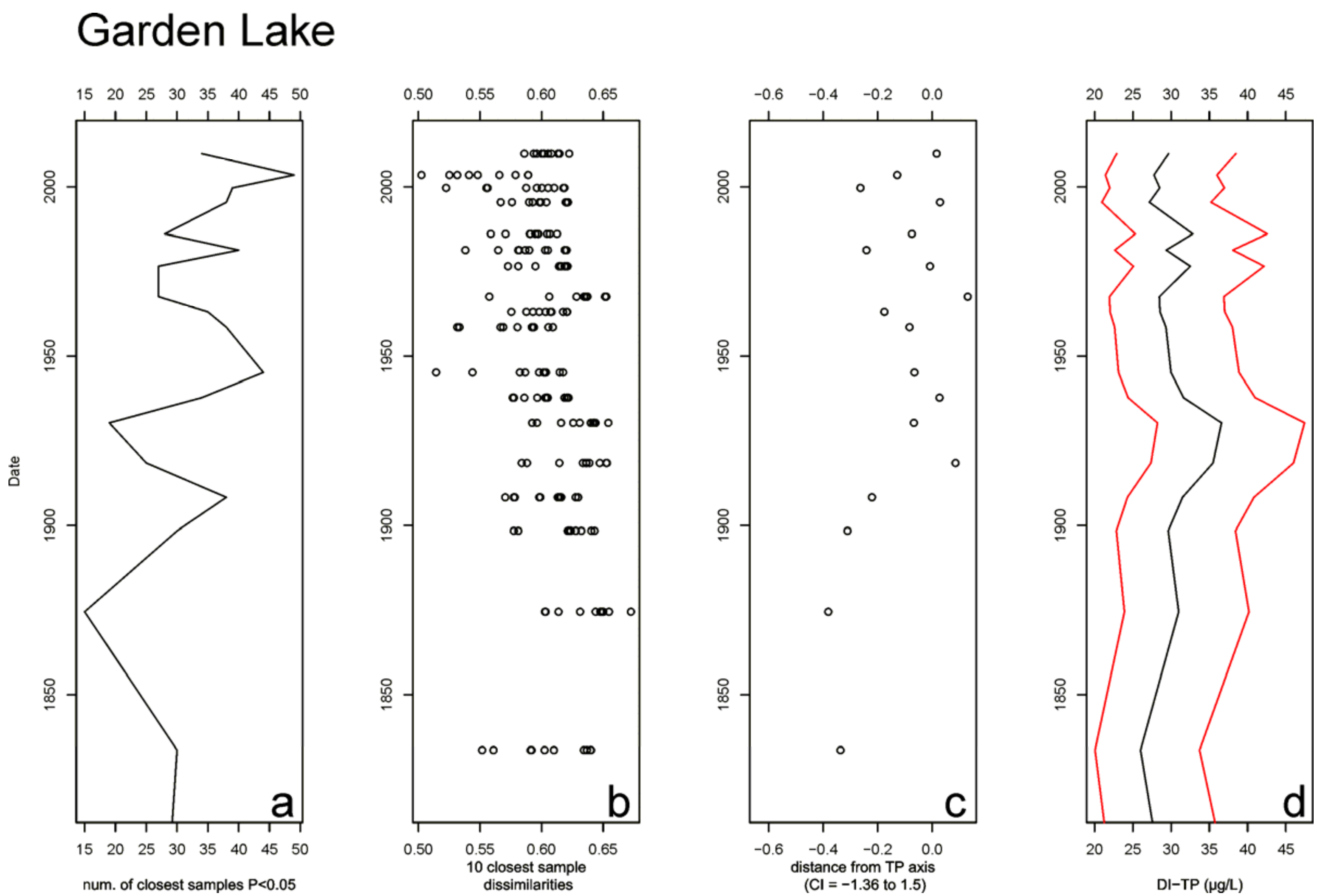

Fig. 19. Diatom assemblage analog, fit-to-TP and DI-TP results for Garden Lake. From left to right stratigraphic plots indicate: (a) the number of significantly close modern phytoplankton sample assemblages to each fossil assemblage based on analog analysis; (b) dissimilarity values for the 10 modern assemblages closest to each fossil assemblage (all values were within the 95th percentile of modern samples); (c) distance of each fossil sample from the TP axis in a CCA constrained to TP (lower and upper extremes of the 95% confidence interval shown in axis label); (d) DI-TP for fossil assemblages (black line indicates inferred TP and red lines indicate the range of model error).

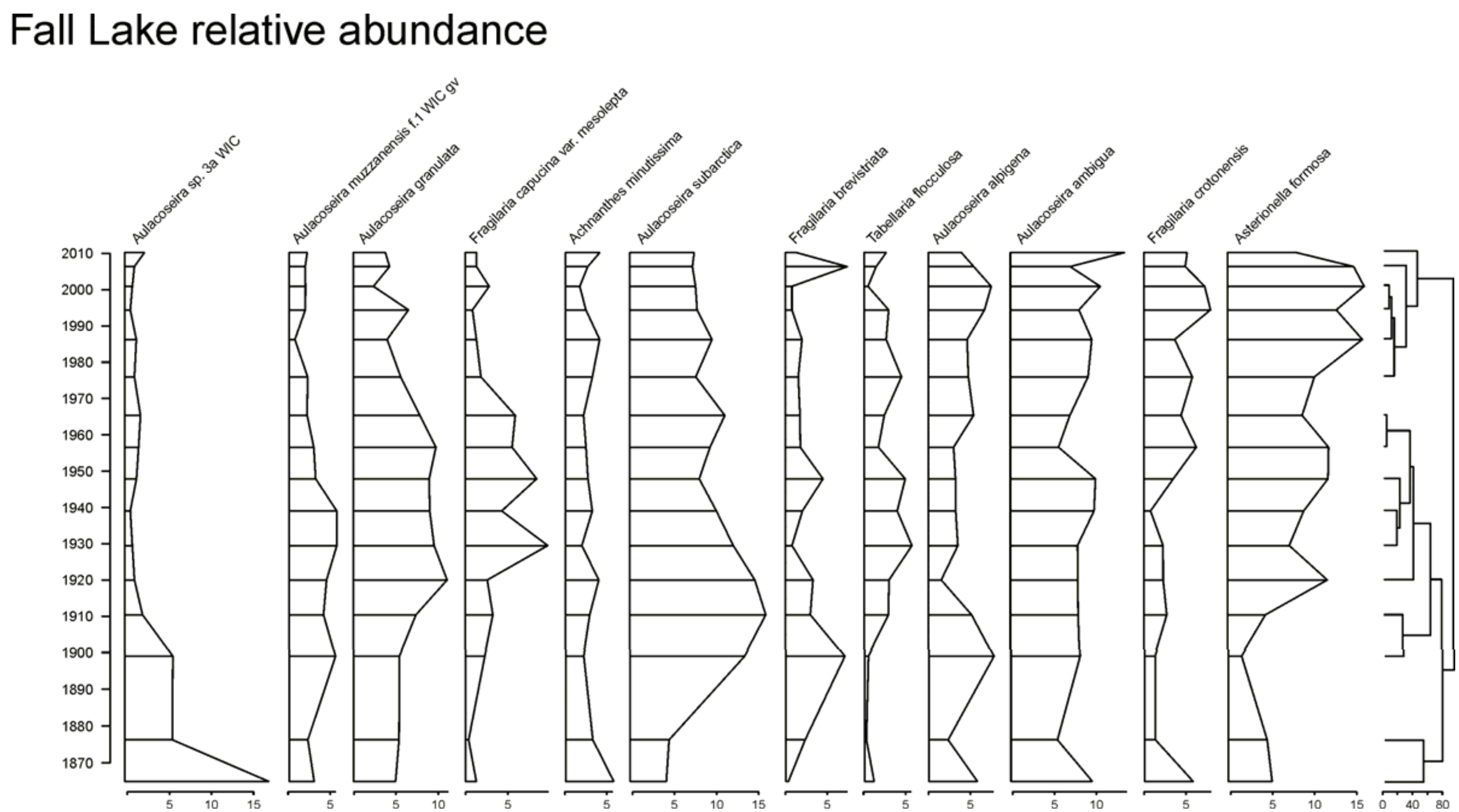

Fig. 20. Downcore relative abundances (%) of diatom sediment-fossil remains for Fall Lake. Taxa shown in this plot occurred at a relative abundance of at least 5 % in at least one sample. A dendrogram illustrating depth-constrained cluster analysis of the diatom assemblages is plotted on the right.

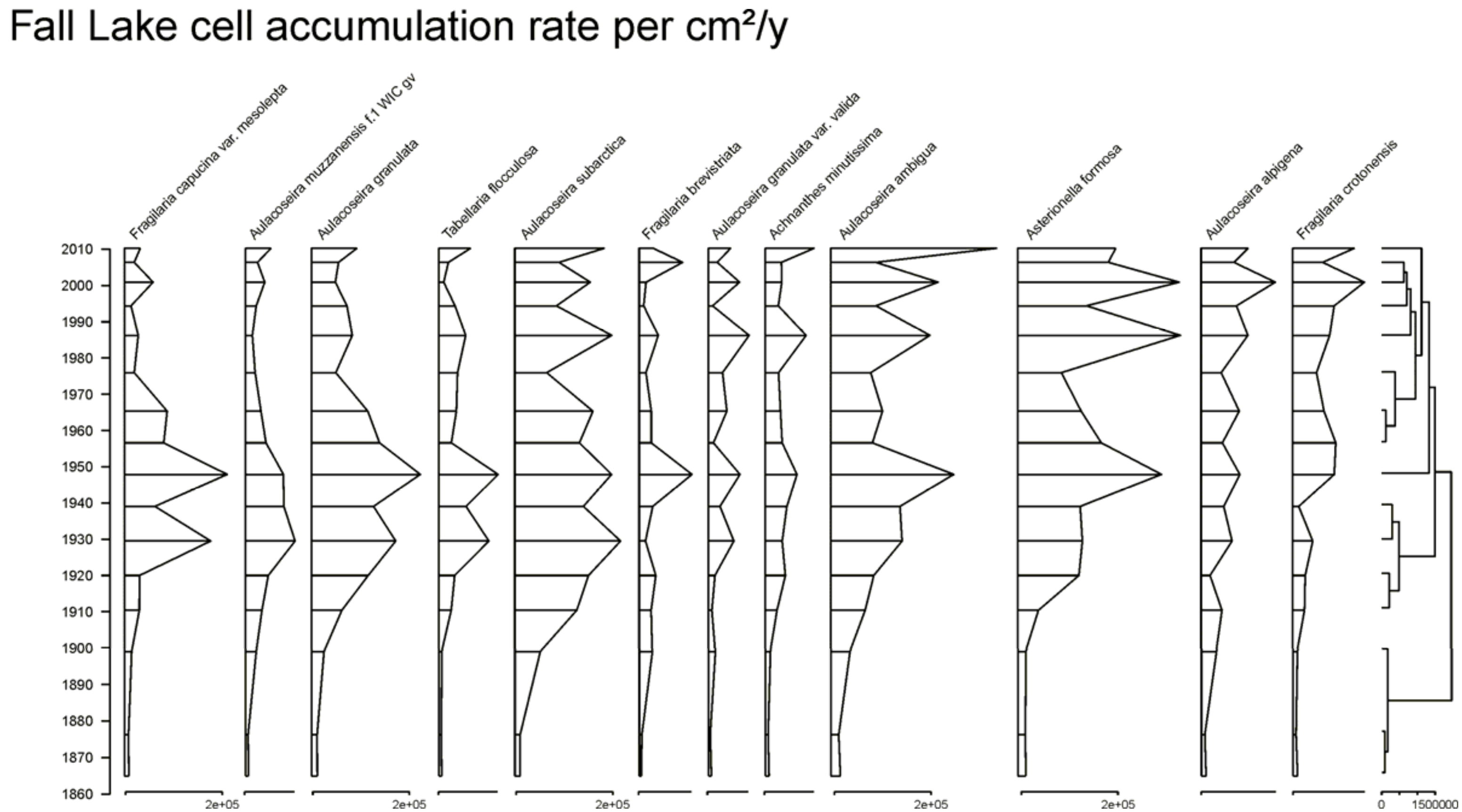

Fig. 21. Downcore accumulation rates of diatom sediment-fossil remains for Fall Lake. Taxa shown in this plot occurred at an accumulation rate of 80,000 cells/cm^2/y in at least one sample. A dendrogram illustrating depth-constrained cluster analysis of the diatom assemblages is plotted on the right.

Fall Lake

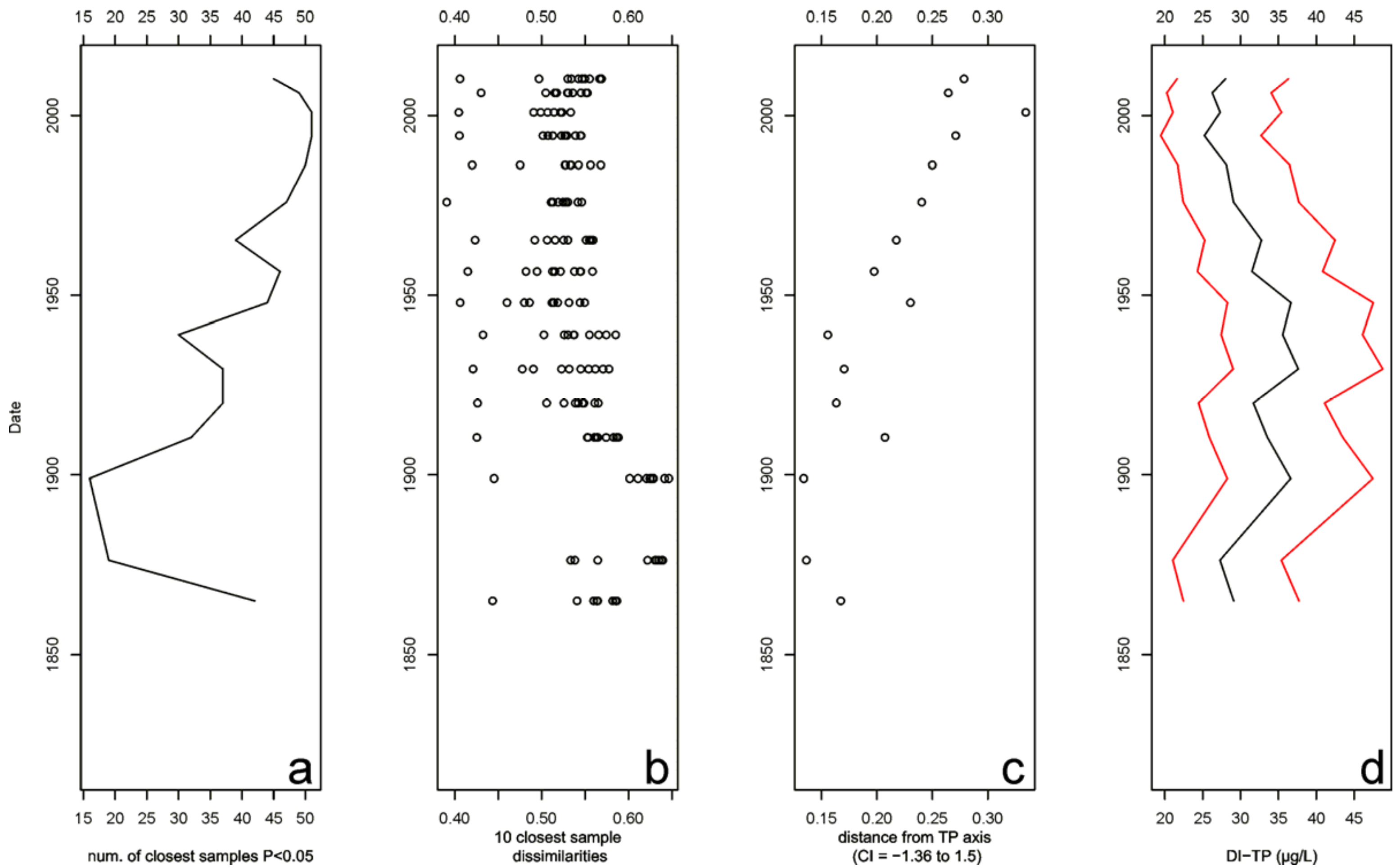

Fig. 22. Diatom assemblage analog, fit-to-TP and DI-TP results for Fall Lake. From left to right stratigraphic plots indicate: (a) the number of significantly close modern phytoplankton sample assemblages to each fossil assemblage based on analog analysis; (b) dissimilarity values for the 10 modern assemblages closest to each fossil assemblage (all values were within the 95th percentile of modern samples); (c) distance of each fossil sample from the TP axis in a CCA constrained to TP (lower and upper extremes of the 95% confidence interval shown in axis label); (d) DI-TP for fossil assemblages (black line indicates inferred TP and red lines indicate the range of model error).