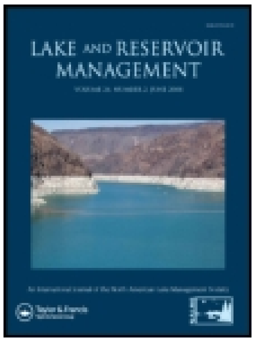

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Total phosphorus changes in New York and New Jersey lakes (USA) inferred from sediment cores

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Abstract

Enache MD, Charles DF, Belton TJ, Callinan CW. 2012. Total phosphorus changes in New York and New Jersey lakes (USA) inferred from sediment cores. Lake Reserv Manage. 28:293–310.

We used sediment diatoms to infer historical changes in total phosphorus (TP) concentration in 26 New York and New Jersey (USA) lakes using a top–bottom approach (i.e., the top of the core represents present conditions and the bottom represents past conditions). Detailed stratigraphic analyses were performed on cores from Greenwood (New Jersey and New York) and Cossayuna lakes (New York). TP reconstructions were performed using 2 weighted-averaging partial-least-squares (WA-PLS) transfer functions: (1) an interregional 278-lake calibration set from northeastern United States (NE-US model: $r^2_{boot} = 0.69$, root mean square error of prediction RMSEP) = 1.8 μg/L); and (2) a regional 33-lake (callibration set from New Jersey and New York (NJ-NY model: $r^2_{boot} = 0.54$, RMSEP = 1.5 μg/L). The NJ-NY model provided better estimates for modern TP but failed to provide reliable estimates for low TP values and reliable modern analogs for half of the bottom samples. Low TP concentrations were better inferred by the NE-US model, which included a higher number of oligotrophic lakes. Average change for all lakes was an increase of 2 to 7 μg/L TP. Greenwood and Cossayuna lakes inferred TP concentrations have increased up to 21 μg/L, presumably as a result of post-settlement anthropogenic activities. The inferred TP temporal changes provide important insight on the magnitude of cultural eutrophication. The use of 2 different inference models demonstrates the advantage of using a regional versus a larger-scale inference model in estimating the degree of change in historical lake TP. Careful interpretation of TP reconstructions can be used to provide reliable estimates of cost-effective targets for lake restoration programs.

Key words: diatoms, lakes, New Jersey, New York, nutrients, phosphorus, sediment cores, transfer functions

New Jersey and New York are 2 of the most populated states on the east coast of the United States. Agriculture, industry, residential development, and recreational activities are all sources of stress on lacustrine ecosystems. Cultural eutrophication has become a major issue for surface waters from the east coast (Ponader et al. 2007). Because the instrumental records exist only for the last few decades, after the aquatic systems have been impacted, the baseline conditions and the degree of impairment induced by human activities are unknown, making the management of water quality a challenging task (Smol 1992). Lake nutrients are a major source of impairment in New Jersey and New York lakes, so knowledge of past conditions is essential for making cost-effective management decisions and designing restoration programs (Bennion et al. 1996, Battarbee 1999, Battarbee et al. 2005).

In the absence of instrumental records, lake sediments constitute a valuable source of information on the extent and timing of anthropogenic impacts on current ecological conditions in lakes. Paleolimnological techniques are a powerful tool for lake managers who must assess and protect water quality of lakes affected by multiple stressors (Hall and Smol 1999, Reavie et al. 2000, Bennion et al. 2001, Bradshaw and Anderson 2001).

Diatoms are one of the most powerful paleolimnological indicators of water quality. They are ecologically diverse, and their silica wall makes them identifiable to the levels of

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species and variety. Variations in fossil diatom abundance and species composition can be used to assess the amount of change that has occurred in lake systems through time (Smol and Stoermer 2010).

A fast way to quantify changes that have affected lake systems since European settlement is to examine how much the sediment diatom assemblages differ between the top (representing present-day conditions) and the bottom of a sediment core (i.e., top–bottom approach). Core bottoms can represent predisturbance conditions when cores are of sufficient length (e.g., >35 cm for lakes from the northeastern United States; Dixit et al. 1999). In this case, the top–bottom approach provides "snap-shots" of environmental conditions before and after human impacts and has proven successful in addressing diverse environmental issues such as the impact of acid rain, eutrophication, and global warming (Cumming et al. 1992, Dixit et al. 1999, Smol et al. 2005).

Although paleolimnological studies have investigated long-term changes in nutrient or pH status of some lakes in New Jersey and New York (e.g., Charles et al. 1990, Sebetich and Messaros 1993, Dixit and Smol 1994, Dixit et al. 1999), little is known about changes in lake total phosphorus (TP) at a larger scale.

This study had 2 main goals: (1) use a diatom-based top–bottom approach to examine the magnitude of change in TP concentration between modern and past conditions in 26 New York and New Jersey lakes; and (2) determine the timing and potential causes of decadal-scale TP changes based on stratigraphic cores from Greenwood and Cossayuna lakes, 2 lakes known to have changed due to human activities typical of the region. To reach these goals, 2 diatom-based TP inference models were developed: one based on a large-scale, interregional calibration set of lakes in the northeastern United States and another limited to lakes in New Jersey and New York. Here we discuss the ability of the 2 inference models to reconstruct historical TP changes and implications for lake managers.

Study sites

The study lakes are located throughout New York and New Jersey (Fig. 1). In addition, we used data on surface

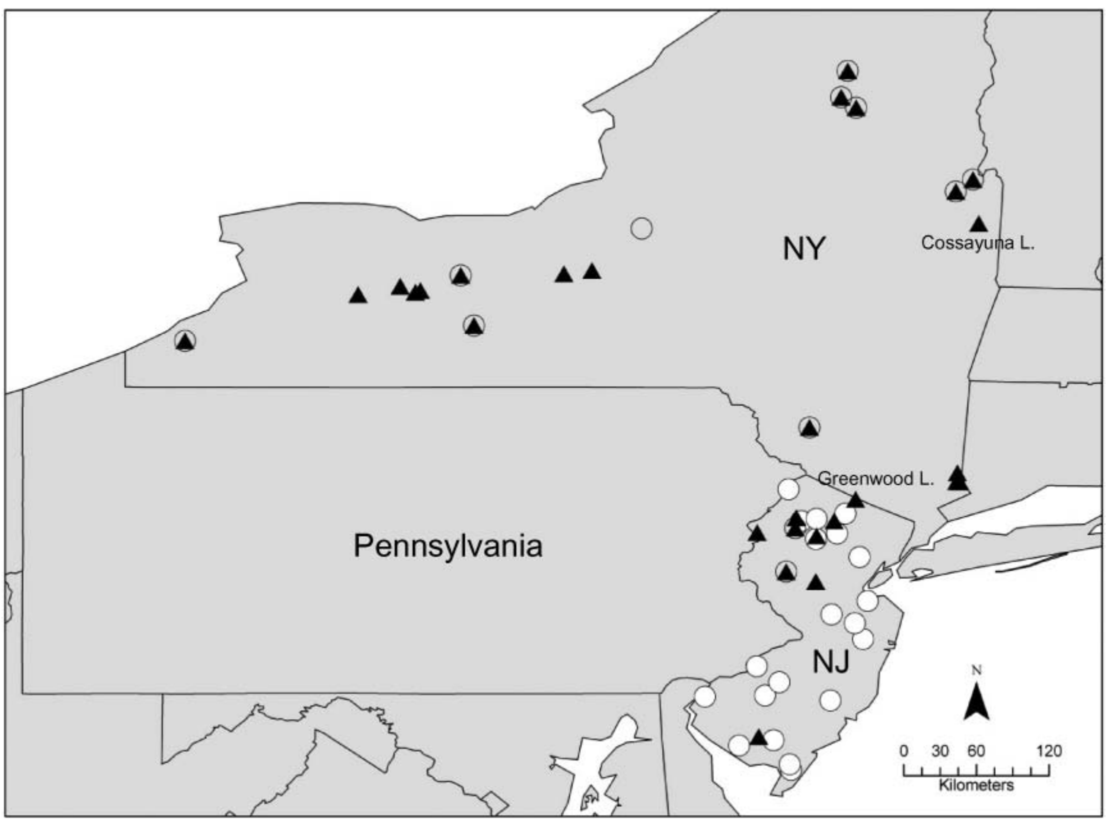

Figure 1.-Location of study lakes in NY and NJ. White-filled circles represent NJ calibration lakes; grey-filled circles represent NLA calibration lakes; triangles represent top–bottom lake. Greenwood and Cossayuna are stratigraphic lakes.

Table 1.-Measured limnological variables of lakes from NJ and NY used in the calibration set and the top–bottom and stratigraphic studies. Asterisk indicates lakes listed as impaired by EPA. Lakes included in the top–bottom and stratigraphic studies are listed in Table 2. Cond = conductivity; TKN = total Kjeldahl nitrogen; TDN = total dissolved nitrogen.

Lake Name	State	Maximum depth (m)	Secchi disk (m)	pH	Cond (umhos/cm)	TP (μg/L)	TKN (μg/L)	TDN (μg/L)
A. Clemente Inc. Pond	NJ	7.5	2.2	7.2	196.0	19.4	500.0	
Anawana	NY		3.5	7.45	90.0	9.0		281.0
Bennets Pond	NJ			6.1		39.1		427.3
Brainerd	NJ	1.9	1.0	6.4	190.0	66.3		849.7
Campbells Pond	NJ	1.8	7.1	5.0	827.0	59.7	500.0	
Canadice*	NY	24.3	6.2	7.5	154.0	8.5	200.0	
Canandaigua	NY	83.5	7.1	7.9	367.0	7.0		390.0
Cassadaga	NY		2.2	8.1	294.0	15.0		367.0
Cedar	NJ			7.3		77.0		1054.7
Cedar 2	NJ			6.9		23.2		439.7
Champlain	NY		0.8	7.7	196.0	2.1		246.0
Chesler	NJ	7.0	2.7	8.3	499.0	11.5	300.0	
Clint Millpond	NJ	1.4	0.5	4.3	77.0	59.0	800.0	
Conesus*	NY	18.3	3.5	8.0	326.0	17.8	500.0	
Cooper River	NJ	1.4	0.5	7.5	269.0	147.4	900.0	
Cossayuna*	NY	7.5		8.1	174.0	35.7		607.5
Crystal Spring	NJ	0.7	0.2	7.0	112.0	198.0	2400.0	
Cumberland Pond	NJ	1.1	1.1	4.4	42.0	45.0	300.0	
Delaware	NJ	3.7	0.9	8.3	439.0	114.3	1400.0	
Dennisville	NJ	1.1	1.1	5.7	71.0	18.5	300.0	
Duck Pond	NJ	1.2	1.2	6.5	105.0	33.6	700.0	
Echo	NJ	8.8	1.4	7.1	142.0	31.8	700.0	
Flamingo	NJ	1.0	0.8	6.6	106.0	21.0	500.0	
Gardners Pond	NJ	11.3	3.0	7.9	400.0	17.3		160.7
Green Pond*	NJ	4.8	3.8	7.2	77.0	17.2	200.0	
Greenwood*	NJ	5.5	1.8	7.5	185.0	28.6	500.0	
Harrisville	NJ	1.0	0.8	4.3	36.0	17.3	200.0	
Hemlock*	NY	24.4	4.6	8.1	210.0	9.7	300.0	
Japanese Garden	NJ	1.2	0.5	7.4	228.0	7.0	1000.0	
Keuka*	NY	55.8	7.3	7.9	300.0	6.0		236.0
Kittatinny Camp	NJ	1.6	0.7	7.6	226.0	40.0	700.0	
Lefferts	NJ	2.9	2.3	6.4	209.0	15.5	300.0	
Little Simon Pond	NY		8.4	7.3	30.0	2.0		551.0
Long Pond	NY		3.7	7.5	29.0	5.0		191.0
Millhurst Pond	NJ	0.9	0.8	6.94	202.0	35.6	400.0	
Muckshaw Pond	NJ	2.4	2.4	7.57	348.0	16.6	500.0	
Oneida	NY	16.8		7.37	359.0	23		333.0
Oscaleta	NY	10.8		7.81	127.0	19.1	0.0	573.9
Otisco	NY	17.9	4.2	8.21	309.0	11.7	300.0	
Owasco*	NY	48.9	4.7	8.41	305.0	7.6	400.0	
Park	NY			7.1	39.0	4.0		219.0
Peach*	NY	7.1		8.08	240.0	34.3		478.3
Round Valley*	NJ		2.5	7.5	20.0	3.5		282.0
Saginaw	NJ	3.7	1.9	7.5	479.0	20.3	500.0	
Silver*	NY	11.0		8.0	211.0	29.4		807.1
Sly Pond*	NY		6.8	6.6	42.0	3.0		166.0
Tranquility	NJ		1.0	8.6	404.0	34.0		729.0
Union	NJ	3.5		5.7	141.0	23.0	500.0	
Waccabuc	NY	14.1		8.0	158.0	16.8		764.2

sediment samples (0–1 cm) that were collected from 245 lakes in the northeastern United States (Northern and Southern Appalachian Plateaus, and Coastal Plain ecoregions) as part of the United States Environmental Protection Agency's (EPA) National Lake Assessment (NLA). The NLA lakes were sampled for both surface sediment samples and water physical–chemical parameters (for details on sample selection and sampling methods see www.epa.gov/owow/lakes/lakessurvey/). Surface samples from an additional 33 New Jersey and New York lakes were collected for a paleolimnological pilot project conducted by the Academy of Natural Sciences. The New Jersey lakes were statistically selected by the New Jersey Department of Environmental Protection (NJDEP) (similar to the NLA approach, see link above) and are periodically sampled; the New York lakes were selected by the New York Department of Environmental Conservation (NYDEC) because of their nutrient impairment status and related management issues. Top–bottom cores were collected from 26 lakes, 11 in New Jersey and 15 in New York. Greenwood and Cossayuna lakes were selected for stratigraphic investigation because of existing historical records, their importance as recreation resorts, and their impaired status. The main sources of anthropogenic impact on the lakes have included logging, agriculture, dam construction, industry and related atmospheric pollution, residential development, tourism, and recreation activities (see Table 1 for lakewater characteristics).

Cossayuna Lake, New York (43°13′02″; 73°25′011″), is a shallow lake (max. depth ~8 m, mean depth 3 m), 4.5 km in length, with a surface and watershed area of 2.6 km^2 and 7.4 km^2, respectively. Critical impairment problems are related to eutrophication, oxygen depletion, and loss of habitat for fish. The lake experiences seasonal algal blooms and plant growth; invasive Eurasian water milfoil was introduced in mid-1970s (Cadmus Group 2008).

The area around Cossayuna Lake was settled in 1765. Soon thereafter, 3 dams were erected at the outlet to develop water power. Later, paper mills and farming became the backbone of the economy of the region. Cossayuna Lake was famous for its bass fishing, and 100 years after the first settlers arrived it became a well-known recreation resort. Today, invasive Eurasian watermilfoil is a major management challenge, and annual harvesting plans have been implemented (LA Group, PC 2001).

Greenwood Lake (41°11′20″; 74°79′20″) is situated at the border of New Jersey and New York. The lake is ~15.4 km long and 1.9 km wide with surface and watershed areas of 7.7 km^2 and 64.9 km^2, respectively. The northern basin has steep slopes and a maximum depth of 18 m. The southern basin has a maximum depth of 3 m and an extensive littoral area. Similar to Cossayuna, Greenwood Lake has experienced increased deterioration of environmental conditions, including eutrophication, oxygen depletion, loss of fish habitat, algae blooms, and nuisance aquatic vegetation (GLC-PRC 2011).

Deterioration of Greenwood Lake water quality was reported as early as 1951, with nuisance densities of macrophytes, blooms of *Spirogyra*, and summer depletion of dissolved oxygen in the hypolimnion. A water level management program was implemented in 1997 to enhance water quality, to control aquatic weeds, and to improve the lake's overall environmental conditions (NJDEP 1997).

Methods

Coring

Surface sediment samples from New Jersey lakes were collected between 1996 and 2007 by NJDEP field crews using a Glew mini-corer (Glew 1991; Fig. 1). Sediment cores used for top–bottom and stratigraphic diatom analyses were collected from the deepest basin of the study lakes during the summer and fall of 2007 by NJDEP and NYDEC. The cores were collected using a Glew-modified K-B corer (Glew 1989) and extruded in the field using a Glew (1988) extruder. Surface sediment samples (0–1 cm) from EPA's NLA lakes were collected during the summer of 2008 from 245 lakes, also using Glew coring devices (National Lake Assessment,http://www.epa.gov/owow/lakes/lakessurvey/).

Diatom analysis

All samples were processed and analyzed following Charles et al. (2002; see also http://diatom.ansp.org/nawqa/protocols.asp). A minimum of 500 diatom valves were identified for each surface sediment sample, 400 for the top–bottom samples, and 300 valves per sample for the stratigraphic cores. Species were identified to the lowest practical taxon (usually species or variety).

Chronology

Ten samples were measured for ^{210}Pb activity by gamma spectrometry for the stratigraphic core from Cossayuna Lake and 12 for Greenwood Lake. All samples were measured in Dr. Richard Bopp's laboratory (Rensselaer Polytechnic Institute, http://ees2.geo.rpi.edu/bopp.html). He provided both Constant Initial Concentration (CIC; Appleby and Oldfield 1978) and constant rate of supply (CRS; Binford 1990) chronologies. Greenwood Lake's CRS and CIC chronologies were in relatively good agreement with the ^{137}Cs peak of 1963. The CIC model did not provide dates in agreement with evidence for a dam built in 1836, identified by a change in sediment color and a shift in diatom species consistent with water level rise; thus, the CRS model dates were used for this core. A mixed interval of 3.5 cm was identified in the top of the Cossayuna Lake core, so the CIC model, which allows mixing, was applied to these core intervals. Because the CIC model does not accommodate increasing sediment accumulation rates with core depth, the chronology below 3.5 cm was established using the CRS model, and these dates were interpolated with the top CIC dates (Fig. 2).

No date information was available for the cores used for the top–bottom approach. Previous studies of lakes in the northeastern United States showed that sediments below 35 cm represent the preindustrial era (∼1850; Dixit et al. 1999), but most of these were lower productivity lakes with lower sedimentation rates than many lakes included in this study. Because our cores were not dated, and several were <35 cm long, we prefer to use the term "historical" for the core bottom interval instead of preindustrial or pre-European settlement (= predisturbance).

Statistical analyses

TP transfer functions were developed using weighted-averaging partial-least-squares (WA-PLS) models (ter Braak and Juggins 1993, Birks 1998). Species and TP data were log-transformed. Two models were developed. The first model (NE-US) was based on a northeastern United States inter-regional (Northern and Southern Appalachian Plateaus, and Coastal Plain ecoregions) dataset derived from 245 NLA lakes and an additional 33 lakes from New Jersey and New York. TP in the NE-US dataset varied from 0.9 to 323 μg/L (mean = 28.7, SD = 47.9 μg/L). After exclusion of outliers, the NE-US TP transfer function was derived from 215 lakes and includes 200 diatom species that occur with >1% relative abundance in at least one sample. The second TP inference model (NJ-NY) was limited to lakes from New Jersey and New York and was based on only 33 calibration sites. TP range in these lakes varied from 7 to 198 μg/L (mean = 40.8, SD = 41 μg/L). It includes 159 diatom species occurring >1% in at least one sample. Modern Analog Technique (MAT) was used to identify fossil samples with poor analogs in modern datasets of the 2 transfer functions. Inference models and MAT were developed and applied using the computer program C^2 (Juggins 2003).

We used ANOVAs and paired t-tests to determine if statistically significant differences ($p < 0.05$) in species relative abundances were present in modern versus historical times. Because of the high number of absences, we could not compute ANOVAs at species level except for *Fragilaria crotonensis*. Consequently, we grouped diatom species in higher taxonomic levels (e.g., *Navicula* spp., *Nitzschia* spp.); we also grouped centric diatom species, known for their mostly planktonic lifestyle. To ensure normal distributions, species relative abundances were either log-transformed (*Navicula* spp.) or ranked (centric diatoms and *Fragilaria crotonensis*). Bivariate analyses were performed to analyze the fit between the TP reconstructions obtained by the 2 inference models described above. Bivariate analyses, ANOVAs and t-tests were performed using the program JMP v. 7.0 (SAS Institute Inc.).

Results

Historical changes in top–bottom lakes

Diatom assemblages

About 500 diatom species were identified in top and bottom sediment samples from the 26 study lakes. The most

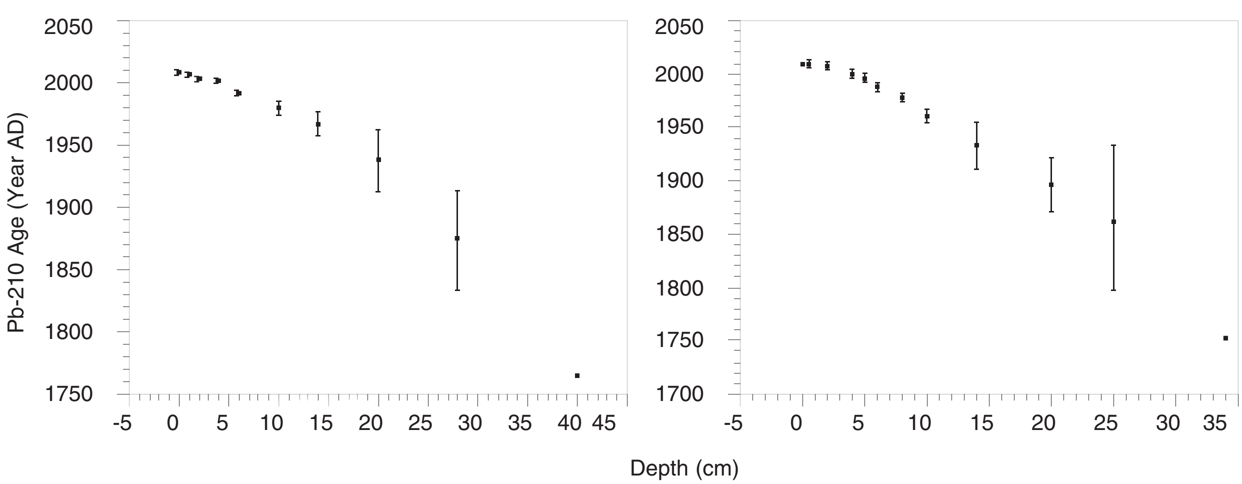

Figure 2.-Age–depth relationship based on estimates from the CRS models in the stratigraphic cores from (a) Cossayuna and (b)Greenwood Lake. A CIC model was applied to a mixed 3.5 cm interval of the top of the core from Cossayuna Lake; the chronology below 3.5 cm was established using the CRS model by interpolation with the top CIC dates (see text).

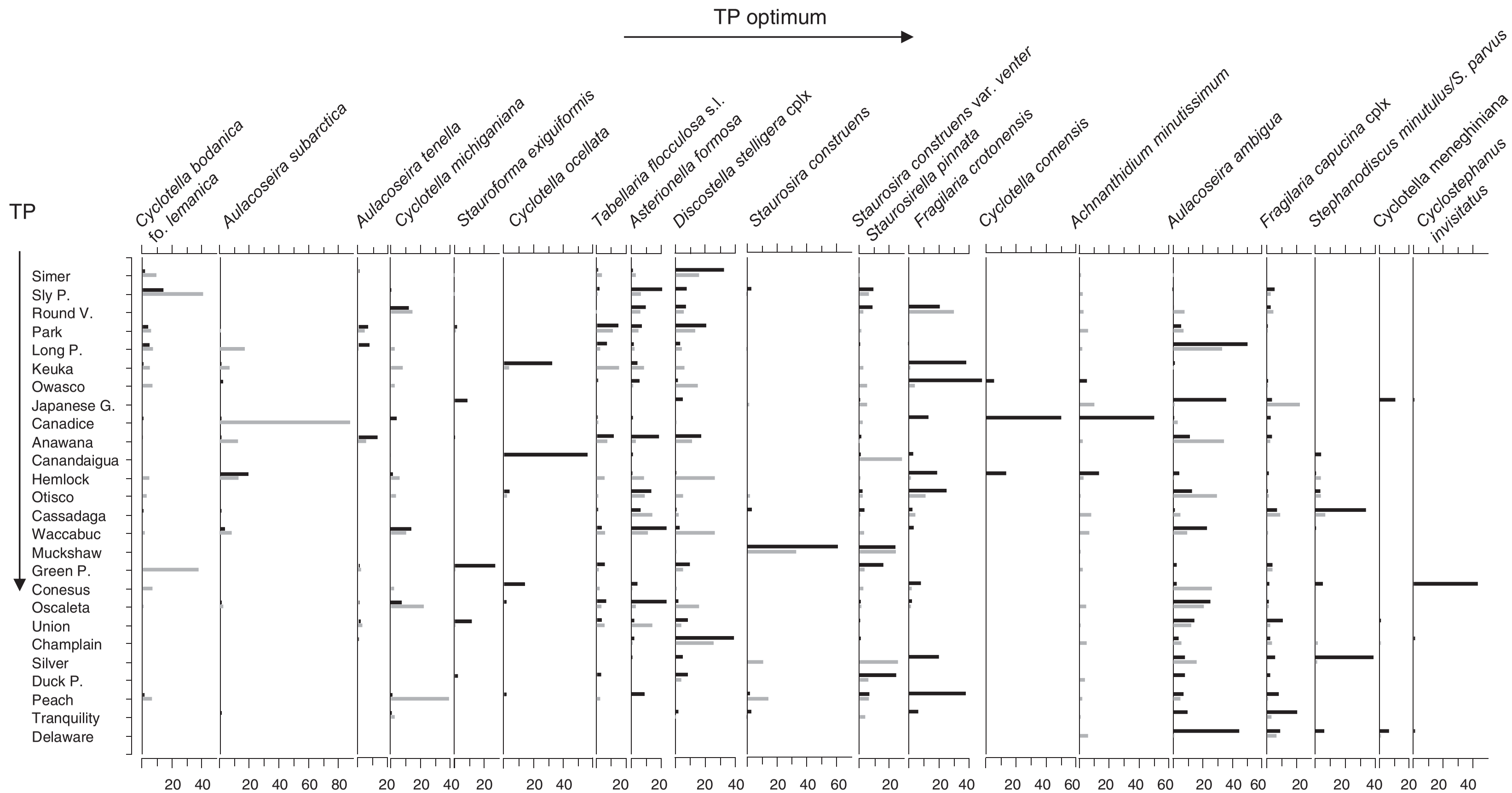

Figure 3.-Diatom species occurring >10% in at least one sample from top and bottom of 26 NJ and NY lakes. Black bar = top; gray bar = bottom. Diatom species are ordered horizontally by increasing TP optima (obtained from the NE-US transfer function); lakes are ordered vertically by increasing limnetic TP.

abundant species in bottom samples were: *Aulacoseira subarctica*, *Cyclotella michiganiana*, *A. granulata*, *A. ambigua*, *Tabellaria* spp. and a diverse benthic flora including *Navicula* spp., *Gomphonema* spp., and *Cymbella* spp. In top samples, the most abundant species were planktonic: *A. ambigua*, *F. crotonensis*, *Asterionella formosa*, and *Discostella* spp.; few benthic species were found. Changes in diatom relative abundances in top versus bottom samples varied (Fig. 3), but generally, taxa associated with high nutrient concentrations were more common in tops of cores and in lakes with currently higher TP.

Planktonic centric diatoms increased up to 70% in the top samples of 12 lakes. Decreases of planktonic diatoms between 5 and 40% took place in another 12 lakes. ANOVA and pair-wise t-tests showed that centric diatoms and *F. crotonensis* significantly increased, while benthic *Navicula* spp. significantly decreased in top core samples (Fig. 4), suggesting important shifts in limnological characteristics took place in study lakes.

Inferred TP

TP was inferred for top and bottom samples using both the NE-US and the NJ-NY transfer functions (Fig. 5). Each function has advantages and disadvantages and is most accurate within a different TP range. Compared with the NJ-NY model, the NE-US model is based on a much larger dataset, includes more taxa, and covers a wider geographic region. It contains many taxa found in the bottom core intervals not included in the NJ-NY calibration set (e.g., *C. bodanica* var. *lemanica*, *A. subarctica*) and also covers a much larger TP gradient, including many sites with TP values <10 μg/L. The NJ-NY model is based on only 33 samples, but the lakes are more similar to those to which it is applied. Another advantage is that the measured TP values in the NJ-NY model are based on averages of a few to several measurements per lake, while the NE-US TP was based on one (or in a few cases, 2) summer measurements made when lakes were stratified and the TP in the epilimnion was probably lower than at other times of the year.

The bootstrapped coefficient of determination of the NE-US transfer function is higher (0.69) than that of the NJ-NY (0.54); however, the root mean squared error of prediction (RMSEP) is higher in the NE-US (1.8 vs 1.5 μg/L), and when applied to reconstruct modern TP in the 28 study lakes (i.e., 26 top–bottom and 2 stratigraphic lakes) the NJ-NY model performs better (Fig. 5). Because the NJ-NY calibration set contains no samples from lakes with TP < 10 μg/L, and only one >100 μg/L, the TP range to which the NJ-NY model can be applied is limited. Differences in the 2 models are apparent on the plots of measured versus inferred TP for the full calibration datasets (Fig. 5a and 5b) and measured versus TP inferred for the tops of the 26 top–bottom cores (Fig. 5c and 5d). The NE-US model more accurately infers TP values less than about 15 μg/L, and the NJ-NY model more accurately infers concentrations from about 15 to 40 μg/L TP, the highest concentration in all but 2 top–bottom lakes. The NE-US transfer function inferred oligotrophic conditions (TP < 10 μg/L; Wetzel 2001) in core bottoms from 13 lakes (46% of study lakes). Six of these

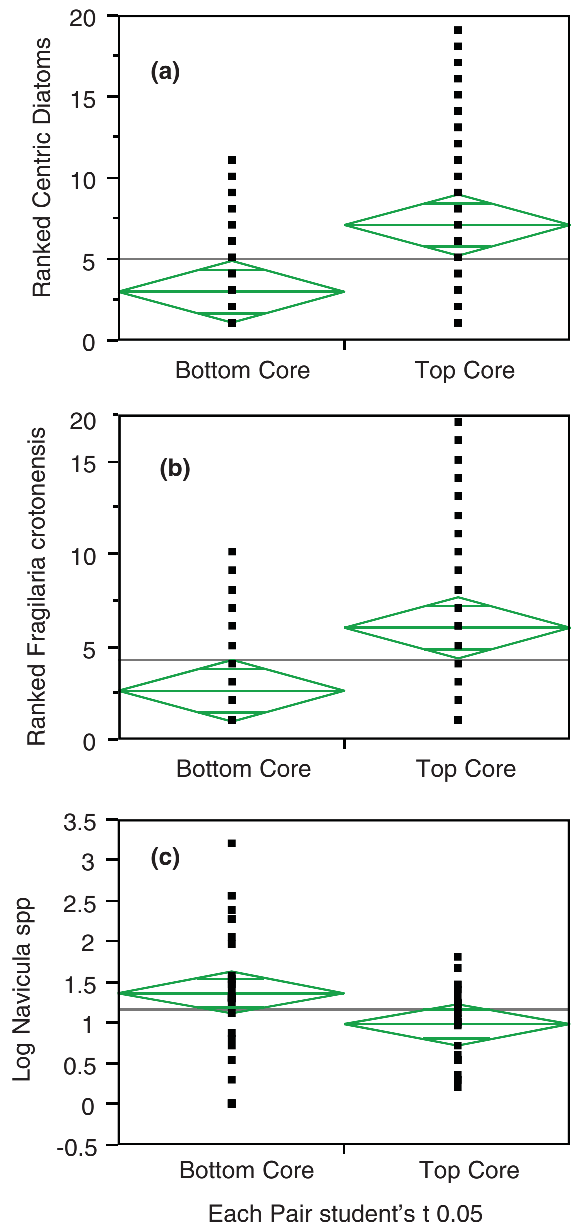

Figure 4.-Mean diamonds of a one-way ANOVA of select diatom species between top and bottom core samples. The vertical span of the diamond represents the 95% confidence interval. (Color figure available online).

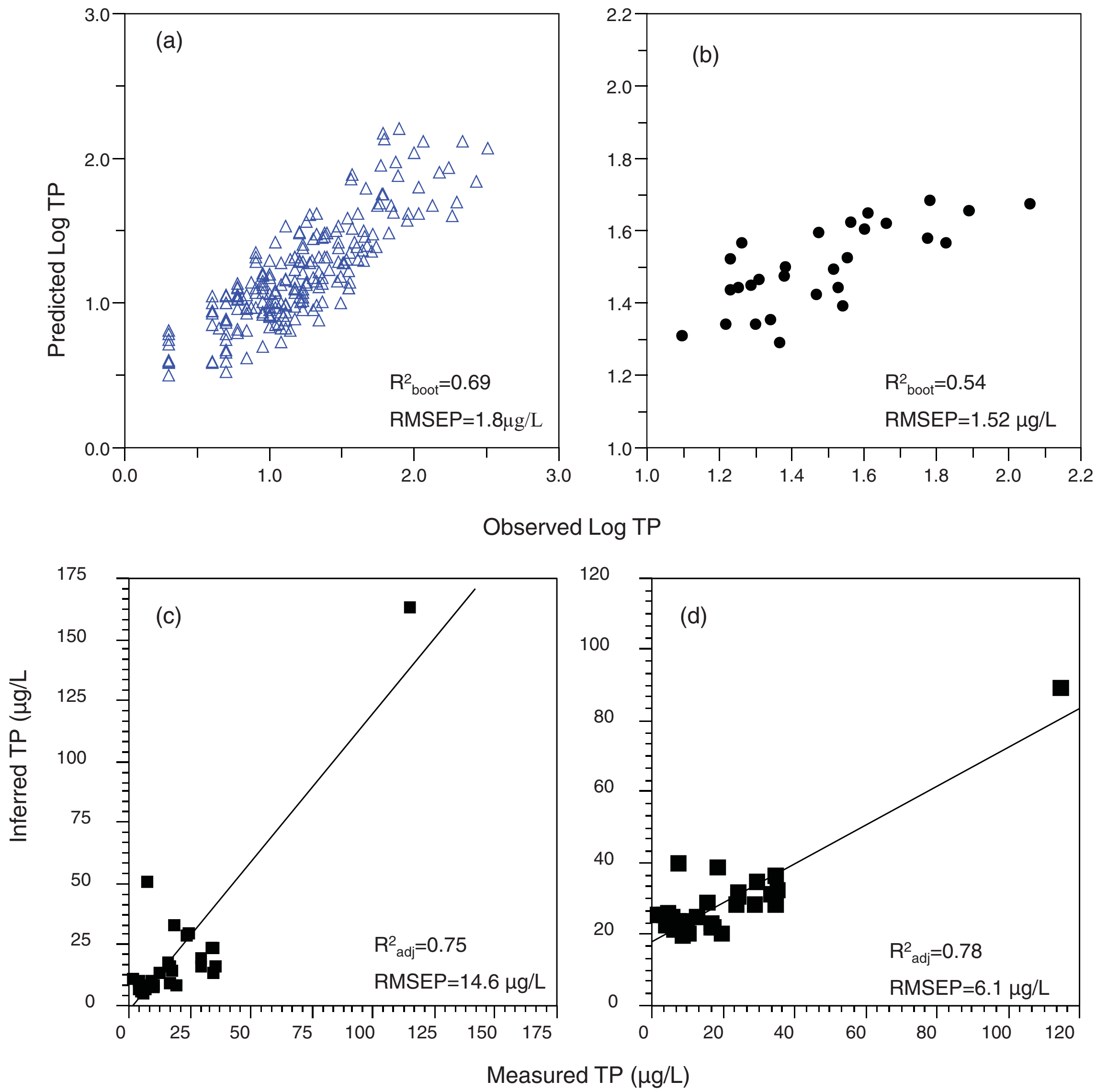

Figure 5.-Observed versus diatom-inferred TP based on WA-PLS component 2 models: (a) NE-US, R^2 apparent = 0.79 and (b) NJ-NY, R^2 apparent = 0.96. Bivariate fit of diatom-inferred versus measured TP in 28 NJ and NY sediment core tops based on: (c) NE-US; and (d) NJ- NY transfer functions. The 28 core tops comprise tops from the 26 top–bottom cores and from Greenwood and Cossayuna lakes (Color figure available online).

lakes now experience mesotrophic or eutrophic conditions based on the recent measured values (Table 1).

Top–bottom differences were calculated using inferred values for the NE-US and NJ-NY models separately and for the combination of values from the 2 models that were most appropriate for the TP range being inferred. When selecting model values to use for combinations, the NE-US model was used for all bottom samples because TP concentrations associated with most bottom samples are <15 μg/L, based on agreement between measured and inferred values for lakes with no or little change between top and bottom values. Also, diatom taxa in several bottom samples were not well represented in the NJ-NY calibration set. About half of the bottom samples had poor analogs in the NJ-NY calibration set in comparison with the NE-US model (Table 2). For top samples, the NE-US inferred value was used if the measured TP was ≤ 15 μg/L and the NJ-NY model value was used if the measured TP was >15 μg/L.

The average top–bottom change calculated using the 3 approaches described above was relatively small, ranging from 2 to 7 μg/L. Ranges varied from a decrease of 13 to an increase of 119 μg/L, although most increases were <20 μg/L.

Table 2.-Diatom-inferred TP in top and bottom samples from 28 study lakes. * = Stratigraphic lakes. Top–Bot. Comb. = Difference between most appropriate combination of NE-US and NJ-NY inferred TP values. Bold names = bottom samples with better modern analog in the NE-US dataset [Difference of Minimum Dissimilarity Coefficient (NJ-NY) − (NE-US) ≥ 1].

	NE-US Inferred TP (μg/L)			NJ-NY Inferred TP (μg/L)				
Lake	**Top**	**Bottom**	**Top–Bot**	**Top**	**Bottom**	**Top–Bot**	**Top–Bot. Comb.**	**Core length (cm)**
Anawana	9.7	11.5	1.9	23.0	21.9	1.1	−2.0	32.0
Canadice	11.0	5.2	5.8	19.0	18.8	0.2	6.0	36.0
Canandaigua	8.1	9.8	−1.7	22.8	36.1	−13.3	−2.0	44.5
Cassadaga	18.9	24.9	−6.0	28.6	33.1	−4.5	4.0	40.0
Champlain	30.5	16.4	14.1	31.1	32.2	−1.1	15.0	56.0
Conesus	34.0	15.4	18.6	38.3	33.7	4.6	23.0	58.0
Cossayuna*	16.7	11.4	5.3	31.7	18.6	13.1	21.0	42.0
Delaware	164.2	45.6	118.7	88.9	46.6	42.4	43.0	41.0
Duck Pond	24.3	36.5	−12.2	30.5	30.1	0.3	−7.0	40.0
Green Pond	15.1	8.0	7.1	21.7	22.3	−0.6	7.0	36.0
Greenwood*	17.0	6.9	10.1	28.1	25.9	2.1	21.0	35.0
Hemlock	9.6	7.1	2.5	19.9	22.5	−2.6	3.0	31.0
Jap Gard	51.8	32.7	19.1	39.3	33.8	5.6	6.0	34.0
Keuka	7.9	5.0	2.8	20.9	21.9	−1.1	3.0	42.0
Long Pond	6.1	6.0	0.1	24.4	22.1	2.3	1.0	47.0
Muckshaw	17.1	18.4	−1.3	22.4	25.0	−2.6	−1.0	43.0
Oscaleta	9.2	6.9	2.3	19.9	18.5	1.4	−1.0	30.0
Otisco	14.2	9.2	5.0	24.5	25.9	−1.3	13.0	50.0
Owasco	9.3	17.5	−8.3	23.4	29.7	−6.3	5.0	28.0
Park	6.9	7.0	−0.1	25.3	24.3	0.9	−8.0	30.0
Peach	14.7	7.3	7.3	27.8	20.8	6.9	0.0	51.0
Round Valley	11.3	15.2	−3.9	22.0	26.7	−4.7	21.0	28.0
Silver	20.0	23.2	−3.1	34.1	31.0	3.1	7.0	52.0
Simer	12.1	10.6	1.5	25.1	24.6	0.4	11.0	35.0
Sly Pond	7.6	7.5	0.0	24.7	27.6	−2.9	0.0	41.0
Tranquility	24.3	24.6	−0.3	36.0	34.3	1.7	11.0	21.5
Union	29.3	35.8	−6.5	27.7	23.9	3.8	−8.0	35.0
Waccabuc	10.6	5.5	5.1	21.4	19.5	1.9	8.0	32.0

Using the RMSEPs as a measure of uncertainty (± 1.8 μg/L), top–bottom differences of <5 μg/L can be roughly considered within model error. Of the 26 top–bottom lakes, 15 had top–bottom differences of 5 μg/L or less using the NE-US model, 22 using the NJ-NY model, and 11 using the combination of the 2 models. Several lakes had differences of >5 but <10 μg/L. Only 5, 3, and 9 lakes had top–bottom differences >10 μg/L using the US-NE, NJ-NY, and combination of models, respectively. Most of these lakes have current TP concentrations >20–30 μg/L.

The NE-US model top–bottom differences are generally greater than those calculated using the NJ-NY model. There is a good fit between reconstructions by both models if Lake Delaware is included (Fig. 6a); however, the fit between the 2 top–bottom reconstructions drops dramatically if Lake Delaware is excluded (Fig. 6b).

To explore how lakes of various modern TP have changed since core bottoms were deposited, the TP changes inferred by the NJ-NY and the NE-US models were plotted against the measured modern lake TP (Fig. 7 and 8). The top–bottom difference increases with increasing measured TP, but the relationship is significant only for the NY-NJ model (Fig. 7).

Historical changes in Greenwood and Cossayuna lakes

Diatom assemblages displayed distinct changes in the sedimentary records of Greenwood and Cossayuna Lakes (Figs 9 and 10). Three zones were identified in each core. Diatom-inferred TP values calculated using the NE-US and NJ-NY models (Fig. 11) increase towards the top of the core in both lakes. The measured TP agrees most closely with the

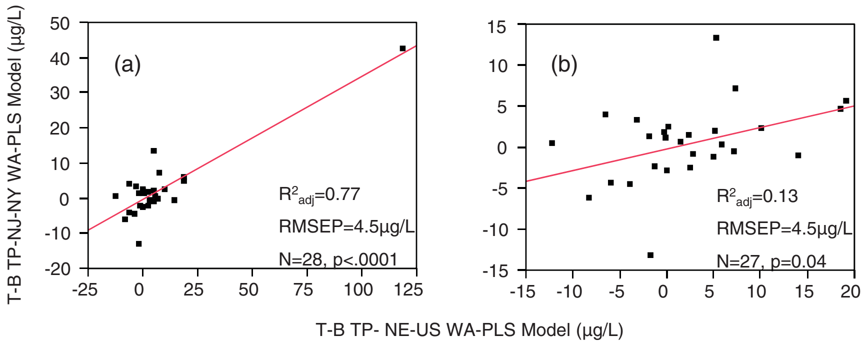

Figure 6.-Bivariate fit between top–bottom (T–B) inferred TP reconstructions using (a) NE-US and NJ-NY inference models and (b) same analysis with Lake Delaware excluded (Color figure available online).

NJ-NY model inferred TP, suggesting values from that model are more accurate, at least near the top of the cores; however, values near the bottom of the cores, especially the Cossayuna core, approach 15 μg/L, a part of the TP range where either model may be most accurate. Bottom values are <15 μg/L for the NE-US model, suggesting they may be the most accurate. MAT also revealed better analogs of core bottom samples in the NE-US model.

Discussion

Historical changes in top–bottom lakes

Nutrient-related changes

Overall, shifts in diatom species in the top–bottom study lakes suggest changes from lower to higher nutrient levels. For example, the eutrophic *Cyclostephanus* and *Stephanodiscus* spp. were abundant in Conesus, Silver, and Cassadaga lake's top samples while they were rare in bottom samples. *F. crotonensis*, a well-known indicator of lake eutrophication (Dixit et al. 1999) increases significantly (20–50%) in top samples in Owasco, Keuka, Peach, Otisco, Silver, and Hemlock lakes (Fig. 3). The hypereutrophic planktonic species *Cyclotella meneghiniana* increases >5% in Japanese Garden and Delaware lakes. Oligotrophic species strongly decrease or disappear from most study lakes and are replaced by meso- and eutrophic species such as *F. crotonensis*, *A. ambigua*, or *Stephanodiscus* spp. (Fig. 3). Some mesoeutrophic species (e.g., *A. ambigua*, *F. crotonensis*, *T. flocculosa*) were found in bottom samples of Anawana, Otisco, Round Valley, and Keuka lakes, suggesting that these lakes may have had moderate nutrient concentrations, or that their sediment cores were not long enough to reach reference conditions. Shero et al. (1978) studied the relationships between diatom taxa groups and productivity in 43 New York lakes and found that the best predictor of productivity was percent *Cyclotella* spp., the percent decreasing with increased productivity. They did not indicate which species they found, so comparisons with this study are difficult. However, *C. bodanica* and *C. michiganiana*, taxa indicating low nutrient levels, decreased in relative abundance toward the surface of our study lakes, consistent with increasing nutrients, while *C. comensis* and *C. meneghiniana*, indicators of higher nutrient concentrations, appeared only in surface sediments. Overall, top–bottom changes in diatom species indicate that nutrient concentrations have increased over a large geographic scale, including in lakes not currently on EPA's list of nutrient-impaired lakes (Table 1). These increases have been found in lakes with watershed development. Lakes with little disturbance in their watersheds, such as those in the Adirondack Park (Long, Sly, and Little Simon ponds), had small or no changes in inferred TP.

A significant feature of the top–bottom changes is the increase in relative abundance of planktonic species, and corresponding sharp decrease in *Navicula* spp., in top samples (Fig. 3). This suggests that important changes took place in lake environmental conditions. In addition to nutrients, these shifts could have been caused by changes in climate, pH (though probably not substantially), sediment input, abundance of aquatic macrophytes, lake depth, and UVB radiation (Enache et al. 2011). Another important factor could be changes in zooplankton populations resulting from introductions or changes in fish predation (e.g., Brown and Balk 2008).

Climate-related changes

Changes related to climate conditions are also indicated by increases of species previously shown to increase in response to warmer temperature, longer ice-free season, and increased length of lake stratification. For example, an increase in the proportion of small *Cyclotella* and *Discostella* species, usually abundant during summer stratification, may indicate a response to recent climate warming (Smol et al.

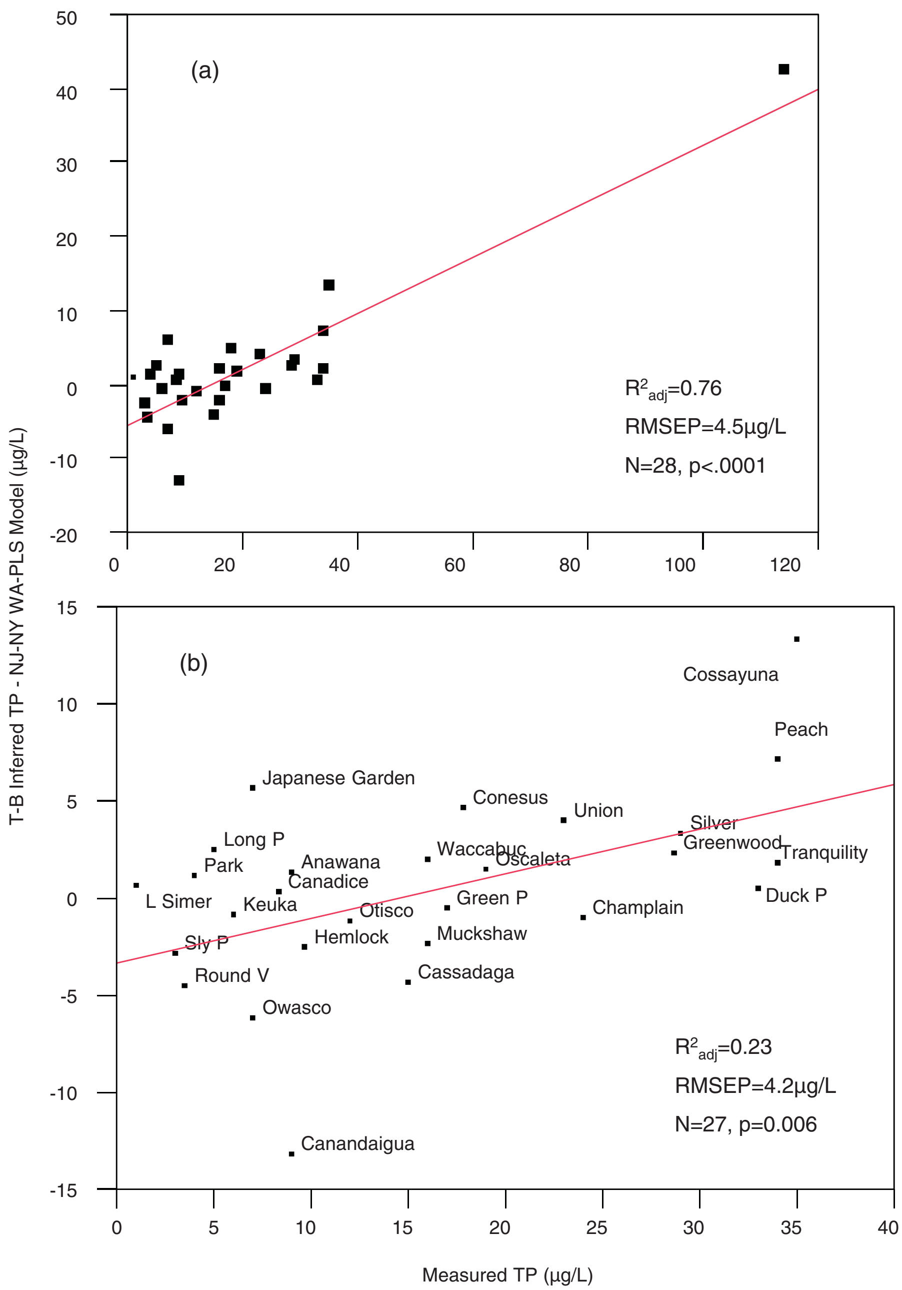

Figure 7.-Bivariate analyses of top–bottom (T–B) TP reconstructions inferred using (a) the NJ-NY inference model versus modern measured TP and (b) same analysis with Lake Delaware excluded. (Color figure available online).

2005, Rühland et al. 2008), such as recorded in Canandaigua, Keuka, and Conesus lakes. Increasing temperatures and prolonged droughts can induce lengthening of the ice-free season and period of lake stratification (Catalan et al. 2002, Sorvari et al. 2002), which can lead to lower nutrient concentrations in the epilimnion favoring small *Discostella* or *Cyclotella* species (Smol et al. 2005). Thus, climate change may be causing reduction of nutrient concentrations over time.

Inferred TP

The changes inferred by the NJ-NY model were higher in high nutrient lakes (Fig. 7), suggesting that these lakes underwent significant trophic changes and that anthropogenic impact may be the principal source of trophic change. The highest TP increases were recorded in Delaware, Peach and Cossayuna lakes, all of which are currently eutrophic. Lakes with currently low TP and little watershed development,

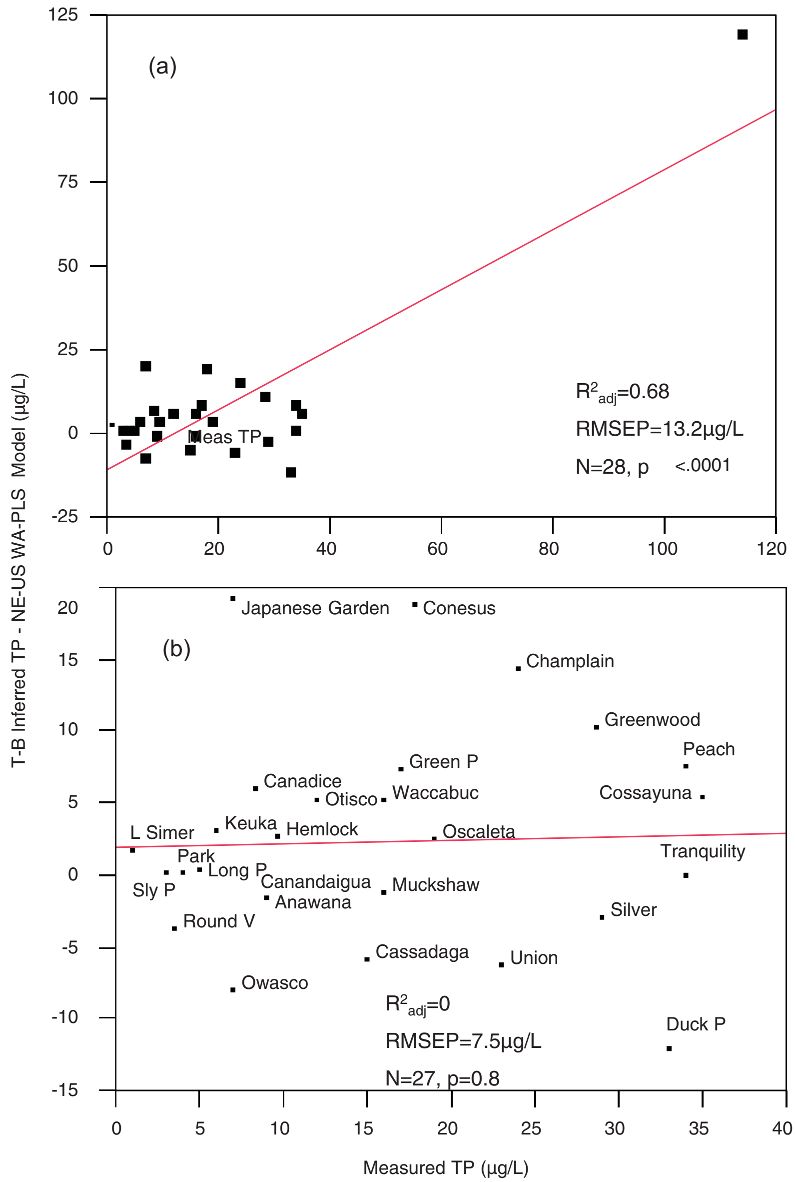

Figure 8.-Bivariate analyses of top–bottom (T–B) TP reconstructions inferred using (a) the NE-US inference model versus modern measured TP and (b) same analysis with Lake Delaware excluded. (Color figure available online).

such as the lakes in the Adirondack Park (Little Simon, Long, and Sly), showed little TP change.

Several types of human disturbance have influenced TP in the study lakes in the past. This, and variability among the types of lakes, has resulted in patterns of changing TP concentrations and diatom assemblages unique to each lake that must be studied individually. Changes in many lakes are consistent with known histories of the lakes (e.g., Finger Lakes; Callinan 2001), although further investigation could be undertaken that considers the uncertainty of dates and inferred TP values.

The core bottoms were not dated in this study; however, sediment accumulation rates (SAR) based on cores collected in the 1990s from some New York lakes (Callinan 2001) can be used to estimate sediment age. Based on these SARs, the bottom sample from Keuka is from the late 1800s, Owasco from ~1920, Canadice from ~1820, Canandaigua from ~1830, Otisco from ~1940, and Conesus from ~1860. If these estimates are accurate, the cores of these lakes did not likely reach predisturbance conditions. European settlement took place as early as the mid-1600s to early-1700s in the eastern United States (Hull 1975, Eaton and Kardos 1978, Forest

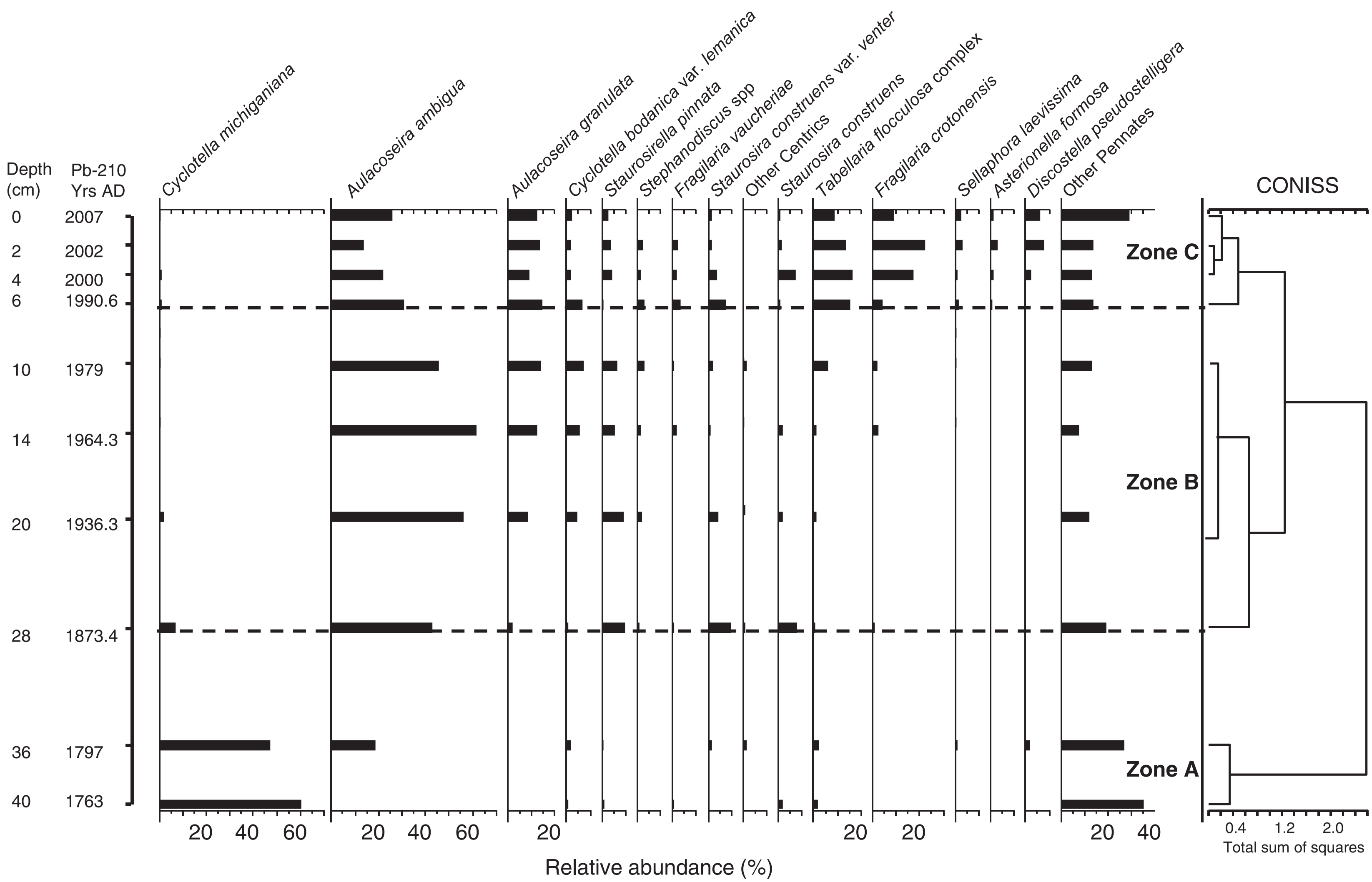

Figure 9.-Relative abundances of diatom species in core stratigraphy of Cossayuna Lake. Diatom taxa are arranged according to the first-axis species scores of a principal component analysis (PCA). A constrained cluster analysis (CONISS), based on a chord distance as the measure of dissimilarity (Grimm 1987), is used to indicate the distinct differences in assemblages (zones) in the 39 cm sediment record.

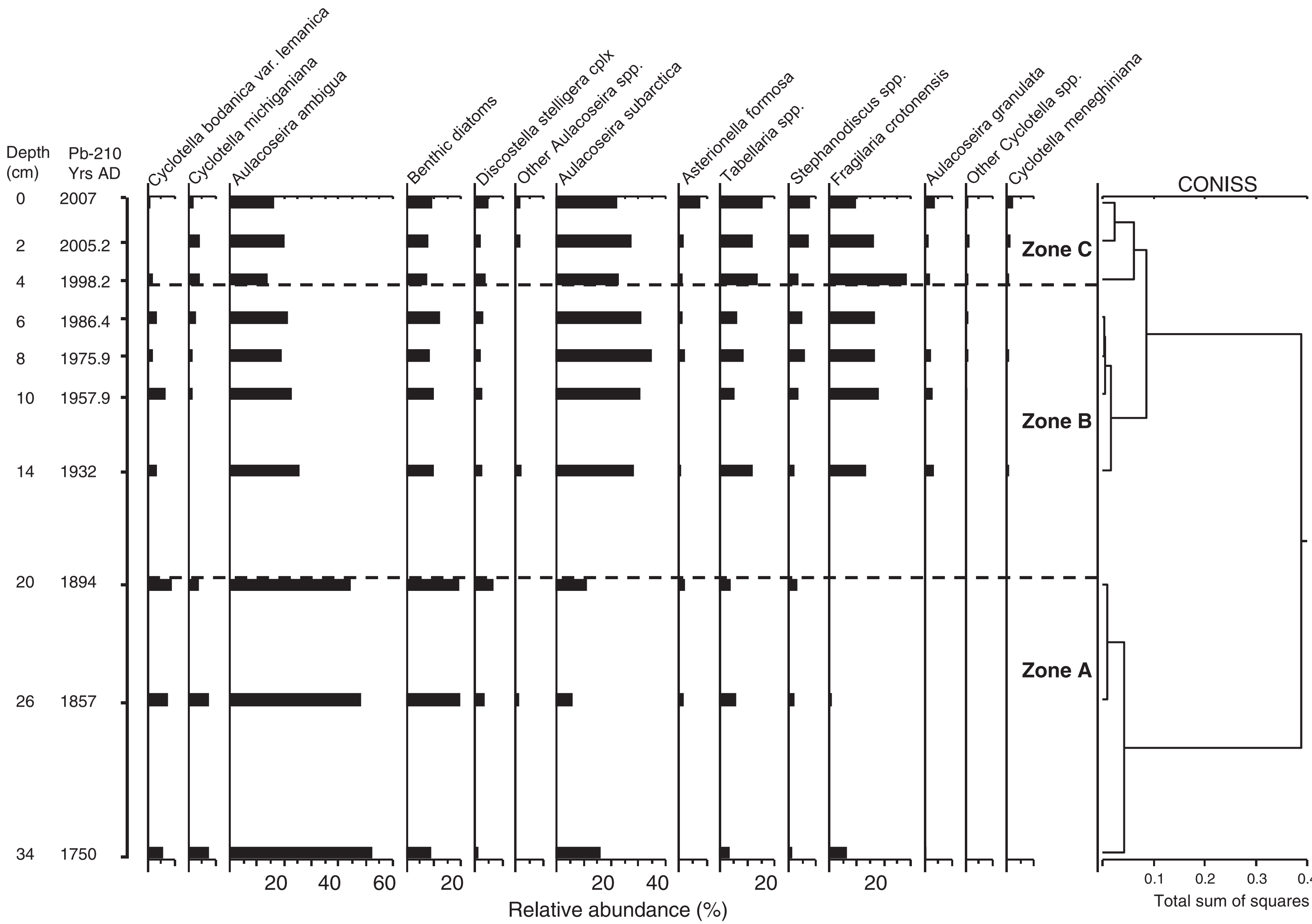

Figure 10.-Relative abundances of diatom species in core stratigraphy of Greenwood Lake. Diatom taxa are arranged according to the first-axis species scores of a principal component analysis (PCA). A constrained cluster analysis (CONISS), based on a chord distance as the measure of dissimilarity (Grimm 1987), is used to indicate the distinct differences in assemblages (zones) in the 35 cm sediment record.

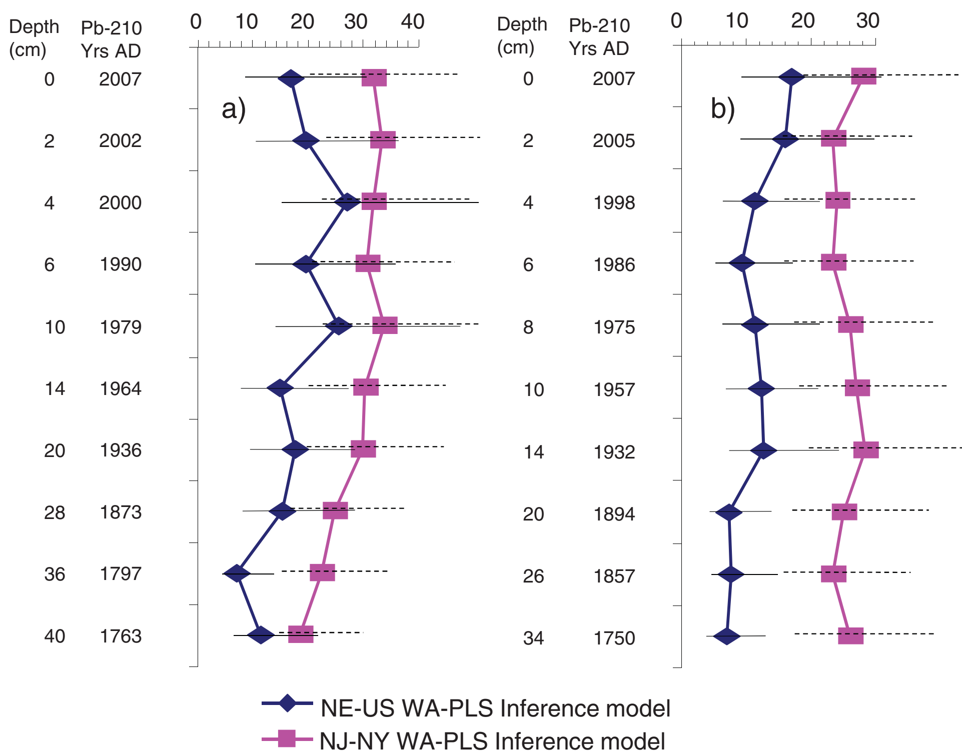

Figure 11.-Diatom-based reconstruction of TP for (a) Cossayuna and (b) Greenwood Lake cores based on the NE-US WA-PLS and NJ-NY transfer functions. Error bars represent ±RMSEP – solid line for the NE-US model, dashed line for the NJ-NY model (Color figure available online).

et al. 1978), and intense agricultural activity may already have been impacting study lakes in a time period earlier than our bottom sediment cores. Given that human disturbance tends to increase nutrient concentrations, bottom interval TP values probably overestimate true predisturbance reference conditions, and top–bottom differences are also probably underestimated for several lakes.

Although many changes in lake TP and primary productivity may reflect anthropogenic effects, changing climate can also affect water level, transparency, temperature and stratification, or extent of surrounding wetlands (Enache et al. 2011). All of these could have influenced TP concentrations in the study lakes. For example, inferred TP decreased in the core tops of some lakes (Table 2); this may be related to lake recovery as a result of recent restoration programs or recent climate change. Eutrophic (Duck Pond, Tranquility, and Silver), oligotrophic (Sly Pond, Round Valley, and Owasco) and many mesotrophic lakes (e.g., Cassadaga and Muckshaw) recorded TP decreases, albeit small, in top samples. Warmer climate may have caused earlier onset and prolonged summer stratification and low epilimnetic TP. A similar phenomenon recorded in remote temperate (Enache et al. 2011) and arctic–alpine lakes (Smol et al. 2005) is considered to be related to recent climate change.

Historical changes in Greenwood and Cossayuna lakes

Cossayuna Lake

Diatom-inferred TP near the bottom of the core (Zone A) shows that oligotrophic (inferred by the NE-US model) or mesotrophic (inferred by the NJ-NY model) conditions were present at the time the core bottom section was deposited (Fig. 11). Zone A represents the mid-to-late 1700s (36 to 41 cm), when the first settlers arrived (Fig. 9). Diatom TP inferences within this zone suggest that at that time there was little impact from the first settlers on lake condition.

Diatom TP reconstructions indicate that about 100 years later, TP levels increase to ~30 μg/L, more than 1.5 to 3 times bottom TP inferred by the NJ-NY and NE-US models, respectively (Fig. 11). Phosphorus increases can be linked to increased human activities in the region. Train service was introduced to Greenwich in 1869 and by 1895 contributed significantly to increased population, farming, and industrial activities.

A slight and short-lived decrease in TP levels was inferred in the 1980s, indicated by the 25165926425166028 825166131225166233 6 decline of *A. ambigua*, possibly in response to management practices implemented to reduce nutrient enrichment (e.g., a water pollution treatment plant in 1970 and the ban of phosphate in detergents). The present-day measured TP of 35 μg/L is similar to NJ-NY model inferred values. In contrast, the NE-US model indicates 16.7 μg/L of TP, strongly underestimating the present day TP (Fig. 11).

Greenwood Lake

TP inferred by both inference models shows similar trends, but the NE-US values are consistently lower (Fig. 11). The high abundance of *A. ambigua*, a mesotrophic to eutrophic species within Zone A, is more consistent with the NJ-NY inferred values. This suggests either that high nutrient reference conditions are characteristic in Greenwood Lake or that the lake had already been impacted by logging and agricultural activities. European settlement started in 1707. A major foundry was established in 1754, and sawmills, forges, and gristmills were constructed throughout the area. In 1765, a small dam was built to provide some of these operations with water power (Hull 1975). The CRS dating of the core bottom section (26–35 cm) indicates mid-1700s to ~mid-1800s (Fig. 10), suggesting that the hypothesis of a system already impacted by anthropogenic activities at the core bottom is plausible.

An important shift in diatom species composition from *A. ambigua* to *A. subarctica* takes place above 20 cm depth of the core (1894) and is concurrent with a change in sediment color from light brown in the bottom to dark brown. *A. subarctica* has lower temperature preference than *A. ambigua*, and this shift is consistent with higher water levels and increased stratification taking place after construction of a dam in 1836 that elevated the lake by 3.7 m. Further, appearance of the eutrophic *F. crotonensis* at 14 cm core depth suggests increasing nutrient levels after 1930. Inferred TP (NJ-NY model) at that time was ~28 μg/L, which is close to the present-day TP level (Fig. 11). Nutrient levels are lower after 1975, possibly in response to the phosphate detergent ban, but a further TP increase was inferred for the top of the core.

Conclusions

Diatom-inferred TP concentrations provided estimates of New Jersey and New York lake environmental changes in the absence of instrumental records. Inferred TP values indicate that 60% of the lakes have increased in TP up to 43 μg/L, which is probably an underestimate for many lakes where top–bottom cores were not long enough to reach predisturbance conditions. The greatest changes were found in lakes with higher modern trophic levels. Inferred TP increases can be used to estimate how much TP would need to be reduced to return lakes to historical conditions, and what TP levels might be reasonable targets for lake restoration practices.

Detailed stratigraphic analyses of Greenwood and Cossayuna cores revealed increases in TP in the range of 10–20 μg/L, consistent with timing of European post-settlement and modern anthropogenic activities. Despite the restoration programs implemented over the last few decades, both Cossayuna and Greenwood lakes have high present day TP values (both measured and diatom-inferred), after a short period of lower diatom-inferred TP in late 20th century.

Shifts in top–bottom diatom assemblages indicate that in addition to human-induced lake eutrophication, climate change is also probably impacting lake water characteristics in New York and New Jerseu. This suggests the need to understand the impact of climate on lakes already subject to land-use impairment, another aspect that should be taken into account in lake management practices.

The application of a large-scale (NE-US) and a region-specific (NJ-NY) transfer function to reconstruct TP in top–bottom and stratigraphic lakes from New Jersey and New York revealed more information on the ability of such models to provide reliable past quantitative reconstructions. Although it comprises only 33 lakes, the NJ-NY transfer function provided better estimates for the modern lake TP of 28 study sites from New Jersey and New York. Because of its small size, however, the regional model did not comprise sufficient lakes at the extremes of the TP gradient, especially at its lower end. This issue particularly affects New Jersey because most lakes are highly impacted by cultural eutrophication, and pristine lakes are very rare. Therefore, because of the limited ability of the NJ-NY model to reconstruct low TP concentrations in bottom samples, we recommend a larger-scale inference model that can provide better modern analogs for fossil samples and more reliable historical reconstructions.

Site selection of calibration sets should be refined to provide more accurate inference models for the lakes being studied. Interpretations of inferred values should account for limnological characteristics of the lakes, watershed histories, human stressors including climate change, and introduced species. Also, the bottom samples of top–bottom sediment cores should be dated by pollen or ^{210}Pb analysis to confirm that the bottom interval reaches presettlement conditions.

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References

Appleby PG, Oldfield F. 1978. The calculation of lead-210 dates assuming a constant rate of supply of unsupported 210-Pb to the sediment. Catena. 5:1–8.

Battarbee RW. 1999. The importance of paleolimnology to lake restoration. Hydrobiologia. 395/396:149–159.

Battarbee RW, Anderson NJ, Jeppesen E, Leavitt PR. 2005. Combining paleolimnological and limnological approaches in assessing lake ecosystem response to nutrient reduction. Freshwater Biol. 50:1772–1780.

Bennion H, Appleby PG, Phillips GL. 2001. Reconstructing nutrient histories in the Norfolk Broads: implications for the application of diatom-phosphorus transfer functions to shallow lake management. J Paleolimnol. 26:181–204.

Bennion H, Juggins S, Anderson NJ. 1996. Predicting epilimnetic phosphorus concentrations using an improved diatom-based transfer function and its application to lake eutrophication management. Environ Sci Technol. 30:2004–2007.

Binford MW. 1990. Calculation and uncertainty analysis of ^{210}Pb dates for PIRLA project lake sediment cores. J Paleolimnol. 3:253–267.

Birks HJB. 1998. Numerical tools in palaeolimnology – Progress, potentialities, and problems. J Paleolimnol. 20:307–332.

Bradshaw EG, Anderson NJ. 2001. Validation of a diatom-phosphorus calibration set for Sweden. Freshwater Biol. 46:1035–1048.

Brown ME, Balk MA. 2008. The potential link between lake productivity and the invasive zooplankter *Cercopagis pengoi* in Owasco Lake (New York, USA). Aquat Invasions. 3: 28–34.

Cadmus Group. 2008. Total maximum daily load (TMDL) for phosphorus in Cossayuna Lake, Washington County, NY. Prepared for the US EPA and N.Y. Department of Environmental Conservation; [cited Oct 30, 2011]. Available from (http://www.dec.ny.gov/docs/water_pdf/tmdlcossayuna08.pdf)

Callinan CW. 2001. *Water quality study of the Finger Lakes*. New York State Department of Environmental Conservation.

Catalan J, Pla S, Rieradevall M, Felip M, Ventura M, Buchaca T, Camarero L, Brancelj A, Appleby PG, Lami A, et al. 2002. Lake Redó ecosystem response to an increasing warming the Pyrenees during the twentieth century. J Paleolimnol. 28:129–145.

Charles DF, Binford MW, Furlong ET, Hites RA, Mitchell MJ, Norton SA, Oldfield F, Paterson MJ, Smol JP, Uutala AJ, et al. 1990. Paleoecological investigation of recent lake acidification in the Adirondack Mountains, NY. J Paleolimnol. 3:195–241.

Charles DF, Knowles C, Davis RS. 2002. Protocols for the analysis of algal samples collected as part of the US Geological Survey National Water-Quality Assessment Program. The Academy of Natural Sciences, Philadelphia. PCER Report No 02–06.

Cumming BF, Smol JP, Kingston JC, Charles DF, Birks HJB, Camburn KE, Dixit SS, Uutala AJ, Selle AR. 1992. How much acidification has occurred in Adirondack region (New York, USA) lakes since pre-industrial times? Can J Fish Aquat Sci. 49:128–141.

Dixit SS, Smol JP. 1994. Diatoms and indicators in the environmental monitoring and assessment program -surface waters (EMAP-SW). Environ Monitor Assess. 31:275–306.

Dixit SS, Smol JP, Charles DF, Hughes RM, Paulsen SG, Collins GB. 1999. Assessing lake water quality changes in the lakes of the northeastern United States using sediment diatoms. Can J Fish Aquat Sci. 56:131–152.

Eaton SW, Kardos LP. 1978. The limnology of Canandaigua Lake. In: JA Bloomfield, editor. *Lakes of NY state. Vol 1. Ecology of the Finger Lakes*. New York (NY): Academic Press. p. 226–311.

Enache MD, Paterson AM, Cumming BF. 2011. Changes in diatom assemblages since pre-industrial times in 40 reference lakes from the Experimental Lakes Area (northwestern Ontario, Canada). J Paleolimnol. 46:1–15.

Forest HS, Wade JQ, Maxwell TF. 1978. The limnology of Conesus Lake. In: JA Bloomfield, editor. *Lakes of NY state. Vol 1. Ecology of the Finger Lakes*. New York (NY): Academic Press. p. 122–224.

Glew JR. 1988. A portable extruding device for close interval sectioning of unconsolidated core samples. J Paleolimnol. 1: 235–239.

Glew JR. 1989. A new trigger mechanism for sediment samplers. J Paleolimnol. 2:241–243.

Glew JR. 1991. Miniature gravity corer for recovering short sediment cores. J Paleolimnol. 5:285–287.

[GLC-PRC] Greenwood Lake Commission and the Passaic River Coalition. 2011. Restoration and protection of the natural resources of the Greenwood Lake watershed in New Jersey. Project Report #1. Available from: http://www.gwlc.org/images/ZMProject%201-1.pdf

Grimm EC. 1987. CONISS: A FORTRAN 77 program for stratigraphically constrained cluster analysis by the method of incremental sum of squares. Computers and Biosciences. 13:13–25.

Hall RI, Smol JP. 1999. Diatoms as indicators of lake eutrophication. In: EF Stoermer, JP Smol, editors. *The diatoms: applications for the environmental and earth sciences*. Cambridge (UK): Cambridge University Press. p. 128–168.

Hull RW. 1975. *People of the Valleys Revisited: History of Warwick, New York. 1700-2005*. Unionville (NY): Royal Fireworks Press.

Juggins S. 2003. *C^2 User guide. Software for ecological and paleoecological data analysis and visualization*. Newcastle upon Tyne (UK): University of Newcastle.

LA Group, PC. 2001. Lake Management Plan. An action plan for the long term management of nuisance aquatic vegetation in

Cossayuna Lake. A Report prepared for the Town of Argyle, NY.

[NJDEP] New Jersey Department of Environmental Protection. 1997. Greenwood Lake. Water Level Management Plan. 1997–2017. NJDEP, Division of Parks and Forestry.

Ponader KC, Charles DF, Belton TJ. 2007. Diatom-based TP and TN inference models and indices for monitoring nutrient enrichment of New Jersey streams. Ecol Indicators. 7:79–93.

Reavie ED, Smol JP, Sharpe ID, Westenhofer LA, Roberts A-M. 2000. Paleolimnological analyses of cultural eutrophication patterns in British Columbia lakes. Can J Bot. 78:873–888.

Rühland K, Paterson AM, Smol JP. 2008. Hemispheric-scale patterns of climate-related increases in planktonic diatoms from North American and European lakes. Glob Change Biol. 14:1–15.

Sebetich MJ, Messaros R. 1993. Paleolimnology of Green Pond, a 200-ha glacial lake in the Highlands Province of New Jersey, USA. Verh Internat Verein Limnol. 25:1065–1071.

Shero BR, Parker M, Stewart KM. 1978. The diatoms, productivity and morphometry of 43 lakes in New York State, U.S.A. Int Rev ges Hydrobiol. 63:365–387.

Smol JP. 1992. Paleolimnology: an important tool for effective ecosystem management. J Aquat Ecosyst Health. 1:49–58.

Smol JP, Stoermer EF, editors. 2010. *The diatoms: Applications for the environmental and earth sciences*. Cambridge (UK): Cambridge University Press.

Smol JP, Wolfe AP, Birks HJB, Douglas MSV, Jones VJ, Korhola A, Pienitz R, Rühland K, Sorvari S, Antoniades D, et al. 2005. Climate-driven regime shifts in the biological communities of arctic lakes. PNAS. 102:4397–4402.

Sorvari S, Korhola A, Thompson R. 2002. Lake diatom response to recent Arctic warming in Finnish Lapland. Glob. Change Biol. 8:171–181.

ter Braak CJF, Juggins S. 1993. Weighted averaging partial least squares regression (WA-PLS): an improved method for reconstructing environmental variables from species assemblages. Hydrobiologia. 269–270:485–502.

Wetzel RG. 2001. Limnology. Lake and river ecosystems. 3rd ed. San Diego (CA): Academic Press.