PALEOLIMNOLOGICAL INVESTIGATION OF POST-SETTLEMENT EUTROPHICATION IN BRITISH COLUMBIA, CANADA

by

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ABSTRACT

Anthropogenic eutrophication is suspected of being a problem in many British Columbia (BC) lakes. However, long-term limnological data are rare or unavailable for most lake systems, and so the natural pre-disturbance conditions of lakes, as well as the timing and magnitude of human-related changes, are unknown. This thesis explores the use of diatoms in paleolimnological analyses of BC lakes to elucidate long-term trends in eutrophication.

A transfer function, derived from the relationship between the limnological characteristics and surface sediment diatom assemblages of 59 BC lakes, was constructed to infer lakewater total phosphorus concentrations ([TP]) from diatom species composition. Eighteen lakes were added to a published calibration set of 46 BC lakes to expand the original range of [TP]. Canonical correspondence analysis was used to explore the relationship between diatom assemblages and environmental variables. Specific conductivity and [TP] each explained significant ($P \leq 0.05$) directions of variance in the distribution of the diatoms. The updated calibration model covers a range of species optima from 6 to 41.9 μg/L TP, and includes a total of 150 diatom taxa.

Six presently eutrophic BC lakes were chosen for detailed paleolimnological analyses. Diatom assemblages preserved in dated sediment cores indicated that all of the lakes had fairly high phosphorus levels (> 13 μg/L) prior to European settlement of inland BC (*circa* A.D. 1850). Three of the lakes showed significant eutrophication since that time, whereas the others were only mildly affected by anthropogenic activities.

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Humans have accelerated the eutrophication process in many lake regions (Vollenweider, 1980) as the export of limiting nutrients, such as phosphorus (Prairie and Kalff, 1986) and nitrogen (Duarte and Kalff, 1989), to lakes has increased. Direct nutrient inputs include sewage, domestic detergents and industrial waste. Some inputs are more diffuse, such as those from urban runoff, agricultural fertilizers, pastures, and erosion from road construction and deforestation.

Severe eutrophication of important resource lakes, such as Lake Erie (Ragotzkie, 1974), has caused a need for rehabilitation. A variety of mitigation methods have been proposed and applied in efforts to recover pre-impact water quality conditions (Henderson-Sellers and Markland, 1987). There is concern, however, that mitigation efforts are sometimes being applied to naturally eutrophic lakes, for which restoration is unnecessary and probably futile. Because data on pre-disturbance lake conditions are lacking, paleolimnology is the only method available to determine the natural conditions of eutrophic lakes.

PALEOLIMNOLOGY

Paleolimnologists have repeatedly shown that lakes and their biota are very sensitive to events occurring within their catchments (reviewed in Smol, 1992). For example, the analysis of midge remains can reveal past trends in hypolimnetic oxygen levels and ion concentrations (Walker, 1987). The stratigraphic analysis of vegetative spores and pollen grains (palynology) can be used to indicate past land-use practices (MacDonald, 1990). The

study of algal remains has allowed reliable reconstructions of acidification (Stevenson et al., 1989), eutrophication (O'Sullivan, 1992), and salinification (Fritz, 1990) patterns. Of the many biomarkers available to paleolimnologists, diatoms are the most widely used group of indicators for eutrophication research.

Diatoms

Diatoms (Bacillariophyceae) represent a significant portion of the Earth's primary production, and their role as monitors of environmental quality is well recognized (Round et al., 1990). Diatom valves are particularly useful in paleolimnological studies because their siliceous cell walls are resistant to dissolution; diatom taxonomy is well defined, in that the size, shape, and sculpturing of their valves are taxon-specific; many species have well-defined environmental optima and tolerances; diatoms are present in almost any aquatic environment, including lakes, rivers, wetlands, oceans and soils; and fossil assemblages are often abundant and diverse (Dixit et al., 1992a).

Until recently, paleoenvironmental inferences have been based on broad qualitative interpretations of diatom assemblages, such as the Araphidae/Centrales (A/C) ratio for inferring trophic status (Stockner and Benson, 1967). Other studies based ecological reconstructions on the known autecological characteristics of a few common species (Stockner and Northcote, 1974). However, by using calibration sets (described in greater detail below), paleolimnologists have recently taken advantage of the aforementioned characteristics of ⁺ₕₑ bᵢᵍ question diatoms to design statistically robust and ecologically realistic methods for quantitatively reconstructing environmental trends.

Chrysophytes

Chrysophytes (Chrysophyceae) are another group of algae used in paleolimnological studies. The thickly-silicified chrysophyte resting cysts (stomatocysts), which are produced by all chrysophytes, are often well-preserved and sufficiently abundant in sediment profiles to document changes in chrysophyte communities (Rybak et al., 1991; Duff et al., 1994). Most chrysophyte species in temperate regions are more abundant in oligotrophic waters (Sandgren, 1988), and hence cyst abundance is often inversely correlated with lakewater nutrient concentrations. Using these empirical relationships, Smol (1985) developed the "chrysophyte cyst to diatom frustule ratio" as an approximate indicator of eutrophication trends in temperate lakes. In general, a decrease in the ratio indicates eutrophication trends (e.g. Smol, 1985; Stoermer et al., 1985a).

Paleolimnology and Eutrophication

Use of paleolimnological methods to study trends in lake trophic status can provide information to policy makers on the effects of cultural activities (Smol, 1992). Quite often, human activities such as agriculture, livestock farming and improper septic procedures result in degradation of lakewater quality, and paleolimnology is the only way to determine the extent of environmental damage in the absence of long-term monitoring data. For example, Stockner and Benson (1967) used fossil diatoms to track 80 years of sewage enrichment in Lake Washington. They observed a shift from *Melosira italica* to the eutrophic *Fragilaria crotonensis* coinciding with the establishment of the city of Seattle. Suburban housing development was shown by Brugam (1978) to cause diatom communities to shift to hypereutrophic assemblages. Brugam and Vallarino (1989) evaluated trophic disturbance in a study of short cores from four western Washington lakes, and demonstrated that prior to

settlement and land perturbation, these lakes were oligotrophic. Following settlement,

Asterionella formosa, an indicator of elevated nutrient concentrations, increased in abundance,

corresponding to deforestation and land development in the lakes' watersheds. O'Sullivan

(1992) documented the limnological consequences of sewage inputs, land use, fertilizer

application and numbers of livestock by inferring 75 years of eutrophication trends in

southwest England. The water quality in his study lake changed from "permissible" to

"dangerous" levels in the period from 1905-1980.

Diatoms have also been used to document eutrophication and recovery trends in lakes

following sewage treatment (Anderson et al., 1990). A sediment core from Lough Augher,

Northern Ireland showed an unambiguous record of diatom response to eutrophication, with

the characteristic increase in small *Stephanodiscus* species and other eutrophic indicators.

Anderson et al.'s (1990) study was particularly interesting because a core taken four years

later showed a dramatic change in the diatom communities, with mesotrophic taxa replacing

more eutrophic taxa. The recovery record corroborated water quality improvements following

effluent redirection for sewage treatment. Clearly, diatoms can accurately track trophic shifts

in any direction.

Paleolimnological Calibration

There have been many recent advances in paleolimnological inference methods, and

presently the most popular and robust approach is the application of surface sediment

calibration sets (Charles and Smol, 1994). Briefly, a large number of lakes (e.g. 40-80) are

selected to encompass a wide range of values of the environmental variables under

consideration (e.g. a total phosphorus range of 2 to 200 $\mu g/L$). At each lake, limnological

variables are measured (preferably for several years) and surface sediments (usually the top 1

cm) are collected. By analyzing the diatom taxa in the surface sediments of these lakes, a statistical relationship between current species assemblages and environmental variables can be determined. Transfer functions, using statistical approaches such as those described below, can then be constructed.

Canonical Correspondence Analysis (CCA) is used to generate ordination axes that are linear combinations of environmental variables chosen to maximize the dispersion of species along the ordination axes (ter Braak, 1987). CCA identifies the variables that are significantly influencing diatom distributions. Once the important variable(s) is (are) identified, weighted-averaging (WA) regression can then be used to estimate the environmental optima and tolerances of diatom taxa to that variable (Line and Birks, 1990). Weighted-averaging calibration is then used to infer quantitatively past lakewater chemistry from the optima and/or tolerances of diatom taxa in sedimentary assemblages.

The development and application of the above quantitative inference techniques has taken place within the last decade. Such approaches have been used extensively to track anthropogenic-related increases in lakewater pH (Birks et al., 1990a; Charles and Smol, 1994). Other applications have included stratigraphic reconstructions of lakewater temperature (Walker et al., 1991), salinity (Fritz, 1990), and metal concentrations (Cumming et al., 1992a; Kingston et al., 1992). A number of workers have recently used surface sediment calibration sets to develop statistical models for inferring trophic variables from diatom assemblages (Whitmore, 1989; Agbeti, 1992; Hall and Smol, 1992; Anderson et al., 1993; Christie and Smol, 1993; Fritz et al., 1993; Bennion, 1994; Dixit and Smol, 1994), and seven recent studies have employed diatoms and quantitative paleolimnological techniques to observe long-term shifts in trophic status. Christie and Smol (1993) found that the distributions of diatom taxa in 54 southeastern Ontario lakes were strongly influenced by

trophic variables such as lakewater nitrogen, phosphorus and chlorophyll *a* concentrations. Their model faithfully tracked the effects of deforestation and agricultural activity in the watersheds of impacted lakes in southern Ontario (Christie, 1993). Fritz et al. (1993) used a suite of forty-two Michigan lakes to create an inference model for phosphorus. Slight improvements in water quality were inferred from the diatom assemblages preserved in cores of four lakes; however the ordination technique was deemed unreliable considering the fine scale of the trophic shifts.

Anderson et al. (1993) tested a calibration set from Ireland on two loughs and established that diatom-inferred phosphorus profiles were highly correlated to suspected changes in trophic status. Bennion (1994) inferred total phosphorus concentrations for a highly eutrophic (present [TP] = 476 μg/L) lake in southeast England. Diatom-inferred [TP] indicated severe eutrophication resulting from sewage effluent. Also, decreases in inferred [TP] corresponded with the modernization of sewage treatment and effluent diversion. Bennion's (1994) model indicates that realistic reconstructions are possible even in extremely degraded systems.

Dixit and Smol (1994) compared diatom-inferred [TP] in surface and pre-industrial sediments of 66 northeastern U.S.A. lakes. Differences between present (top) and pre-industrial (bottom) values provided a regional assessment of eutrophication trends. Their reconstructions clearly indicated that, since pre-industrial times, the proportion of eutrophic lakes in the region increased.

Hall (1993) and Hall and Smol (1993) used a training set from British Columbia to infer total phosphorus in five southern Ontario lakes back in time to the period of the last deglaciation. Hall (1993) was able to track episodes of natural and anthropogenic eutrophication, however he acknowledged that the training set lacked highly eutrophic lakes

and was thus unreliable in hypereutrophic systems. Although present literature shows that

nutrient calibration sets can reliably describe past eutrophication events, most authors

generally concede that such diatom-based training sets are still in their infancy, and require

further development.

EUTROPHICATION PATTERNS IN BRITISH COLUMBIA

Circa A.D. 1858, an influx of European and American gold miners began the

extensive settlement and population explosion of inland BC (Anderson, 1947). The native

Indian way of life was rapidly changed to one of coexistence with whites. As tribal territory

became occupied by opportunist miners, the gold fields became more competitive and settlers

turned to homesteading as a more secure source of wealth. The most concentrated population

centres formed near lakes in inland BC. These lakes are believed to have received large

quantities of nutrients from anthropogenic sources, causing water quality degradation.

For several decades there has been an increasing demand for water from inland BC

population centres (Stockner and Northcote, 1974; Walker et al., 1994). In addition, the

disposal of wastes into aquatic systems has led to a serious degradation of surface water

quality. Many lakes in the southern Interior Plateau are presently eutrophic. Marked oxygen

depletions in hypolimnia are typical, as are dense summer blooms of blue-green algae.

Recreational fishing success has declined in many lakes due to eutrophication

Fishing industries have noticed a profound decline in fish populations (R. N. Nordin, pers.

comm.), and many of these lakes must be regularly stocked with rainbow trout to maintain

fisheries.

The major goal of this thesis was to undertake a paleolimnological investigation of

eutrophication trends in British Columbia lakes. Following this introduction and literature

review (Chapter 1), two manuscripts are presented. The first manuscript (Chapter 2) details

the construction of a weighted-averaging regression and calibration model for inferring [TP]

from fossil diatom assemblages in BC. A previously developed model for inferring lakewater

TP concentrations from sedimentary diatom assemblages (Hall and Smol, 1992) was expanded

to accomodate reconstructions of highly eutrophic conditions.

In Chapter 3, the above inference model was employed to infer TP concentrations

from diatom assemblages in sediment cores from six BC lakes which were suspected of

suffering from cultural eutrophication. Wood Lake (Okanagan Valley) was recognized to be

eutrophic in the early 1970s (Stockner and Northcote, 1974), but it has apparently undergone

some rehabilitation since 1971 with the installation of a distillery pumping station which

decreased the retention time of the lake (Walker et al., 1994). Dutch Lake is a eutrophic

system that has been proposed for mitigation (Nordin, 1982), but the natural trophic state of

the lake was unknown. A nearby lake (Pinantan) was also selected for this study because it

appears to suffer from similar problems as Dutch Lake. The city of Prince George, in central

BC, depends upon Tabor and Norman lakes for water supplies and recreational facilities, but

these lakes are also considered to have poor water quality (BC Ministry of Environment and

Parks, unpublished data). Finally, Charlie Lake, located in the Peace River Environmental

Planning Unit, 8 km northwest of Fort St. John, was selected for study because of a noted

degradation in water quality (Nordin and Pommen, 1985). Full site descriptions are in

Chapter 3.

A comparison of the eutrophication histories of these lakes provides important

information on the relative effects of cultural impacts on them. The following questions are

addressed in Chapter 3: Are the lakes naturally eutrophic or have they changed as a result of

cultural impacts? At what time, and to what extent, did changes occur? Have the lakes

followed similar patterns in their histories? Collectively, these data should assist limnologists and lake managers concerned about water quality changes in BC.

CHAPTER 2

AN EXPANDED WEIGHTED-AVERAGING REGRESSION AND CALIBRATION MODEL FOR INFERRING PAST TOTAL PHOSPHORUS CONCENTRATIONS FROM DIATOM ASSEMBLAGES IN EUTROPHIC BRITISH COLUMBIA (CANADA) LAKES

INTRODUCTION

Water quality has declined in many western Canadian lakes as a result of anthropogenic nutrient loading. These lakes have often been subject to perturbations since European settlement began in the mid-1800s, and in some areas, Indian settlements may have affected water quality well before this time (Woolliams, 1979). However, water quality information measured before 1950 is scarce or lacking for most systems, and realistic estimates of previous ecological conditions are often only available in the form of proxy data from the sediment record. Paleolimnological approaches can provide information on the causes, timing and extent of ecological disturbances. Such data are often necessary for proper management decisions (Smol, 1992). Moreover, with knowledge of the pre-disturbance state, realistic restoration goals can be established.

The siliceous remains (valves) of diatoms have been used as biomarkers to elucidate environmental conditions in lakes (e.g. Dixit et al., 1992a). Powerful statistical methods have been developed for inferring ecological conditions from diatom assemblages using multivariate ordination and weighted averaging (WA) regression and calibration based upon the optima and tolerances of several species (Birks et al., 1990b). Ecological parameters of the diatom species are determined by relating modern limnological variables to surface sediment diatom

assemblages. The species-environment relationships are then used to infer environmental conditions from fossil diatom assemblages (i.e. in a sediment core). These techniques are statistically and ecologically robust.

Inference models have been developed for inferring trophic variables in Florida (Whitmore, 1989), southern Ontario (Agbeti, 1992; Christie and Smol, 1992), northeast United States (Fritz et al., 1993; Dixit and Smol, 1994), Northern Ireland (Anderson et al., 1993) and southeast England (Bennion, 1994). A strong relationship has generally been found between diatom species composition in the surface sediments of the calibration lakes and the nutrient concentrations of the lakewater.

Hall and Smol (1992) described the relationship between several measured environmental variables and diatom distributions in 37 British Columbia (BC) lakes, and generated a WA model for inferring lakewater [TP] from sedimentary diatom assemblages. Their model was developed to infer historical [TP] within the range of 5 to 28 μg/L from fossil diatom assemblages, a rather limited range of trophic states. Because many lake management problems tend to develop above the upper limit of their model (i.e. above 30 μg/L), we attempted to expand the range of this transfer function.

The present study sought to expand the Hall and Smol (1992) calibration set to include more eutrophic lakes. This expansion was considered necessary for three reasons. First, only two lakes in the old set contain TP in ambient concentrations greater than 30 μg/L, however many BC lakes with water quality problems have much higher TP concentrations. By including additional lakes in the eutrophic range, we hope to provide more realistic inferences for highly eutrophic systems. Second, many of the small benthic diatoms (primarily *Fragilaria pinnata*) are common in BC lakes and exhibit a wide tolerance to trophic status, hence they are difficult to classify ecologically. As a result, they are considered relatively

poor as indicators. Expansion of the lake set should provide more calibration data to define more accurately the environmental optima of species already described by Hall and Smol (1992). In this way, we hoped that the obfuscating effect of small benthic diatoms can be reduced. Third, a superior error-estimating procedure (bootstrapping; described by Line et al., 1994) has recently become available. In this paper, we compare old error-estimation methods (used by Hall and Smol, 1992) to bootstrapped error-estimation methods using our data. We also describe the effect of transforming species data (Cumming and Smol, 1993a) on the old and new inference models.

Our primary goal was to develop a more robust quantitative model for inferring past lake trophic status in BC lakes. Throughout this study, comparisons are made between the original and updated models in order to outline changes incurred by expanding the lake set.

METHODS

Sixty-four study lakes were chosen from throughout BC (Fig. 2.1). Considering the heterogeneity of BC soils, geology, climate, vegetation and topography, it was inevitable that a broad range of physical, chemical and biological characteristics would be represented by the lake set (Table 2.1). We selected lakes to maximize the [TP] gradient, and the lakes range from 5 to 138 μg/L. Other components which can strongly influence diatom distributions were selected more conservatively. We selected lakes with pH between 6.2 and 8.5, conductivity less than 460 μS/cm, and, in order to avoid problems associated with sediment mixing, lake depths greater than 4 m. Six biogeoclimatic zones (defined by Beil et al., 1976) were included in the study area: the Interior Douglas Fir, Coastal Douglas Fir, Caribou Aspen - Lodgepole Pine, Interior Western Hemlock, Boreal White and Black Spruce, and

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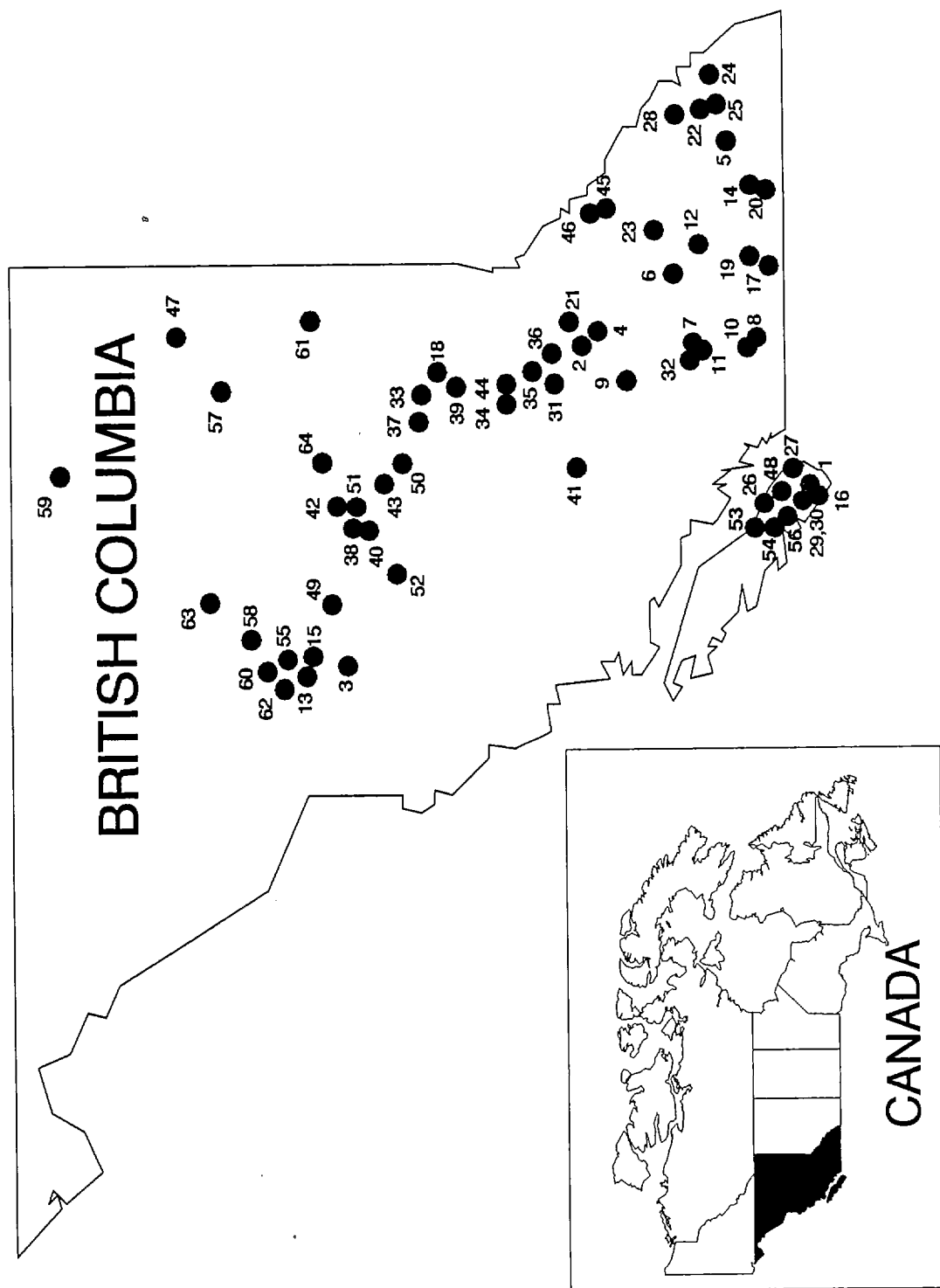
Table 2.1: Selected limnological data from the 64 study lakes in British Columbia. Only the important variables identified by canonical correspondence analysis are shown here.

Lake name	Lake no.	Location (N°,W°)	Zone	TP (μg/L)	Conductivity (μS/cm)	Max. Depth (m)	Mg (mg/L)	pH
Langford	1	48°30',123°40'	CDF	9	160	15	3.0	7.9
King West	2	51°45',120°50'	IDF	10	262	18	3.9	8.1
Hilltout	3	55°00',127°00'	CALP	9	75	13	1.4	7.7
King East	4	51°45',120°50'	IDF	11	147	13	3.7	8.0
Kootenay	5	49°40',116°50'	IWH	--	157	154	5.0	8.0
Gardom	6	50°40',119°10'	IDF	16	455	24	20.6	8.2
Davis	7	49°50',120°45'	IDF	13	318	17	10.5	8.2
McCaffrey	8	49°30',120°30'	IDF	9	399	11	14.8	8.3
Pavillion	9	50°55',121°40'	IDF	10	392	56	19.8	8.4
Laird	10	49°30',120°30'	IDF	9	390	11	14.7	8.4
Boss	11	49°50',120°45'	IDF	14	227	11	0.2	8.3
Echo	12	51°10',118°45'	IDF	8	224	50	4.9	8.2
Tyhee	13	54°45',127°15'	CALP	46	261	22	10.0	8.5
Champion #3	14	49°15',117°30'	IWH	5	101	24	1.2	7.9
McQuarrie	15	54°45',126°50'	CALP	9	92	11	1.6	7.9
Florence	16	48°30',123°30'	CDF	15	132	6	2.6	7.6
Wilgress	17	49°10',118°40'	IDF	8	410	18	5.2	8.2
Green	18	52°30',121°00'	IDF	15	239	17	8.8	8.3
Jewel	19	49°10',118°40'	IDF	13	158	21	3.0	8.4
Champion #2	20	49°15',117°30'	IWH	7	120	11	1.3	8.0
Lac de Roche	21	51°45',120°45'	IDF	10	229	47	7.9	8.4
Columbia	22	50°15',115°15'	IDF	12	320	6	15.7	8.5
Three Valley	23	50°50',118°15'	IDF	7	63	--	--	7.5
Whiteswan	24	50°15',115°30'	IDF	10	268	20	10.5	8.4
Alces	25	50°15',115°50'	IDF	7	330	15	12.1	8.4
Holden	26	49°05',123°45'	CDF	45	110	6	2.0	8.0
Elk	27	48°30',123°25'	CDF	23	159	15	3.8	8.3
Windermere	28	50°30',115°30'	IDF	8	240	8	14.0	8.4
Glen	29	48°30',123°30'	CDF	16	137	15	3.6	7.7
Durrance	30	48°30',123°25'	CDF	9	154	17	2.8	7.9
Sneezie	31	51°55',121°10'	CALP	9	252	24	8.0	8.4
Tahla	32	49°50',120°45'	IDF	12	278	8	9.2	8.2
Spanish	33	52°35',121°20'	IDF	8	97	73	3.5	7.6
Spout	34	52°00',121°20'	CALP	17	225	18	9.1	8.3
Lang	35	52°00',121°00'	IDF	14	95	73	4.1	7.8
Hawkins	36	51°55',120°55'	IDF	24	193	18	7.0	8.2
Morehead	37	52°35',121°45'	IDF	26	82	17	3.0	7.7

Lake name	Lake no.	Location (N°,W°)	Zone	TP (µg/L)	Conductivity (µS/cm)	Max. Depth (m)	Mg (mg/L)	pH
Little Bobtail	38	53°40',123°40'	CALP	85	167	15	11.9	8.3
Keno	39	52°20',121°20'	IWH	--	111	33	3.2	8.2
Woodcock	40	53°40',123°40'	CALP	28	116	15	4.7	7.5
Elkin	41	51°40',123°45'	CALP	17	71	35	2.2	7.8
Dahl	42	53°50',123°15'	CALP	19	131	24	9.6	8.0
Milburn	43	53°00',122°40'	CALP	23	191	8	7.3	7.9
Murphy	44	52°00',121°15'	IDF	19	164	23	8.2	8.2
Aid	45	51°40',117°20'	IDF	8	228	5	13.6	8.1
Comfort	46	51°40',117°20'	IDF	5	269	6	13.3	8.2
Charlie	47	56°40',121°00'	BWBS	103	169	12	3.4	8.5
Bullocks	48	48°90',123°70'	CDF	40	126	8	3.1	7.4
Nellian	49	54°40',125°85'	SBS	38	26	5	1.4	7.4
Dragon	50	52°95',122°40'	CALP	135	360	7	13.5	8.1
Grizzly	51	53°75',123°50'	CALP	36	97	7	5.1	7.4
Top	52	53°15',125°45'	CALP	138	90	20	2.3	7.9
Somenos	53	48°90',123°80'	CDF	118	125	7	--	8.6
McKay	54	49°10',124°00'	CDF	78	59	--	1.9	6.2
Round	55	54°70',127°00'	CALP	53	232	20	9.9	8.0
Quamichan	56	48°95',123°90'	CDF	34	123	8	--	8.0
Chinaman	57	56°20',122°10'	CALP	31	127	6	4.4	8.0
Fission	58	55°40',126°55'	SBS	23	32	8	1.21	6.5
Beaver	59	58°95',123°10'	BWBS	21	203	9	6.9	8.5
Seymour	60	54°80',127°20'	CALP	22	83	9	4.0	7.4
Boot	61	55°15',120°45'	SBS	28	198	6	6.6	8.3
Kathlyn	62	54°80',127°20'	CALP	22	58	10	1.4	7.0
Tutizzi	63	56°30',125°65'	SBS	21	45	30	0.4	7.4
Chief	64	54°10',123°10'	SBS	26	67	6	2.7	7.5

Sub-Boreal Spruce regions.

All sampling was performed by the British Columbia Ministry of the Environment from the deepest portion of each lake. Water for chemical analyses was taken from a 1 m depth using a Van Dorn sampler. The top 1 cm of surface sediment was carefully collected with an Ekman grab. Hall and Smol (1992) provide complete details concerning sampling.

The water chemistry data for the original 46 lakes has been updated and includes additional data that has been made available since Hall and Smol's (1992) study was completed (BC Ministry of Environment, unpublished data). The new data were primarily trophic status measurements from archived documents. Newly-acquired data was combined with Hall and Smol's original measurements, and average TP concentrations, for example, were recalculated for some lakes.

Laboratory Analysis

Many of the physical and chemical data were measured prior to surface sediment sampling, and chemical measurements for all lakes were taken during sediment sampling; chemical measurements presented here are averages of available data. Measurements took place in 1983, 1984, and 1986 (Hall and Smol, 1992). Chemical analysis of water samples was performed by the British Columbia MOE using methods described by McQuaker (1976). The following environmental and limnological variables were measured: altitude, watershed area, surface area, maximum depth, mean depth, Secchi depth, pH, specific conductivity, total dissolved solids (TDS), hardness, and concentrations of ammonium, nitrate, organic nitrogen, total phosphorus (TP), calcium, iron, magnesium, manganese, and aluminum. In the few cases where data were missing, median values were substituted during statistical analyses. To satisfy the statistical assumptions of ordination, values for variables with skewed

distributions (TP, Fe, Mg, Mn, Al, watershed area, lake area and maximum depth) were normalized using a ln(x+1) transformation (Zar, 1984). Ammonium and nitrate were eliminated from numerical analyses because they were below detection limits for most of the lakes. Organic nitrogen data were lacking for several lakes, and so this variable was also omitted.

Diatom slides were prepared and counted using standard techniques, as described in Hall and Smol (1992).

Data Analysis

Ordination: The computer program CANOCO (ter Braak, 1988) version 3.12 (ter Braak, 1990a, b, unpublished update) was used for the species and sample ordinations. Canonical correspondence analysis (CCA), using forward selection and a Monte Carlo permutation test with 999 unrestricted permutations (ter Braak, 1990a), was employed to determine which environmental variables explained the maximum amount of variance in the species data. Diatom species that occurred in more than one sample and greater than 1% abundance in at least one sample were used in ordination calculations.

Data screening to remove extraneous data followed Hall and Smol (1992). Briefly, it was necessary to remove: (1) environmental variables that had little or no influence on diatom distributions; (2) "outlier" samples with particularly unusual diatom assemblages or environmental characteristics; and (3) redundancies in environmental information, such as variables that were correlated among lakes but did not independently influence diatom distributions. Inspections of CCA, a detrended correspondence analysis (DCA) and a principal component analysis (PCA), identified superfluous biological and environmental data. Ultimately, statistical calculations included 64 lake sediment samples, 150 diatom species, and

16 environmental variables.

Weighted-Averaging Regression and Calibration: It was assumed that the diatom assemblages were representative of living communities during the period of available environmental chemistry measurements, and that their sedimentary species distributions reflect environmental characteristics. If diatom species show a strong relation to a particular environmental variable, then realistic inferences should be possible from diatom assemblages preserved in a sediment core. CCA was used to demonstrate the strength of the relationship between diatom taxa and environmental variables. This was done by testing the significance of the TP axis in a constrained CCA with [TP] as the sole environmental variable (Jongman et al., 1987). WACALIB version 3.3 (Line et al., 1994) was subsequently used to infer [TP] based on the weighted averages (optima) of the diatom species. Classical regression (Birks et al., 1990b) produced the most evenly distributed residuals and was thus the only type of regression used in these analyses.

Error Estimation: Advanced statistical methods of error estimation have recently become available (Line et al., 1994). The root mean square error (RMSE) of prediction for environmental inferences, training samples, fossil samples, and the training set as a whole, was estimated using bootstrapping analysis with the computer program WACALIB 3.3 (Line et al., 1994). Birks et al. (1990a) describe the theory of bootstrapping. To estimate error components of the BC calibration set, we ran WACALIB with 999 bootstrap cycles and square root transformation of the species data (transforming the species data effectively reduces the overwhelming influence of highly abundant species, and gives more weight to less abundant taxa). For each cycle, the computer program WACALIB performed the following steps: WACALIB randomly selected 64 samples with replacement (that is, samples could be chosen more than once), and the samples not chosen formed a test set; the bootstrap training

set was then used to infer [TP] for each lake in the test set. The prediction error due to the error in estimating the optima and tolerances of taxa (s_1) is the standard deviation of the inferred values; s_2 is the error due to variations in the relative abundance of taxa at a given environmental value, and so this component remains constant. The RMSE of prediction was calculated as the square root of the sum of squares of these two components, and provides an overall estimation of the integrity of prediction of, for example, inferred [TP] from the training set. Birks et al. (1990a), Kingston et al. (1992) and Pienitz et al. (1994) demonstrate bootstrapping applications.

The bootstrap-estimated RMSE is considered superior to the RMSE calculated from simple WA regression (Birks et al., 1990a). For example, using Hall and Smol's (1992) method, inferred [TP] calculated from a diatom assemblage is the weighted average of the [TP] optima of all the species present, so the RMSE calculation is based on their single set of 37 lakes. During bootstrapping, a more realistic, if somewhat pessimistic, estimate is obtained. The biased circular inference is eliminated by the random creation of 999 transfer functions and calculating the mean RMSE of these inferences.

Testing the model

A diatom stratigraphy used in a previous study (Walker et al., 1994a) has been selected to compare the original (37 lake) and updated (59 lake) models with respect to inferring [TP] from fossil diatom assemblages. Wood Lake was chosen because a great deal of historical information is available regarding its past trophic status (Walker et al., 1994a; Stockner and Northcote, 1974) and also because the downcore diatom assemblages show interesting diatom shifts related to human activities.

RESULTS

Data screening

As previously described, the three nitrogen variables were eliminated because of several undetectable concentrations and missing data. Total sulfur, hardness, calcium and mean depth were also excluded, as they did not independently influence diatom distributions and/or they were significantly correlated with other environmental variables (Table 2.2). The remaining environmental variables were pH, conductivity, maximum depth, Secchi depth, altitude, surface area, watershed area, total phosphorus, magnesium, manganese, iron and aluminum. The addition of 18 lakes filled gaps in the environmental gradients of a number of important variables. As a result, all of the lakes identified as "outliers" by Hall and Smol (1992) could now be included in the data set. None of the 64 samples were removed according to the defined criteria (DCA and PCA analyses; see Hall and Smol, 1992).

A preliminary test of the 64 lake data set to infer [TP] (using WACALIB, as described below) showed that [TP] values were consistently underestimated (by more than 50 μg/L) in those lakes with measured [TP] values greater than ~ 85 μg/L (Fig. 2.2). Based on this poor relationship, four lakes with high [TP] were removed. Additionally, Gardom Lake (6, numbers in parentheses following lake names refer to the lake number in Fig. 2.1 and Table 2.1) appeared as an extreme outlier (Fig. 2.2), likely due to the influence of conductivity and related variables (see Table 2.1), and so it was also removed from the lake set. A secondary CCA was performed following the removal of these lakes.

Canonical Correspondence Analysis

An exploratory CCA ordination of the 59 lake data set constrained to the 11

Table 2.2: Correlation coefficients for the environmental variables used in a preliminary CCA. Boldface values identify significant (P < 0.1) correlations leading to the removal of four non-independent variables: maximum depth, TDS, hardness and calcium.

	Altitude	Water-shed	Lake area	Mean depth	Max. depth	Secchi depth	pH	TDS	Conduc-tivity	Hardness	TP	Ca	Fe	Mg	Mn	Al
Altitude	1.000															
Watershed	0.124	1.000														
Lake area	0.003	0.792	1.000													
Mean depth	-0.002	0.464	0.445	1.000												
Maximum depth	0.130	0.451	0.429	0.931	1.000											
Secchi depth	0.306	0.065	-0.088	0.454	0.470	1.000										
pH	0.053	0.092	0.054	0.153	0.113	0.254	1.000									
TDS	0.128	-0.083	-0.171	0.070	0.015	0.300	**0.593**	1.000								
Conductivity	0.113	-0.039	-0.129	0.115	0.042	0.384	**0.647**	**0.957**	1.000							
Hardness	0.170	0.018	-0.108	0.099	0.051	0.380	**0.605**	**0.889**	**0.952**	1.000						
TP	-0.329	0.017	0.237	-0.185	-0.233	-0.537	-0.165	-0.153	-0.221	-0.220	1.000					
Ca	0.168	-0.084	-0.215	0.141	0.082	0.450	**0.587**	**0.871**	**0.922**	**0.954**	-0.261	1.000				
Fe	-0.187	-0.213	-0.212	-0.344	-0.340	-0.588	-0.329	-0.162	-0.336	-0.374	0.247	-0.395	1.000			
Mg	0.070	0.153	0.104	0.036	0.011	0.103	0.510	**0.654**	**0.745**	**0.837**	-0.009	0.674	-0.274	1.000		
Mn	-0.229	-0.188	-0.174	-0.308	-0.343	-0.435	-0.107	0.073	-0.023	-0.115	0.288	-0.133	0.478	-0.030	1.000	
Al	0.068	-0.057	0.008	-0.168	-0.137	-0.173	-0.217	-0.126	-0.154	-0.172	0.110	-0.206	0.264	-0.092	0.080	1.000
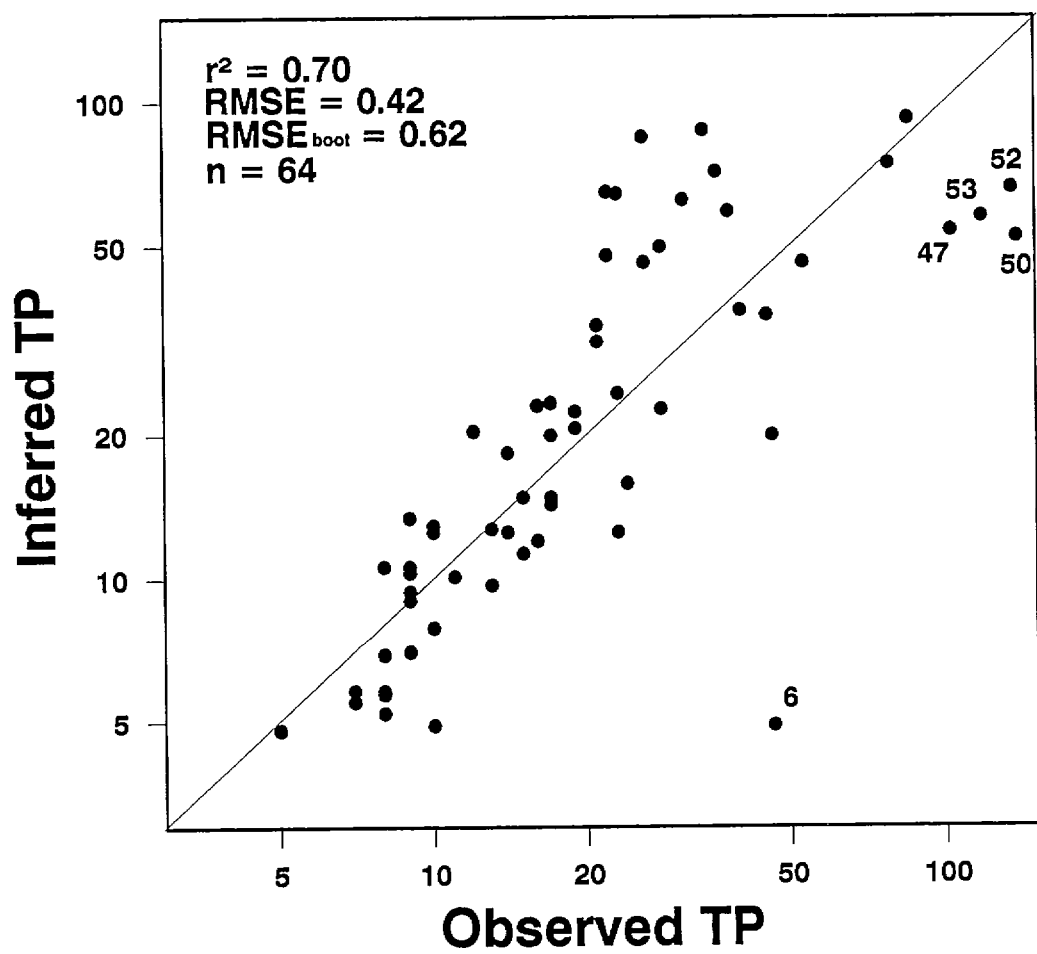
remaining variables produced the following explanations for variance in the data. CCA axis 1 ($\lambda=0.33$) and axis 2 ($\lambda=0.30$) explained 15.2% of the cumulative variance in the weighted averages of the diatom taxa (Fig 2.3). A large proportion (50.6%) of the diatom-environmental relationship was explained by the first two axes. Also, the correlation between the species and environmental variables was high for axis 1 (0.91) and axis 2 (0.78). Multiple t-tests determined the significant contributors to CCA axes 1 and/or 2: lake surface area (axis 1), maximum depth (axis 2), conductivity (axis 1), total phosphorus (axis 1 and 2), Fe (axis 1) and Al (axis 2) (Table 2.3). When compared to the original calibration set, the contributions of most environmental variables to the first two axes did not change drastically with the addition of lakes. Secchi depth, however, was no longer a significant contributor to either axis, and maximum depth no longer contributed significantly to axis 1.

As expected, [TP] and Secchi depth arrows pointed in opposite directions, indicating the typical negative correlation of these trophic measurements (Fritz et al., 1993). Multiple Monte Carlo permutation tests in CCA indicated which variables made significant (999 permutations, $P \leq 0.01$) contributions to explaining the variation in the diatom assemblages, using both the original (Hall and Smol, 1992) and expanded lake sets (Table 2.4). Several ordinations constrained to single variables (ter Braak, 1990a, b) were used to assess the strength with which each variable influenced diatom distributions. In the analysis for each variable, the first eigenvalue represents the proportion of species variance explained by that variable, and the second eigenvalue represents all additional variance not attributable to that variable. The ratio of the two eigenvalues is used to describe the relative strength of the environmental variables constrained to the first axis. For both lake sets, conductivity exerted the strongest influence on the diatoms. The effect of [TP] was secondary, but still high, and was followed by maximum depth and magnesium. The eigenvalue ratios (Table 2.4) suggest

Table 2.3: Canonical coefficients of 12 environmental variables included in the canonical correspondence analysis and their approximate t-values (boldface t-values were significant at $P \leq 0.05$; n = 57, df = 45). Comparisons with the original (37-lake) data set are made. Variables not considered by Hall and Smol (1992) are marked with "—".

Environmental variable	Old model Canonical coefficient		Old model t-value		New model Canonical coefficient		New model t-value	
	Axis 1	Axis 2	Axis 1	Axis 2	Axis 1	Axis 2	Axis 1	Axis 2
Altitude	0.16	0.00	1.72	0.03	0.029	-0.017	0.54	-0.20
Watershed area	—	—	—	—	-0.119	0.058	-1.34	0.40
Surface area	0.27	0.13	**2.36**	0.98	0.232	0.316	**2.28**	1.92
Maximum depth	-0.56	0.37	**-5.02**	**2.83**	-0.098	-0.687	-1.40	**-6.06**
Secchi depth	0.23	0.45	1.88	**3.18**	0.120	0.161	1.40	1.16
pH	—	—	—	—	0.007	-0.009	0.09	-0.08
Specific conductivity	1.65	0.48	**5.91**	1.47	0.648	-0.257	**6.41**	-1.57
Total phosphorus	-0.30	0.64	**-2.77**	**4.99**	-0.226	-0.256	**-3.32**	**-2.32**
Fe	-1.37	0.20	**-4.81**	0.59	0.163	-0.218	**2.34**	-1.93
Mg	—	—	—	—	-0.026	0.175	-0.30	1.26
Mn	—	—	—	—	-0.031	0.032	-0.50	0.31
Al	0.16	-0.29	1.73	**-2.67**	-0.017	0.222	-0.32	**2.58**

Table 2.4: Strength of the relationships between diatoms and the 12 environmental variables used in the ordination, as estimated by a series of ordinations constrained separately to each of the environmental variables. The variables are ranked according to strength of influence over diatom assemblages according to the eigenvalue ratio, axis 1 versus axis 2. Values from the original and updated calibration sets are presented for comparison.

Environmental variable	Old training set			New training set		
	λ_1	λ_2	λ_1/λ_2	λ_1	λ_2	λ_1/λ_2
Specific conductivity	0.28	0.50	0.56	0.31	0.60	0.52
Total phosphorus	0.23	0.52	0.44	0.25	0.60	0.42
Maximum depth	0.22	0.46	0.48	0.24	0.59	0.41
Magnesium	0.21	0.50	0.42	0.24	0.60	0.40
pH	0.22	0.51	0.43	0.23	0.60	0.38
Secchi depth	0.22	0.52	0.42	0.19	0.60	0.32
Iron	0.12	0.52	0.23	0.18	0.58	0.31
Lake surface area	0.17	0.52	0.33	0.16	0.59	0.27
Aluminum	0.13	0.50	0.26	0.15	0.59	0.25
Watershed area	0.16	0.51	0.31	0.13	0.60	0.22
Manganese	0.12	0.52	0.23	0.12	0.60	0.20
Altitude	0.10	0.52	0.19	0.09	0.60	0.15

that reconstructions of these four variables are acceptable from the data set. There were not many differences between the old and new models; however, Hall and Smol's (1992) analyses indicated that maximum depth exerted a stronger influence on diatom distributions than TP. With the addition of more eutrophic lakes, [TP] had a greater relative influence on diatom distributions than depth.

The ordination of the sample scores effectively separated high and low [TP] lakes along the TP axis (Fig. 2.3). Lakes with relatively high levels of [TP] (>20 μg/L), such as Holden (26), Seymour (60), Elk (27), Morehead (37), and Hawkins (36), occur on the positive side of the TP vector, whereas low [TP] (<11 μg/L) lakes (e.g. McCaffey (8), DeRoche (21), Laird (10), and many others) occur in the opposing direction.

Planktonic taxa characteristic of mesotrophic and eutrophic conditions (e.g. *Aulacoseira italica* var. *subarctica* (99, numbers in parentheses following diatom taxa refer to the taxon numbers in Fig. 2.4), *Au. distans* (91), *Au. granulata* var. *angustissima* (145), *Cyclotella stelligera* (75) and *Au. undulata* (106)) are positioned along the TP vector in the lower left quadrant of the CCA biplot (Fig. 2.4). Other planktonic taxa (e.g. *Stephanodiscus minutulus* (78), *Fragilaria crotonensis* (107), *Fragilaria capucina* var. *mesolepta* (94), *Cyclostephanos tholiformis* (134) and *Asterionella formosa* (86)) occur between the TP and maximum depth vectors. In the opposing quadrant, low-nutrient planktonic diatoms such as *C. bodanica* var. *affinis* (26) and *C. meneghiniana* (34) occur. As expected, planktonic taxa are roughly distributed in the lower half of the biplot, in association with the depth vector, whereas most of the benthics are abundant in shallower lakes, and occur in the upper half of the ordination diagram. The effect of conductivity is also prominent. Indicators commonly found in moderately saline conditions (*Navicula laevissima* (28), *Amphora libyca* (41) and *Achnanthes pinnata* (33)) occurred along the conductivity arrow.
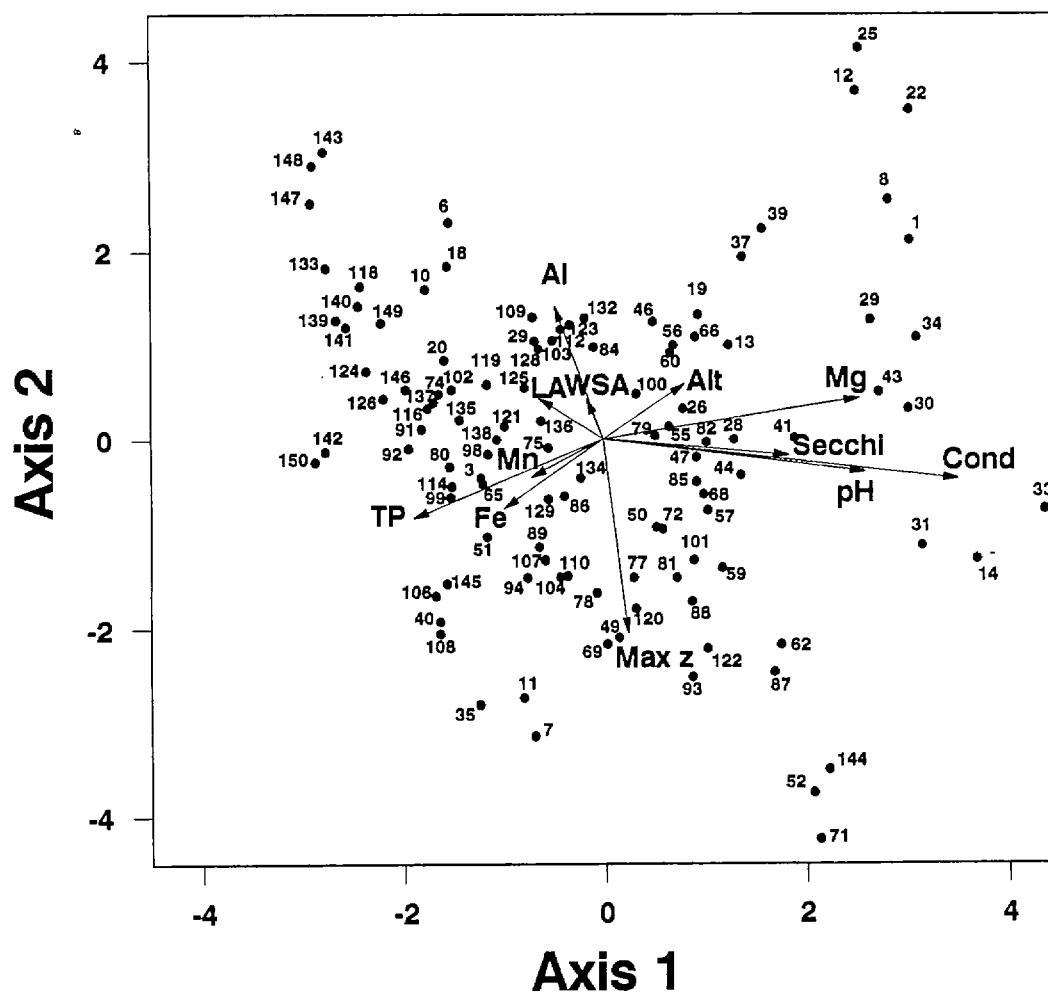
Weighted-averaging models

When using simple WA regression and calibration with classical deshrinking, the relationship between observed versus inferred [TP] did not improve after the addition of eutrophic lakes (Figs. 2.5a, b). In fact, the regression coefficient (r^2) decreased from 0.75 to 0.61 and the RMSE increased from 0.25 to 0.47. The same procedure employing transformation of species data for the old data set had little effect on these values (r^2 increased from 0.75 to 0.76 and the RMSE decreased from 0.25 to 0.22; Figs. 2.5a, c). However, transformation of species data showed a significant improvement over untransformed data in the expanded data set (r^2 increased to 0.73 and RMSE decreased to 0.33; Figs. 2.5b, d). The differences between r^2 and RMSE in the old and new models were also less important when species data were transformed (Figs. 2.5c, d). Diatom-inferred [TP] values were also more highly correlated to measured [TP] following screening of the new lake set (Figs. 2.2, 2.5d). The r^2 derived from bootstrapping and transforming species data in the old data set was much lower than other methods ($r^2 = 0.30$; Figs. 2.5a, c, e). The same procedure used on the new data also provided a lower r^2 (0.46) and higher $RMSE_{boot}$ (0.48) than other methods (Figs. 2.5b, d, f). However, a higher regression coefficient was obtained in the new model when compared to the old model (Figs. 2.5e, f). Also, the $RMSE_{boot}$ was somewhat higher for the new data than the old data. Our new model provided a fairly even distribution of residuals (Fig. 2.6), although [TP] is somewhat overestimated in lakes with [TP] between 20 and 40 $\mu g/L$, and underestimated in more eutrophic (>40 $\mu g/L$) lakes.

Table 2.5 presents estimates of [TP] optima for the 150 diatom taxa used in our model, calculated using WACALIB, and Figure 2.7 graphically illustrates how species optima changed following expansion of the lake set. The [TP] optima represented by the taxa increased from a range of 16.5 (6.8 to 23.3) $\mu g/L$ (Hall and Smol, 1992) to 35.9 (6.0 to
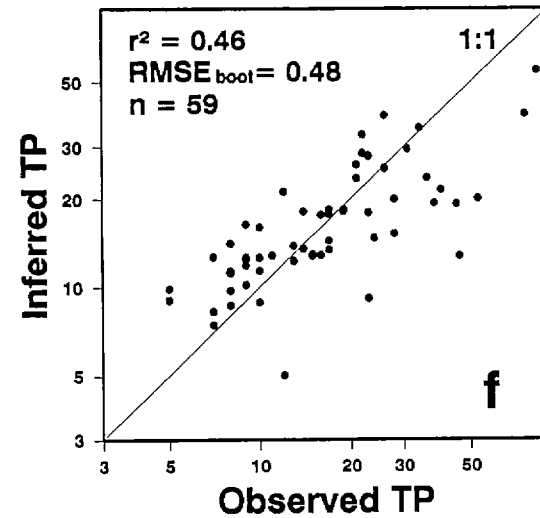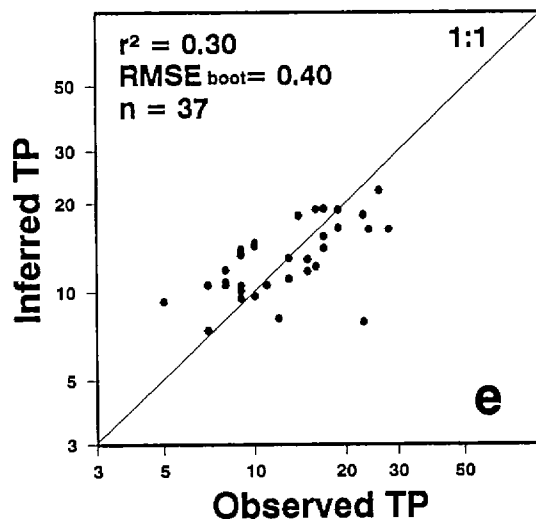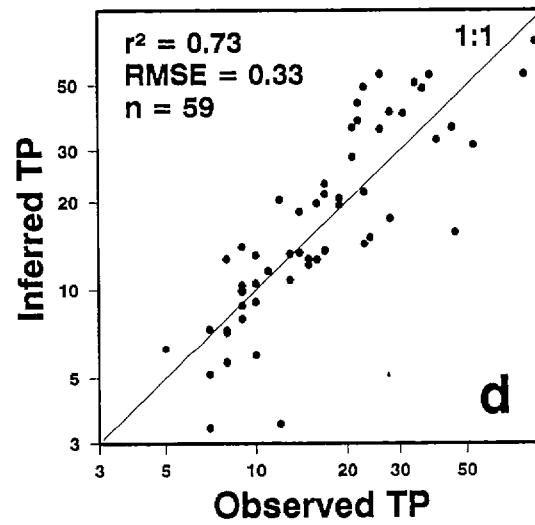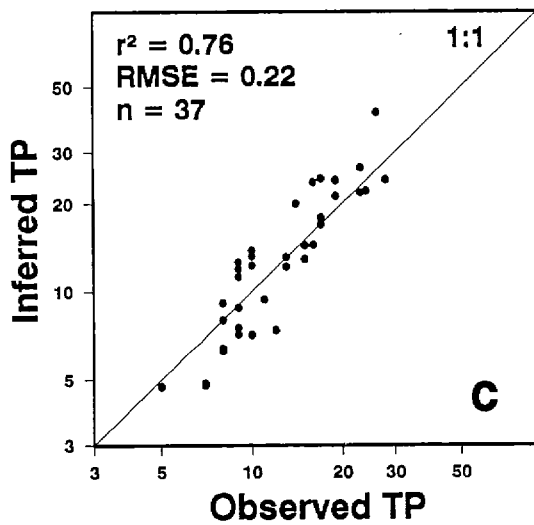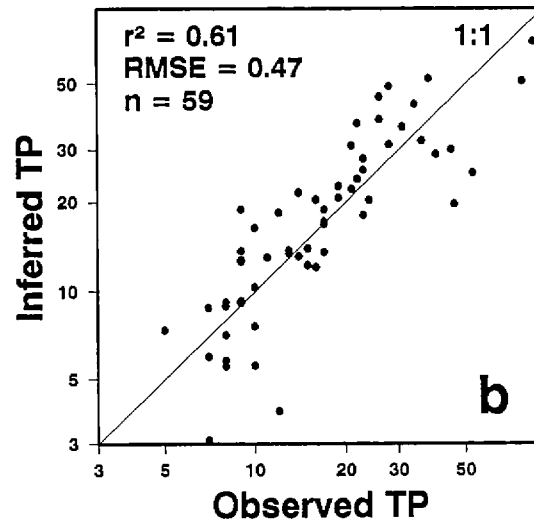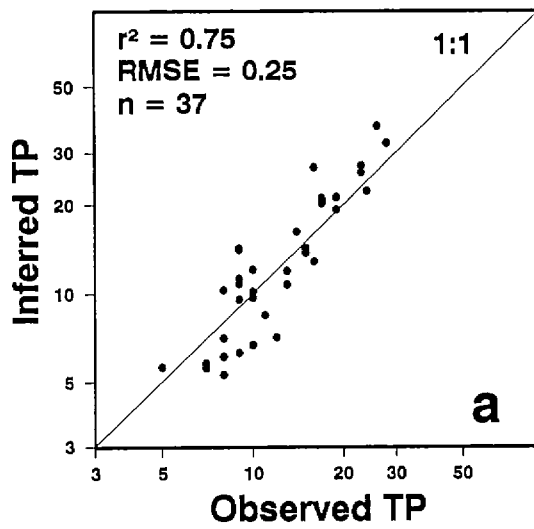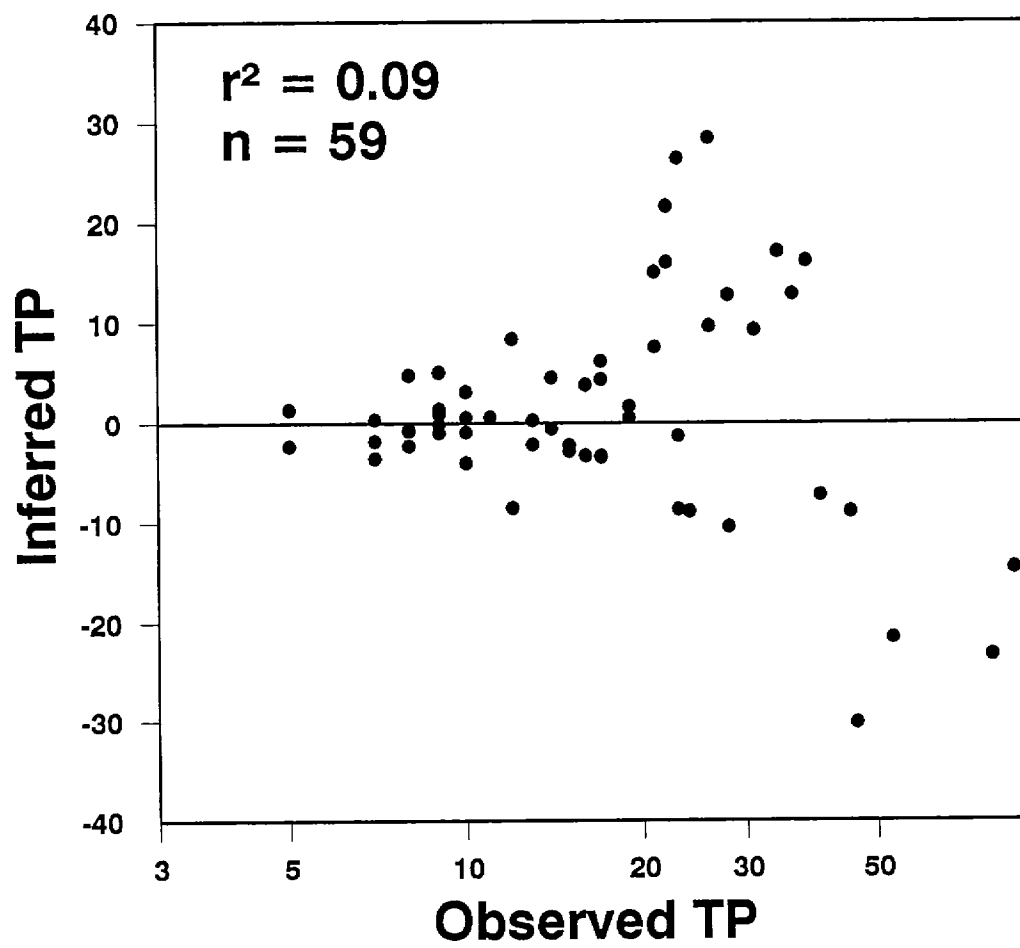
Table 2.5: Weighted averages of diatom taxa to [TP] estimated using weighted averaging calibration and regression. Included are values indicating the number of lakes in which each species occurred. Taxa labelled with "BC" have a previously unknown taxonomic description, but were observed by Hall and Smol (1992).

Number	Diatom taxon	TP optimum (μg/L)	Number of occurrences
1	*Fragilaria brevistriata* var. *capitata* Hérib.	6.0	7
2	*Aulacoseira* sp. 12 PIRLA	6.8	2
3	*Achnanthes levanderi* Hust.	6.9	3
4	*Navicula explanata* Hust.	7.3	3
5	*Denticula kuetzingii* Grun.	7.8	6
6	*Fragilaria virescens* var. *exigua* Grun.	7.8	2
7	*Cyclotella kuetzingiana* var. *planetophora* Fricke	8.0	2
8	*Fragilaria* sp. 4 PIRLA	8.0	8
9	*Anomoeoneis vitrea* (Grun.) R. Ross	8.2	12
10	*Achnanthes bioreti* Berm.	8.2	4
11	*Cyclotella kuetzingiana* var. *radiosa* Fricke	8.3	5
12	*Cymbella diluviana* (Krasske) M. B. Florin	8.4	9
13	*Fragilaria* sp. 3 PIRLA	8.6	4
14	*Amphora parallelistriata* Manguin	8.7	3
15	*Cymbella cesatii* (Rabenh.) Grun.	9.0	4
16	*Stephanodiscus* sp. 5 BC	9.0	2
17	*Stauroneis phoenicenteron* (Nitzsch) Ehrenb.	9.1	8
18	*Synedra radians* Kütz.	9.2	2
19	*Fragilaria brevistriata* var. *inflata* (Pant.) Hust.	9.3	25
20	*Diatoma hiemale* var. *mesodon* (Ehrenb.) Grun.	9.4	5
21	*Navicula cryptocephala* Kütz.	9.5	16
22	*Amphora thumensis* (Mayer) Krieger	9.6	13
23	*Gomphonema angustum* (Agardh)	9.6	5
24	*Cymbella minuta* Hilse ex Rabenh.	9.8	15
25	*Achnanthes biasolettiana* Grun.	10.0	20
26	*Cyclotella bodanica* var. *affinis* Grun.	10.0	18
27	*Achnanthes microcephala* (Kütz.) Cleve	10.1	12
28	*Navicula laevissima* Kütz.	10.2	15
29	*Navicula utermoehlii* Hust.	10.4	7
30	*Mastogloia smithii* Thwaites ex W. Smith	10.4	8
31	*Achnanthes conspicua* A. Mayer	10.6	27
32	*Achnanthes linearis* (W. Smith) Grun.	10.6	15
33	*Achnanthes pinnata* Hust.	10.7	4
34	*Cyclotella meneghiniana* Kütz.	10.7	5
35	*Tabellaria fenestrata* (Lyngb.) Kütz.	10.8	2
36	*Cymbella microcephala* Grun.	10.8	21
37	*Navicula subminuscula* Manguin	10.8	14
38	*Gomphonema subtile* Ehrenb.	10.9	4
39	*Fragilaria* sp. 2 PIRLA	10.9	16
40	*Achnanthes suchlandtii* Hust.	11.0	7
41	*Amphora libyca* Ehrenb. ex Kütz.	11.0	27
42	*Epithemia argus* (Ehrenb.) Kütz.	11.2	13
43	*Amphora pediculus* (Kütz.) Grun. ex A Schmidt	11.3	42
44	*Synedra delicatissima* Grun.	11.4	5
45	*Cyclotella* cf. *radiosa* (Grun.) Lemm.	11.4	3
46	*Fragilaria pinnata* var. *intercedens* (Grun.) Hust.	11.5	25
47	*Cyclotella bodanica* Grun.	11.6	28
48	*Navicula radiosa* Kütz.	11.8	13
49	*Rhizosolenia eriensis* W. Smith	11.8	10
50	*Stephanodiscus alpinus* Hust.	11.9	15
51	*Rhopalodia gibba* (Ehrenb.) O. Müll.	11.9	7

Number	Diatom taxon	TP optimum (μg/L)	Number of occurrences
52	*Cyclotella* sp. 2 BC	12.0	2
53	*Achnanthes clevei* Grun.	12.1	20
54	*Fragilaria pinnata* var. *lancettula* (Schum.) Hust.	12.2	32
55	*Nitzschia fonticola* Grun.	12.2	27
56	*Fragilaria brevistriata* Grun.	12.2	52
57	*Asterionella ralfsii* W. Smith	12.2	5
58	*Achnanthes detha* Hohn & Hellerman	12.3	17
59	*Cymbella delicatula* Kütz.	12.5	10
60	*Fragilaria construens* var. *venter* (Ehrenb.) Grun.	12.5	47
61	*Nitzschia denticula* Grun.	12.6	15
62	*Synedra ulna* var. *chaseana* Thomas	12.6	6
63	*Synedra parasitica* (W. Smith) Hust.	12.7	19
64	*Aulacoseira italica* var. *tenuissima* (Grun.) Simonsen	12.7	13
65	*Fragilaria* sp. 1 BC	12.8	20
66	*Achnanthes exigua* Grun.	12.8	26
67	*Cyclotella kuetzingiana* Thwaites	13.0	13
68	*Navicula cari* Ehrenb.	13.0	15
69	*Stephanodiscus medius* Håkansson	13.0	13
70	*Fragilaria lapponica* Grun.	13.1	35
71	*Stephanodiscus* sp. 2 BC	13.2	2
72	*Synedra cyclopum* Brutschy	13.3	12
73	*Navicula schadei* Krasske	13.4	9
74	*Achnanthes minutissima* Kütz.	13.5	42
75	*Cyclotella stelligera* (Cleve & Grun.) Van Heurck	13.5	37
76	*Navicula subatomoides* (Hust.) Lange-Bertalot & Archibald	13.8	26
77	*Diatoma tenue* var. *elongatum* Lyngb.	13.8	8
78	*Stephanodiscus minutulus* (Kütz.) Cleve & Möller	13.9	29
79	*Navicula pupula* Kütz.	14.0	26
80	*Cyclotella comensis* Grun.	14.0	12
81	*Stephanodiscus hantzschii* Grun.	14.1	19
82	*Cocconeis diminuta* Pant.	14.2	26
83	*Achnanthes lanceolata* var. *dubia* Grun.	14.3	19
84	*Navicula pseudoventralis* Hust.	14.5	19
85	*Cocconeis placentula* var. *lineata* (Ehrenb.) Van Heurck	14.6	13
86	*Asterionella formosa* Hasall	14.6	37
87	*Stephanodiscus hantzschii* fo. *tenuis* Håkansson & Stoermer	14.7	9
88	*Cocconeis placentula* var. *euglypta* (Ehrenb.) Grun.	14.8	10
89	*Fragilaria construens* var. *binodis* (Ehrenb.) Grun.	14.8	14
90	*Achnanthes lanceolata* (Bréb. ex Kütz.) Grun.	15.0	26
91	*Aulacoseira distans* (Ehrenb.) Simonsen	15.1	14
92	*Cymbella sinuata* Gregory	15.3	8
93	*Stephanodiscus* cf. *vestibulis* Håkansson, Theriot, & Stoermer	15.5	3
94	*Fragilaria capucina* var. *mesolepta* (Rabenh.) Rabenh.	15.7	10
95	*Achnanthes exigua* var. *heterovalva* Krasske	15.8	27
96	*Fragilaria brevistriata* var. *elliptica* Hérib.	16.3	17
97	*Achnanthes lanceolata* var. *elliptica* Cleve	16.5	17
98	*Stephanodiscus* cf. *pseudoexcentricus* Håkansson & Stoermer	16.5	3
99	*Aulacoseira italica* ssp. *subarctica* (O. Müll.) Simonsen	16.7	21
100	*Navicula vitabunda* Hust.	17.1	25
101	*Stephanodiscus parvus* Stoermer & Håkansson	17.1	20
102	*Caloneis bacillum* (Grun.) Cleve	17.1	4
103	*Navicula ventralis* Krasske	17.4	6
104	*Fragilaria vaucheriae* (Kütz.) J. B. Petersen	17.4	11
105	*Navicula cryptotenella* Lange-Bertalot	17.5	16
106	*Aulacoseira undulata* (Ehrenb.) Kütz.	17.6	5
107	*Fragilaria crotonensis* Kitton	17.6	31
108	*Tabellaria flocculosa* str. IIIp sensu Koppen	17.9	7

Number	Diatom taxon	TP optimum (μg/L)	Number of occurrences
109	*Nitzschia palea* (Kütz.) W. Smith	17.9	6
110	*Cocconeis placentula* Ehrenb.	18.3	11
111	*Neidium ampliatum* (Ehrenb.) Krammer	18.8	4
112	*Pinnularia braunii* (Grun.) Cleve	18.8	3
113	*Navicula absoluta* Hust.	18.9	16
114	*Gomphonema parvulum* (Kütz.) Kütz.	19.2	6
115	*Stephanodiscus niagarae* Ehrenb.	19.3	20
116	*Cyclotella ocellata* Pant.	19.5	11
117	*Achnanthes delicatula* (Kütz.) Grun.	19.5	6
118	*Achnanthes petersenii* Hust.	19.8	3
119	*Navicula kuelbsii* Lange-Bertalot	19.9	11
120	*Fragilaria nanana* Lange-Bertalot	20.2	13
121	*Fragilaria construens* (Ehrenb.) Grun.	20.5	40
122	*Gomphonema angustatum* (Kütz.) Rabenh.	20.6	10
123	*Stauroneis anceps* Ehrenb.	20.7	9
124	*Cyclotella bodanica* var. *lemanica* (O. Müller ex Schröter)	21.0	7
125	*Nitzschia gracilis* Hant.	21.0	9
126	*Cymbella silesiaca* Bleisch	21.1	12
127	*Pinnularia viridis* (Nitzsch) Ehrenb.	21.2	7
128	*Pinnularia nodosa* Ehrenb.	21.4	3
129	*Navicula scutteloides* W. Smith ex Greg.	21.7	10
130	*Fragilaria pinnata* Ehrenb.	21.8	53
131	*Navicula seminuloides* Hust.	21.9	26
132	*Gyrosigma accuminatum* (Kütz.) Rabenh.	22.0	4
133	*Navicula farta* Hust.	22.1	3
134	*Cyclostephanos tholiformis* Stoermer, Håkansson, and Theriot	22.2	11
135	*Aulacoseira ambigua* (Grun.) Simonsen	22.6	27
136	*Navicula minima* Grun.	22.7	28
137	*Tabellaria flocculosa* str. III sensu Koppen	23.1	11
138	*Aulacoseira lirata* var. *lacustris* (Grun.) R. Ross	23.1	7
139	*Fragilaria exigua* Grun.	23.2	2
140	*Fragilaria capucina* Desm.	23.3	7
141	*Achnanthes* sp. 1 BC	24.2	9
142	*Aulacoseira italica* (Ehrenb.) Simonsen	24.7	3
143	*Aulacoseira perglabra* var. *florinae* Camburn	26.0	7
144	*Cyclotella michiganiana* Skvortzow	26.3	22
145	*Aulacoseira granulata* var. *angustissima* (O. Müll.) Simonsen	26.9	4
146	*Fragilaria pinnata* var. *acuminata* A. Mayer	27.4	11
147	*Aulacoseira distans* var. *nivalis* (W. Smith) Kirchner	27.5	5
148	*Aulacoseira lirata* (Ehrenb.) Ross	28.8	4
149	*Aulacoseira granulata* (Ehrenb.) Simonsen	36.0	2
150	*Synedra minuscula* Grun.	41.9	2
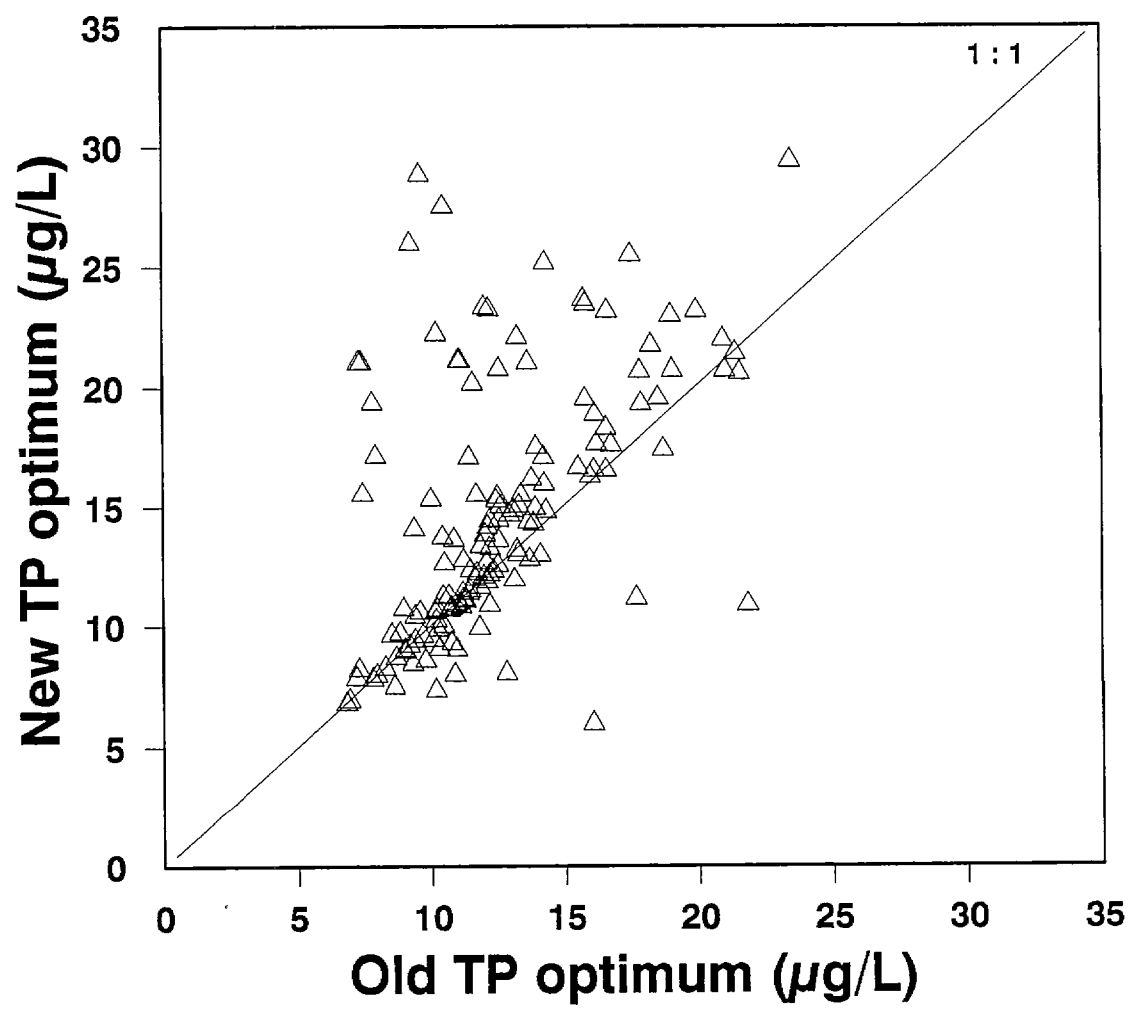
41.9) μg/L. Many species, such as *Cyclotella michiganiana*, *Navicula minima* and *Fragilaria pinnata*, now have strikingly higher estimated optima, whereas the [TP] optima of most other taxa increased only slightly or were unaffected. The optima of a few taxa, such as *Denticula kuetzingii* (*Denticula elegans* in Hall and Smol, 1992), decreased.

Our [TP] inference model was developed from lakes ranging between 5 and 85 μg/L, and can now be used to infer past lakewater [TP] within this range from fossil diatom assemblages. The model could be used to infer past [TP] in lakes that also fall within the range of values for the important environmental variables (conductivity 0-410 μS/cm; lake depth >4m; pH 6.2-8.5). Lakewater [TP] can be estimated using the classical deshrinking regression equation $y = 0.18x + 2.30$ where x and y represent respectively uncorrected and corrected [TP] inferences (Birks et al., 1990a, b). This equation was derived from the model illustrated in Figure 2.5d.

Applying the Model: Remarkable differences were noted between the original and updated model when inferring [TP] from the fossil assemblages of Wood Lake (Fig. 2.8). The reconstruction using the 37 lake model (Hall and Smol, 1992) described a mild eutrophication event which coincided with the relative increase in *Stephanodiscus minutulus*; an increase from 21 to 29 μg/L TP from the bottom of the core to the mid-1960s. From the 1960s to 1992, [TP] apparently dipped slightly but generally maintained a mildly eutrophic level.

The 59 lake model provided a [TP] profile more consistent with known historical changes at Wood Lake (Walker et al., 1994a). The approximately 85-year period of eutrophication described above is, according to the new model, an increase from 35 to 58 μg/L TP. This inference more closely parallels past measurements by Stockner and Northcote (1974). Water quality improvement within the last two decades (Walker et al., 1994a) has
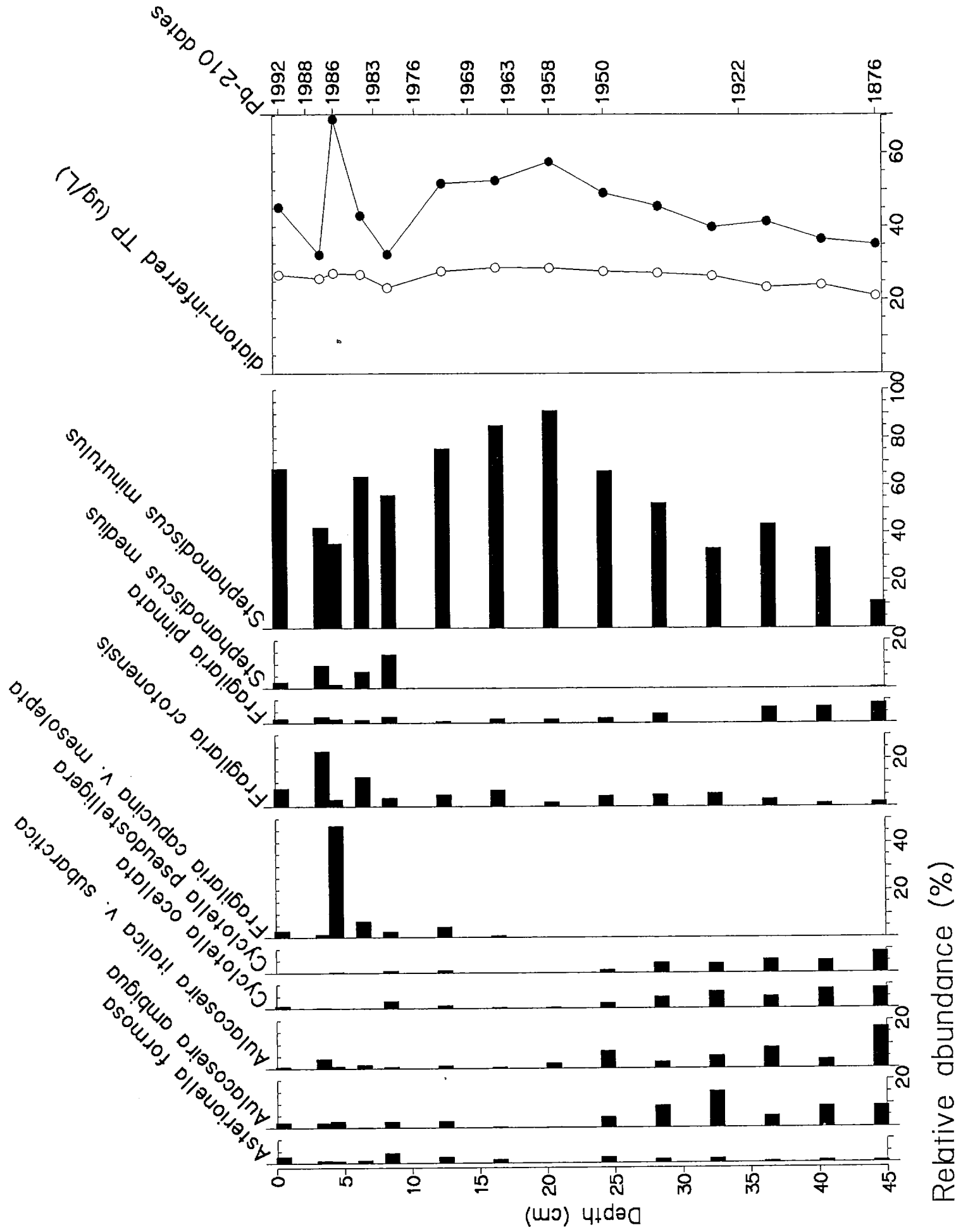
been clearly tracked using the new model, and the inferred TP concentration for the 0-1 cm interval (45.8 μg/L) is almost identical to the average measured [TP] for 1991 and 1992 (45.4 μg/L). The striking peak in inferred-[TP] ($\sim$ 1986) coincides with an unexplained bloom of *Fragilaria capucina* var. *mesolepta*. However, we do not rely on the accuracy of the inference in this interval because the assemblage provided a poor species analogue to the calibration set.

DISCUSSION

Data screening

It is unfortunate that lakes with [TP] greater than 85 μg/L needed to be removed from the data set, but it is not surprising to observe the confounding effect of high nutrient levels on diatom assemblages. Theoretically, diatoms possess an upper limit for any quantitative environmental variable, above which two general outcomes are possible: (1) all of the species perish, or (2) physiological mechanisms prevent toxicity and the remaining species sustain a monotonic distribution. Presumably the assemblages present at the highest [TP] are the most tolerant of eutrophic conditions, and therefore species composition will not respond markedly to further increases in phosphorus concentration. For example, a diatom assemblage from the surface sediments of a lake with 100 μg/L TP will likely not have an inferred [TP] higher than that for a lake with a similar diatom assemblage and 80 μg/L TP. This limitation was observed by Anderson et al. (1993), who also noted that diatom assemblages became uniform at a certain TP concentration. This effect can be attributed to certain diatom taxa, such as small *Stephanodiscus*, which compete well in eutrophic lakes because they are favoured over other diatom taxa at low Si/P ratios (Bradbury, 1988). Furthermore, and perhaps most

importantly, these species bloom in early spring, thereby avoiding competition with green and blue-green algae which generally bloom during the summer (Bradbury, 1988). Hence sedimentary diatom assemblages are usually dominated by small *Stephanodiscus* in lakes with > 85 μg/L TP. A similar, well-defined inference plateau was shown by Walker et al. (1994b) when developing a calibration set using chironomid remains to infer salinity. In their study, a specific taxon was noted to consistently dominate chironomid assemblages at high salinities.

Four of the high [TP] lakes (McKay (54), Somenos (53), Dragon (50) and Top (52) lakes) were dominated by small benthic *Fragilaria* species, primarily *F. pinnata* and *F. brevistriata*. Charlie (47), Little Bobtail (38) and Bullocks (48) lakes had diatom assemblages represented almost solely by small *Stephanodiscus* species. It is this low diversity that likely resulted in an underestimation of [TP] because inferences were based almost solely on the [TP] optima of highly represented taxa. The highest [TP] optimum before data screening was only 62.6 μg/L (for *Navicula vitabunda*), therefore it would have been unlikely even with weighted averaging to accurately infer [TP] levels greater than the selected maximum of 85 μg/L. Even after we removed four high [TP] lakes, the phosphorus gradient was more than doubled by adding lakes to the original data set.

As predicted by Hall and Smol (1992), the addition of more lakes with TP between 28 and 85 μg/L prevented lakes such as Holden (26) from being eliminated as outliers; Holden Lake's TP concentration of 45 μg/L was previously considered extreme under Hall and Smol's criteria. Remaining samples with a poor fit to the TP axis in a constrained CCA were not removed as before because the information contained in the diatom-environmental relationships was still considered essential to reconstructions, and any potential discrepancies (e.g. faulty [TP] measurements or taxonomic inconsistencies) were likely diminished by the larger data set.

Canonical Correspondence Analysis

In our data set, conductivity exerted the strongest influence on diatom distributions. The strong influence of conductivity and salinity on diatoms has been known for a long time, and recent studies in BC have documented robust relationships between these variables and diatom distributions (Cumming and Smol, 1993b; Wilson et al., 1994). However, the relationship between diatoms and [TP] is also sufficiently strong, thus warranting the development of WA regression and calibration models to infer [TP] from diatom assemblages. The addition of relatively high [TP] lakes to Hall and Smol's (1992) lake set did not improve the strength of the relationship between [TP] and diatoms, but it reduced the previous influence of calcium, depth and surface area.

We believe that other environmental variables not available in this data set (such as silica and turbulence) likely also had significant effects on diatom distributions and could improve the ordination. Hall and Smol (1992) and Anderson et al. (1993) discuss such explanations for unaccounted variance in diatom data when inferring phosphorus.

Total phosphorus accounted for a statistically significant direction of variance in the diatom data, as determined from a Monte Carlo permutation test in a CCA constrained to TP. This, in conjunction with a high eigenvalue ratio, makes the relationship between diatoms and [TP] sufficiently strong to warrant reconstructions of trophic status using our data set. Although our conductivity gradient is relatively small compared to other studies (e.g. Cumming and Smol, 1993b; Wilson et al., 1994), one could also infer this variable from the diatom assemblages of lakes with compatible present-day specific conductivity measurements.

Reconstruction

An environmental variable such as pH exerts a direct effect upon diatoms, often

resulting in a strong relationship between diatom distributions and that variable (e.g. Dixit et al., 1991). Total phosphorus is generally not as influential because it often depends upon other variables, such as silica availability and the rate of phosphorus uptake. However, even in the case of [TP], other scientists have established strong correlations. For example, Fritz et al. (1993) calculated the ratio of λ_1/λ_2 for [TP] to be 1.57, a value considerably larger than our eigenvalue ratio ($\lambda_1/\lambda_2=0.42$). The diatom assemblages from these Michigan lakes are undoubtedly highly regulated by trophic variables, as demonstrated by their relationship to total nitrogen, Secchi depth and chlorophyll a (as described in a CCA). On the other hand, British Columbia lakes exhibit vast extremes of environmental conditions (mainly because we sampled a wider range of biogeoclimatic zones), and the [TP] signal has probably been "blurred" by the cumulative influence of these conditions.

The availability of environmental data is a problem when compiling any training set (Anderson et al., 1993; Christie and Smol, 1993). Ideally, the chemical information for each lake represents an average of several measurements, taken over a few years prior to the sampling of surficial sediment. In this way there is little error in assuming that the diatom communities represented in the sediments are a product of the measured environmental conditions. Often it is not logistically possible to obtain a rigorous assessment of environmental variability for all sites, primarily because of climate, access, and resource constraints. These sources of error were inevitable in this calibration; however, by considering numerous sites, inaccuracies have been reduced.

The WA calibration calculations were performed using classical regression (Fig. 2.5; inverse regression data not shown), because this resulted in the lowest RMSE and the least noticeable trend in the residuals from the observed versus inferred [TP] analyses. Hall and Smol (1992) used similar methods. WA calibration without species transformation resulted in

a decrease in the strength of the relationship with an increase in the lake set (Figs. 2.5a, b), whereas transformation resulted in a similar, but less important effect (Figs. 2.5c, d). The addition of lakes apparently increased the spread of points around the 1:1 line, thereby decreasing the WA regression coefficients and increasing RMSE values. However, the bootstrapped relationship was stronger in the augmented lake set, likely because the increased sample size and enlarged [TP] gradient (Figs. 2.5e, f) permitted more accurate estimates of the true [TP] optima of the diatom taxa; the new lakes also provided additional data for the random selection algorithm used during bootstrapping. Although our new model (Fig. 2.5d) had a lower regression coefficient (Figs. 2.5c, d) and a larger error (Figs. 2.5e, f) than the old model, we believe that it can provide more realistic inferences of TP. We make this assertion because the [TP] gradient has been greatly enlarged, and the diatom optima have likely been more realistically defined (discussed below).

The new WA model can be used to infer past lakewater [TP] from diatom assemblages preserved in the sediments of British Columbia lakes, ideally within the range of 5-85 μg/L. Our [TP] reconstructions for Wood Lake clearly illustrate the limitations of the original 37-lake training set. Although a slight eutrophication trend was inferred by the old model, it appeared unresponsive to the severe changes which occurred as a result of cultural nutrient loading to Wood Lake (Stockner and Northcote, 1974). Furthermore, the new model clearly indicates the improvement in water quality caused by a decrease in the retention time of the lake. A distillery, which was constructed in 1971, pumps large volumes of cooling water from nearby Okanagan Lake into a creek upstream from W̶o̶o̶d̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ce then the water replacement time has decreased from 30 to 14 ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶he new model also recognizes the F. capucina var. mesolepta ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶· nutrient peak. It is possible that a sudden nutrient-rich input o ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶ ̶tely,

historical measurements are unavailable for confirmation. The new model also satisfactorily inferred the recent TP concentration in Wood Lake (actual TP 45.4 μg/L, inferred TP 45.8 μg/L; Reavie and Smol, 1994; see Chapter 3), whereas the old model inferred a much lower value (27.1 μg/L).

By expanding the dataset we have developed a more useful tool for inferring nutrient conditions in BC lakes. Most importantly, the range of [TP] optima has been increased, making the model more useful for inferring eutrophic conditions. Also, weighted average optima for nineteen more taxa have been added. The possibility of encountering a lake with incompatible nutrient conditions, or fossil diatom assemblages with poor analogues to the calibration set, has become less likely.

The [TP] range of our model is comparable with that of others (Table 2.6). Agbeti (1992), Christie and Smol (1993), Fritz et al. (1993) and Dixit and Smol (1994) employ a lake [TP] range similar to ours. A clear difference in lake nutrient burdens occurs between North America and the British Isles. Models from Northern Ireland (Anderson et al., 1993) and England (Bennion, 1994) reach TP concentrations almost ten-fold of our maximum [TP]. Moreover, the British Isles calibration sets are devoid of oligotrophic lakes (<25 μg/L), while the North American models sufficiently fill this portion of the [TP] gradient. The North American models generally inferred [TP] more accurately within the oligo-mesotrophic [TP] spectrum, whereas the British models were more useful inferring extremely eutrophic conditions. The high productivity of British lakes is probably attributable to a very shallow average depth and a greater degree of land use. This comparison between available trophic inference models illustrates that regional differences in limnological characteristics occur. Furthermore, species optima change relative to the size of the [TP] gradient.

Benefits of the calibration approach are made clear by applications, such as our Wood

Table 2.6: Summary of the available [TP] inference models and their geographical locations and [TP] ranges. Whitmore's (1989) model used a broad trophic classification scheme, so TP concentrations were not available.

Author(s)	Study location	Main trophic variable(s)	Number of lakes	Lake TP range (μg/L) lower	upper
Whitmore (1989)	Florida	"trophic state index"	30	--	--
Agbeti (1992)	Southern Ontario	TP, Secchi	28	2	63
Hall and Smol (1992)	British Columbia	TP	37	5	28
Anderson et al. (1993)	Northern Ireland	TP	43	25	800
Christie and Smol (1993)	Southeastern Ontario	TP, total nitrogen	36	6	75.6
Fritz et al. (1993)	Michigan	TP	41	1	51
Bennion (1994)	Southeastern England	TP	30	25	646
Dixit and Smol (1994)	Northeastern U.S.A.	TP, Cl, pH, Secchi	64	0.8	154.5
Our model	British Columbia	TP	59	5	85

Lake example. What begins as an array of complex diatom stratigraphic data, can be transformed into information that is easy to understand and communicate, and is useful for lake management. Quantitative reconstructions can define the timing and extent of cultural disturbance, and identify pre-disturbance conditions. Reconstructions can also reveal geographic areas where lakes tend to be naturally eutrophic, and hence where remediation is not likely to improve water quality.

CHAPTER 3

POST-SETTLEMENT EUTROPHICATION HISTORIES OF SIX BRITISH

COLUMBIA (CANADA) LAKES

INTRODUCTION

British Columbia (BC) contains many lakes that are believed to have become

eutrophic as a result of human activities since European settlement, circa A.D. 1850

(Northcote and Larkin, 1963; Stockner and Northcote, 1974). Possible nutrient sources have

included direct inputs (such as sewage disposal) and indirect sources (such as runoff from

pastures and agricultural fertilizers). Deterioration of water quality, together with increased

agricultural and industrial water demands, have jeopardized aspects of the BC economy which

are based on tourism and recreational activities. Lake restoration measures are therefore

desirable, but it is first necessary to determine the causes and timing of lakewater quality

deterioration. Unfortunately, as is typically the case with water quality assessments, most

lake monitoring programs in BC did not begin until a problem had been identified. Using

paleolimnology, we can identify past trophic changes that occurred before conventional

monitoring programs were initiated (Charles and Smol, 1994).

Paleolimnology is the synthetic science that is used to describe the physical, chemical,

and biological histories of aquatic systems. This is achieved by analyzing materials archived

in the sedimentary record. Paleolimnological analyses aimed at reconstructing past

eutrophication trends have been conducted in many regions (Anderson et al., 1993; Fritz et

al., 1993; Salonen et al., 1993; Brugam, 1988; Stoermer et al., 1985b), including a few in

BC (e.g. Stockner and Northcote, 1974). Sedimented diatoms are sensitive indicators of

nutrient concentrations and can be used to infer former water quality. The remains of diatoms have been considered the most useful paleoindicators for several reasons: their siliceous cell walls make them resistant to dissolution and breakage, they occur in most environments where water is present, they are very diverse and often identifiable to a level of species or subspecies, and their taxonomy is well described (Dixit et al., 1992b). Stockner and Northcote (1974) used diatoms in a pioneering study to reconstruct the trophic histories of four BC lakes, but their methods were based only on the environmental preferences of a few common diatom genera. Paleolimnologists now use more advanced inference methods to infer past lakewater quality.

It is possible to determine quantitatively the relationship of a taxon to a given environmental variable, and to use such relationships to infer that variable from fossil diatom assemblages (Charles and Smol, 1994). These relationships are obtained by assessing the diatom assemblages from the surficial sediments of several lakes, and relating the species distributions to modern limnological conditions. Several diatom-based transfer functions have been developed to infer environmental characteristics from diatom assemblage data. For example, pH (Birks et al., 1990a), salinity (Cumming and Smol, 1993b) and nutrient (e.g. Agbeti, 1992; Anderson et al., 1993; Hall and Smol, 1992) inference functions have been constructed using weighted-averaging regression and calibration approaches. This method assumes that diatom species respond in a unimodal fashion to environmental variables, and thus an environmental optimum can be determined for each taxon. A quantitative estimate for a fossil sample is then determined from the average of the optima of all taxa in that sample, weighted by their relative abundances.

The main objective of this paper was to infer past trends in TP concentration for six presently eutrophic BC lakes. To gain a historical perspective on eutrophication in BC, past

changes in total phosphorus (TP) were inferred from diatom valves in lake sediment cores from the region. We used a [TP] inference model that was originally designed by Hall and Smol (1992) and has been recently updated (Reavie et al., 1994; see Chapter 2) to increase the range of [TP] optima represented by the diatom species. We also evaluate the reconstructive ability of this new model. The inferred data identify the timing and rate of nutrient fluctuations, and pre-disturbance conditions for each lake. In some cases our findings identified naturally eutrophic systems.

SITE DESCRIPTIONS

The lakes selected for this study are scattered over a wide area of inland BC (Fig. 3.1), and were all suspected of being eutrophic due to anthropogenic activities. Three of the lakes (Charlie, Tabor, and Norman) were of interest to the BC Ministry of the Environment (MOE) because they have highly developed watersheds. Wood Lake, in the Okanagan Basin, has been notably affected by human influence over the last few decades, and has been the subject of much research (Walker et al., 1994a; Stockner and Northcote, 1974). Pinantan and Dutch lakes are presently eutrophic, and they were arbitrarily selected for study from a list of "problem" lakes (BC Ministry of Environment, unpublished data). Limnological characteristics of these lakes are presented in Table 3.1 and morphometric maps are in Figures 3.2 to 3.7. All of the lakes would be considered eutrophic by general standards (Wetzel, 1983). However, as explained below, they differ greatly in many respects.

The Okanagan Valley

The Okanagan basin is a U-shaped valley where over 100 headwater lakes support a

.

BRITISH COLUMBIA

Charlie △

Norman △ △ Tabor

Dutch △

Pinantan △

Wood △

CANADA
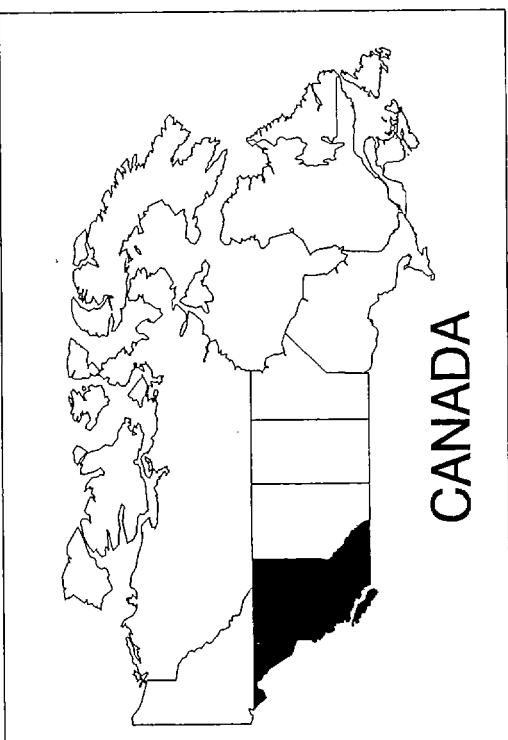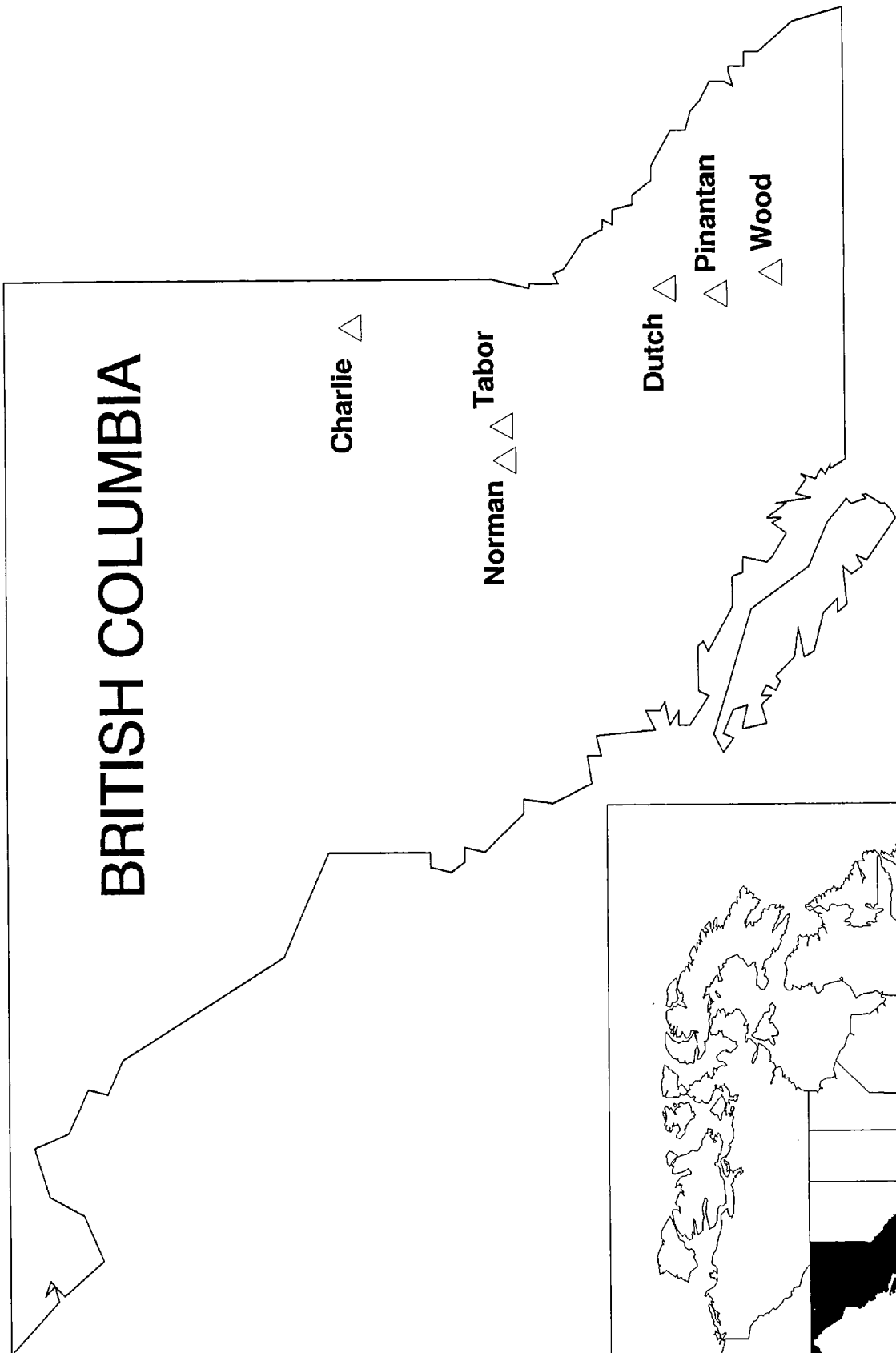
Table 3.1: Environmental characteristics of the study lakes. The most important variables, according to Reavie et al. (1994), are presented. Chlorophyll *a* data were unavailable for Pinantan Lake.

	Physical						Chemical			
Lake	Location (N°,W°)	Surface area (ha)	Max. depth (m)	Mean depth (m)	Watershed area (km²)	Altitude (m)	TP (µg/L)	Chl. *a* (µg/L)	pH	Conductivity (µS/cm)
Wood	50°10',119°38'	930	34	21.5	190	390	45.4	4.1	8.3	325
Pinantan	50°75',120°00'	73	19	10	10	853	54	--	7.8	420
Dutch	51°65',120°02'	65.5	39	14.4	3.9	765	46	3.0	8.1	235
Charlie	56°40',121°00'	3120	12	5	298	691	103	47.3	8.1	169
Tabor	53°55',122°33'	417	9	4.6	44	702	70	7.6	7.8	145
Norman	53°47',123°22'	510	24	12	129.4	876	23.7	4.5	7.8	137

5
10
15
20
25
30

N

0   500   1000   1500

METRES

DEPTHS ARE IN METRES

DEPTHS ARE IN METRES

METRES

100   0   100   200   300

36 7.5 15 22.5 30 38 *

N

KILOMETRES

DEPTHS ARE IN METRES

N

1000

500

METRES

DEPTHS ARE IN METRES

0

21

15

N

DEPTHS ARE IN METRES

1000   0   1000   2000   3000   4000

METRES

N

3

6

9

I 0 1 2 3 4

KILOMETRES

DEPTHS ARE IN METRES
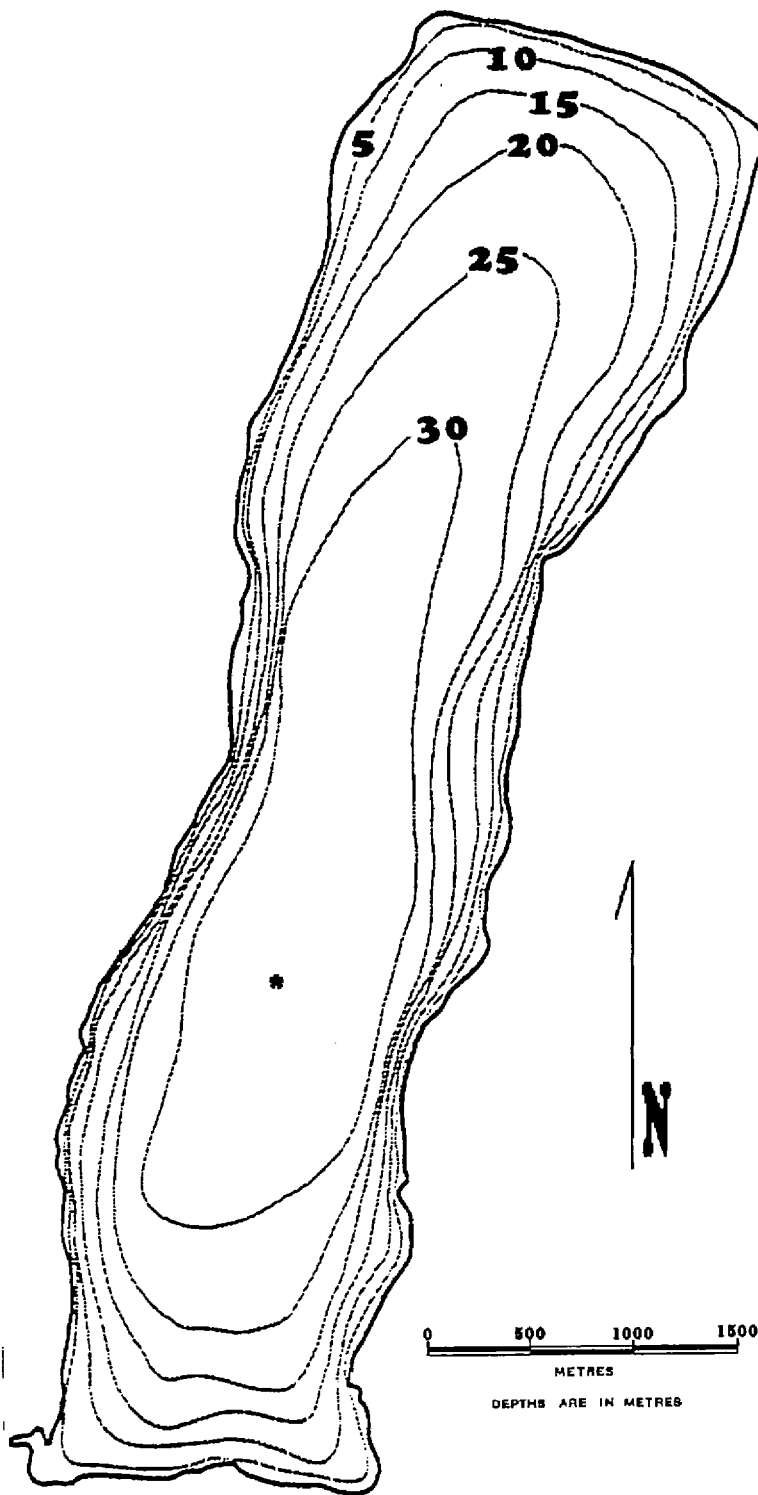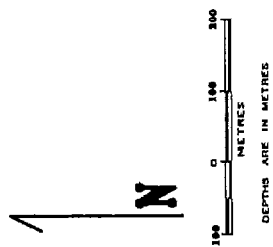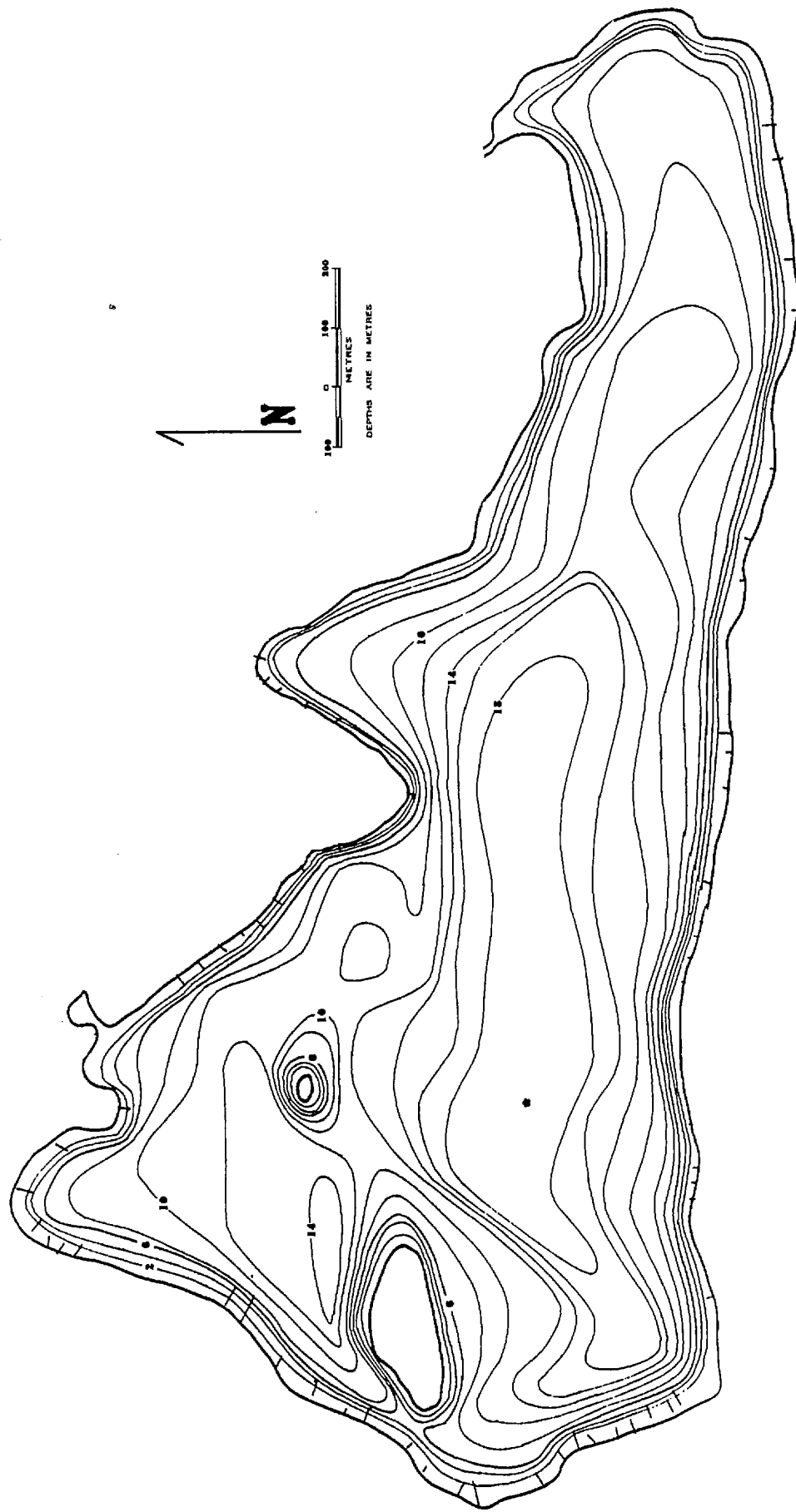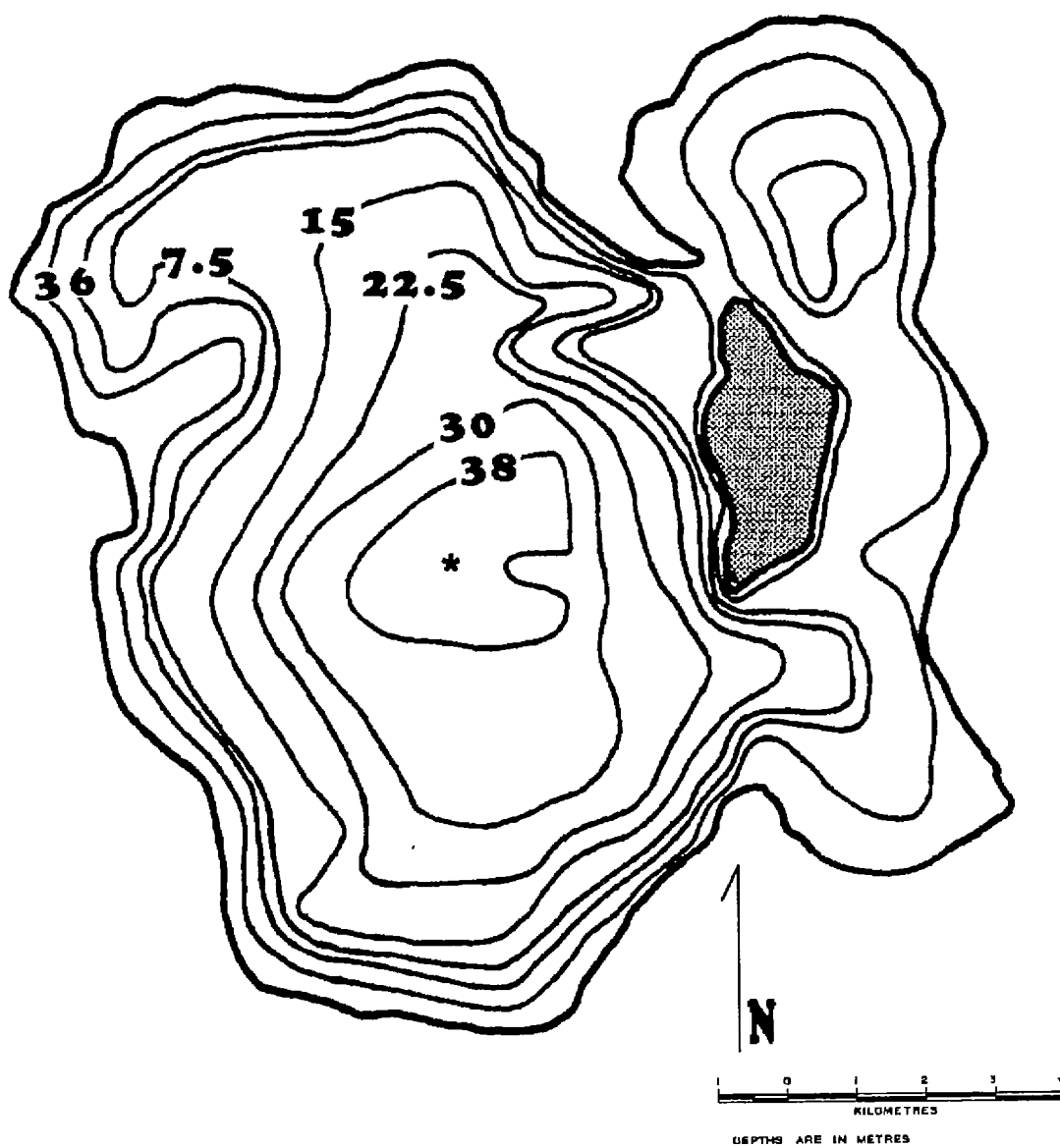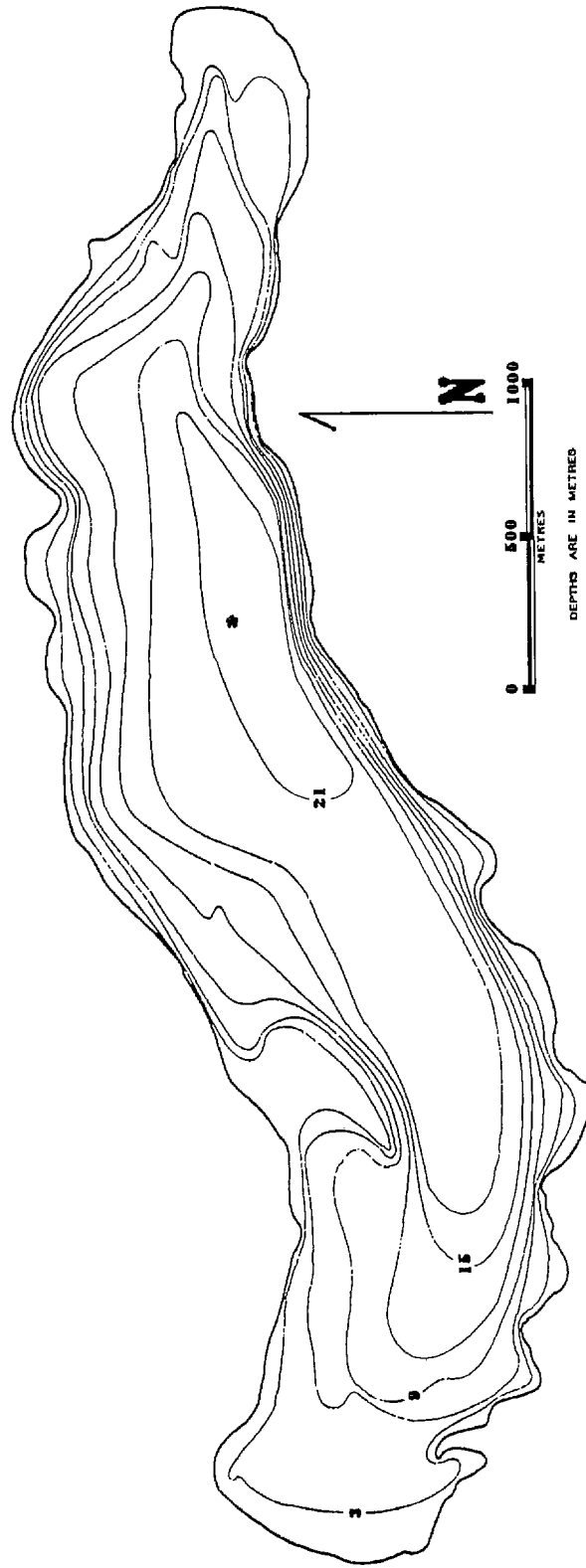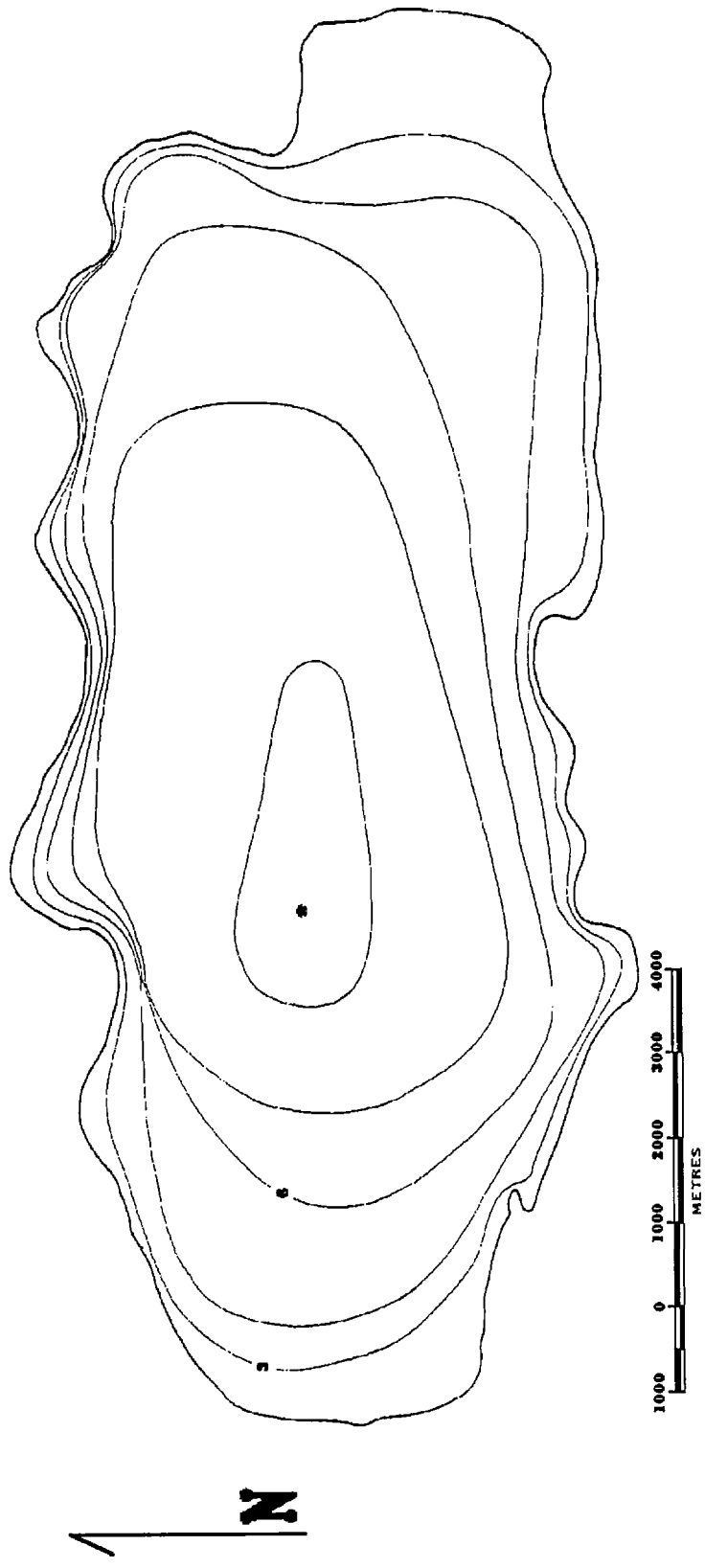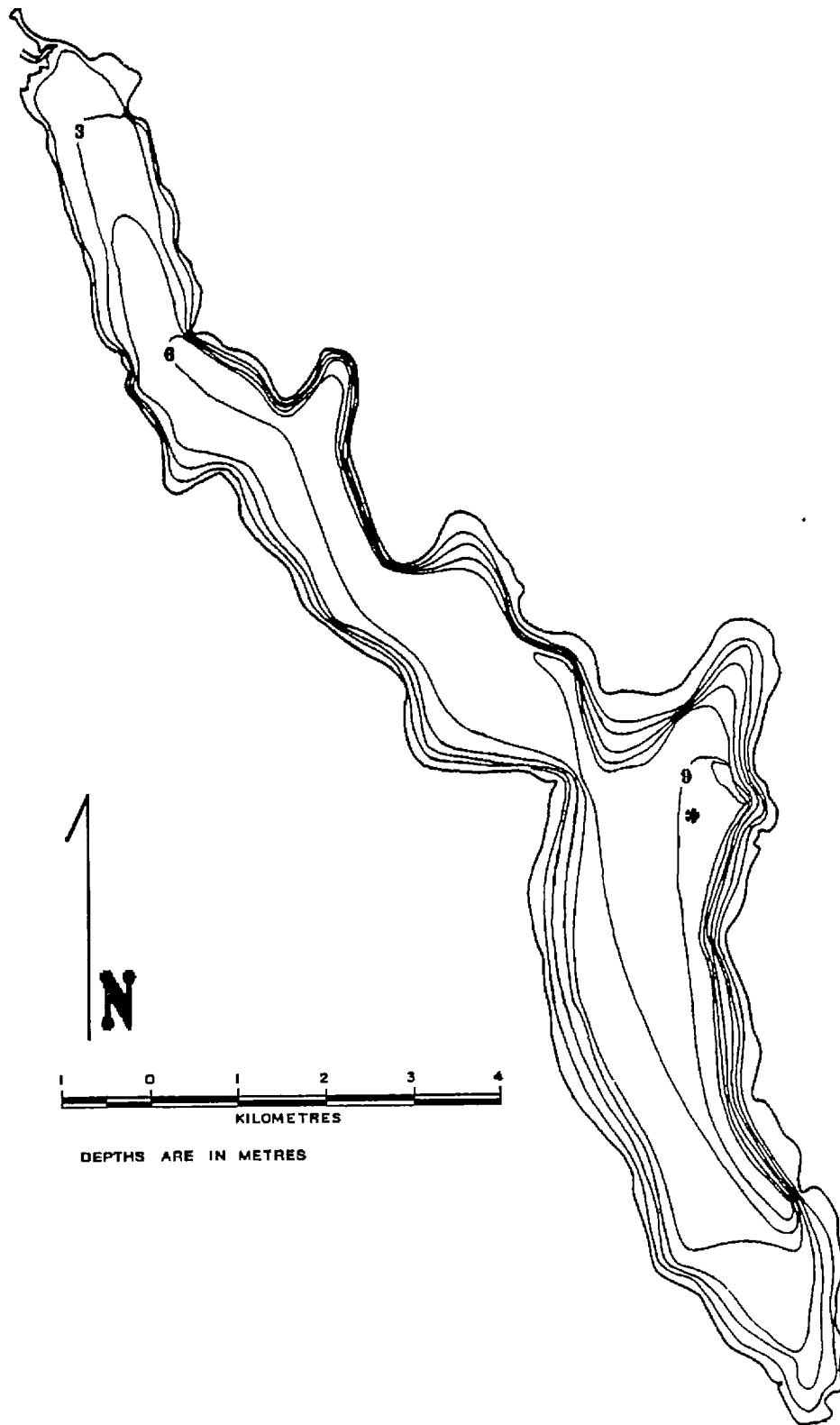
valuable sports fishing industry. The lowest lands, which contain the largest lakes, support

considerable cottage and tourist industries as well as several dairy farms. The middle

"benchlands" contain fertile soil with fruit orchards. The uppermost open forest lands are

predominantly used for cattle grazing.

.Wood Lake (Fig. 3.2) is somewhat unique for this region because the history of

European settlement in the watershed is well known for the past century. The catchment has

been highly disrupted. Natural vegetation has been replaced by orchards and pasture land,

and the southern shore is now surrounded with summer homes. The retention time of the *(year round)*

lake may have been reduced by the installation of a pumping station in 1971 (Gray and

Jasper, 1982). Wood Lake receives nutrients primarily from leaking septic tanks and Vernon

Creek, which drains agricultural lands (Walker et al., 1994a). The annual phosphorus load is

considered to be relatively low, although the small volume and shallow depth of the lake

(Table 3.1) make it particularly susceptible to the effects of eutrophication. Stein and

Coulthard (1971) speculated that Wood Lake was naturally eutrophic, but this has not been

firmly established because long-term limnological measurements are not available. One recent

study shows that specific eutrophication events in Wood Lake's catchment are well associated

with trends in the sedimentary diatom and chironomid fossil remains (Walker et al., 1994a).

We present the diatom data again because quantitative [TP] reconstruction has not yet been

performed for Wood Lake.

Kamloops Area

Dutch Lake (Fig. 3.3) is located north of Clearwater in a popular tourist region.

Information on the trophic status of Dutch Lake was first collected in the early 1970s when

Lovell and Sackey (1973) attributed the poor flavour of fish to the excessive growth of blue-

Lovell and Sackey (1973) is a paper concerned with off-flavour of fish flesh due to blue-green algae. They never sampled Dutch Lake!

green algae. Shortly thereafter the Thompson - Nicola Regional District and the Clearwater

Improvement District requested that an evaluation of the lake's water quality be undertaken

(Nordin, 1982). The filling time of the lake was estimated at 20 years; since closure of the

only outflow by road construction, most of the lake's water loss is by evaporation, hence the

term "filling time". Because virtually no nutrient material can be removed from the lake

basin, the lake is very susceptible to nutrient inputs. Hypolimnetic oxygen deficits, high

chlorophyll a and nutrient concentrations, and dense algal blooms have been noted.

Numerous septic tanks appear to be the most probable source of nutrients (Nordin, 1982). In

the last decade, considerable development has occurred north of the lake. No long term

limnological data, however, are available for this period, except for some very recent

measurements.

Less information is available concerning Pinantan Lake (Fig. 3.4), except that it is

presently a eutrophic system (R. Nordin, pers. comm.). Impacts from development and

tourism are similar to those affecting Dutch Lake. However, Pinantan has a relatively small

drainage basin, and presumably has been less affected by anthropogenic nutrient inputs.

Prince George Region

The economic activity surrounding Prince George consists largely of forest-related

industries, and several major beef and dairy enterprises (Collins, 1985). Two lakes near

Prince George were chosen for detailed paleolimnological analyses.

Norman Lake (Fig. 3.5) is slightly eutrophic (Table 3.1) and receives a higher than

"permissible" annual phosphorus load of 1213 kg/yr (Carmichael, 1992; Dillon and Rigler,

1975). The north shore is developed with summer homes (about 140 cabins, many with

outhouses) and many of the local residents use outmoded methods of sewage disposal.

Carmichael (1992), after assessing the general limnology of Norman Lake, recommended land use strategies to decrease anthropogenic nutrient inputs. He also suspected, however, that much of the phosphorus load to the lake came from precipitation and natural land sources. Unfortunately, no monitoring of trophic variables (e.g. TP, chlorophyll a, Secchi depth) was performed before 1988.

Tabor Lake (Fig. 3.6), located approximately 11 km east of Prince George along the Yellowhead Highway, is highly eutrophic (Table 3.1). Water clarity is better than that in Norman Lake (N. B. Carmicheal, pers. comm.), although Tabor has higher [TP] and chl. a concentrations (Table 3.1). Tabor Lake also experiences severe mid-summer algal blooms. Hypolimnetic oxygen depletions in recent years have resulted in fish kills, so it is believed that severe eutrophication is only a recent phenomenon, but long-term data are not available. Nutrient inputs are suspected to originate from 166 domestic residences, pulp mills, and sawmill burners established on the north and west shores. Also, in 1961, a significant fire occured in the lake's watershed, and in the early 1970s, the diversion of an inflow stream resulted in a longer water replacement time (N. B. Carmicheal, pers. comm.).

Fort St. John Area

Charlie Lake (Fig. 3.7) is one of the most eutrophic freshwater lakes in British Columbia. It is one of seven aquatic systems in the Peace River area that has been selected for water quality assessment and development of rehabilitation objectives (Nordin and Pommen, 1985), primarily because the lake provides the city of Fort St. John (population 20,000) with fisheries, recreation, industrial and domestic water resources. There is concern that agricultural inputs and the disposal of sewage directly into groundwater from domestic residences has been providing the lake with nutrient-rich effluent. Water quality assessments

of Charlie Lake have shown that trophic variables are at highly undesirable levels (Table 3.1), and dense blue-green algal (*Aphanizomenon*) blooms have occurred in recent summers (N. B. Carmicheal, pers. comm.). Unfortunately, historical records on the lake's trophic status are not available.

MATERIALS AND METHODS

Coring Procedures

Charlie, Norman and Tabor lakes were cored in late summer of 1991 by the BC Ministry of the Environment personnel. Wood Lake was sampled in the summer of 1992 (Walker et al., 1994a). Pinantan and Dutch lakes were cored in the spring of 1993.

The sediment cores, ranging from 27 to 45 cm in length, were collected using a modified Kajak-Brinkhurst (KB) gravity corer equipped with a 6.35 cm inside diameter core tube (Glew, 1989). Sites were located in or near the deepest part of each lake (Figs. 3.2 to 3.7). Cores were immediately sectioned on shore using a close-interval extruder (Glew, 1988) and 1 cm slices were stored in Whirlpak® bags prior to subsampling. In the laboratory, subsamples of sediment were taken for ^{210}Pb dating and diatom preparation.

^{210}Pb Dating

Sediment subsamples (approx. 30 g) of selected intervals were weighed, oven-dried (24 hrs at 110 °C) and ground to a fine dust. Samples were reweighed to determine dry weight and submitted to Atomic Energy of Canada Limited (Chalk River, Ontario) for ^{210}Pb analysis and dating.

^{210}Pb dating is actually calculated from determinations of ^{210}Po, a decay product of

^{210}Pb. Quantitative measurements were made using alpha spectroscopy (Cornett et al., 1984). Unsupported ^{210}Pb was calculated by subtracting supported ^{210}Pb (the baseline ^{210}Pb activity naturally present in the sediments) from total activity at each level. Dates were then determined from unsupported isotopes using the constant rate of supply (C.R.S.) model (Appleby and Oldfield, 1978). ^{210}Pb dating is limited to ~150 years before present, so extrapolations beyond this period were made based on calculated sediment accumulation rates for the lowest intervals of the core. Cores from Pinantan and Dutch lakes were not dated. Varves were counted to determine the chronology of Pinantan Lake, and Dutch Lake will be ^{210}Pb dated in the near future.

Diatom Preparation

Twelve evenly-spaced intervals were selected from each core for diatom analysis. Extra intervals were added later for some cores, for which better stratigraphic resolution was desired. Subsamples of wet sediment (0.3 - 1.0 g) were heated for one hour in a mixture of potassium dichromate and sulphuric acid to digest organic matter. Samples were then repeatedly washed in distilled water and centrifuged until they were clear and free of residual acid. The siliceous remains were settled onto coverslips, and the coverslips were mounted on glass slides using Hyrax®. For each slide, at least 500 diatom valves were identified and counted along transects under oil immersion (1000X) using an Olympus BH-2 microscope. Diatom taxonomy was based primarily on Krammer and Lange-Bertalot (1986, 1988, 1991a, b), Camburn et al. (1984-1986), and Patrick and Reimer (1966). Most of the taxa used in this study have been compiled in Cumming et al. (1994). Chrysophyte cysts were also enumerated and expressed relative to the diatom sum (Smol, 1985).

Inferring Total Lakewater Phosphorus

The transfer function we used was constructed by determining the relationship between water chemistry variables and diatom distributions in the surface sediments of 59 BC lakes (Reavie et al., 1994; see Chapter 2). Canonical correspondence analysis (CCA) was used to identify environmental variables with strong correlations to modern diatom assemblages (ter Braak, 1990a, b). Lakewater [TP] had a significant influence on the diatom communities in our lakes, so weighted averaging (WA) calibration was used to estimate the [TP] optima for common diatom taxa (Line et al., 1994). The final regression equations were based on the [TP] optima of the 150 most common diatom taxa found in the 59 calibration lakes (Reavie et al., 1994).

Quantitative [TP] inferences were performed using the computer program WACALIB version 3.3 (Line et al., 1994). The distribution of [TP] measurements for the calibration set was skewed toward the oligotrophic extreme of the spectrum, so calculations were performed using transformed (ln(TP + 1)) data (see Birks et al. (1990a) for details). The inferred [TP] values furnished by WACALIB were subsequently untransformed.

Two versions of the inference equation were used in this study. For Norman, Pinantan, Dutch and Wood lakes, the 59-lake model described by Reavie et al. (1994; see Chapter 2) was applied. However, a larger training set (63 lakes) was employed to infer [TP] for Charlie and Tabor lakes (hereafter termed the "high TP" lakes). Four hypereutrophic lakes were originally removed from the training set because inferred values provided by the transfer function underestimated actual [TP] in very eutrophic systems. These four "outlier" lakes were readmitted to the data set because the larger model provided additional modern analogues to assemblages in the cores from the high [TP] lakes, and because recent [TP] for Charlie and Tabor lakes were within the range of the expanded, but not the restricted, model.

For reasons described in Reavie et al. (1994), the inference error increased after including the four hypereutrophic lakes in the training set; we are confident, however, that the full data set is more appropriate for inferring the trophic status of these extremely eutrophic systems.

Fit to [TP] and Modern Analogue Measures

A canonical correspondence analysis (CCA) constrained to [TP] was used to evaluate our [TP] reconstructions. The residual distance of diatom assemblages to the TP axis provided a measure to assess "lack-of-fit" to TP. First, CCA determined what were extreme residual distances from the TP axis based on modern samples with environmental data (Reavie et al., 1994). Core samples were then run passively in CCA and positioned, by means of transition formulae (ter Braak, 1990a), with respect to the TP axis. Fossil samples with residual distances greater than the 90% and 95% confidence limit of the calibration set were considered to have "poor" and "very poor" fits to TP, respectively.

Fossil samples with close modern analogues are more likely to provide reliable inferred [TP] values (Birks et al., 1990b). Using the computer program ANALOG (Line and Birks, unpublished program), we performed analogue matching to identify fossil diatom assemblages with poor analogues to modern assemblages in the training set. Details of this method are described by Hall and Smol (1993).

RESULTS AND DISCUSSION

Okanagan Valley

The pre-19th century sediments of the Wood Lake core were dominated by a complex of planktonic diatoms, primarily *Aulacoseira ambigua*, *Au. italica* var. *subarctica*, *Cyclotella*

ocellata, *C. pseudostelligera* (Krammer and Lange-Bertalot, 1991a) and *Stephanodiscus*

minutulus (Fig. 3.8). Total phosphorus inferred from this assemblage indicated that the lake

already had relatively high nutrient levels ([TP] = 35 μg/L) before cultural impacts occurred

in the drainage basin. *Stephanodiscus minutulus*, a eutrophic taxon (Håkansson, 1976),

increased considerably since the late-19th century, until approximately A.D. 1958, when it

comprised 93% of the diatom assemblage. Chrysophyte stomatocysts gradually decreased

relative to the diatoms; cyst:diatom ratios declined from 0.14 in the older sediments (~A.D.

1876) to trace amounts by approximately A.D. 1958. Also, during this same period, inferred

[TP] increased from 35 to 58 μg/L.

The eutrophication indicated by these trends is corroborated by other studies. In

1974, a study of the Okanagan Basin suggested that Wood Lake had rapidly begun to

deteriorate in the early 1900s, due to agricultural development and diversion of inflow streams

for irrigation (Anonymous, 1974). Other analyses from our sediment core (Walker et al.,

1994a) also noted a disappearance of the profundal chironomids ca. 1940. Such an ·

occurrence is characteristic of hypolimnetic anoxia resulting from eutrophication.

Diatom-inferred [TP] and the relative abundance of *S. minutulus* decreased after A.D.

1958, indicating that water quality improved. The opening of a Hiram Walker distillery ca.

A.D. 1970 is considered to be the most important event to have affected Wood Lake in recent

years. Since 1971, the distillery has been pumping large volumes of water from nearby

Okanagan Lake for cooling, and releasing it into Vernon Creek, upstream from Wood Lake

(Anonymous, 1974). This pumping resulted in a reduction of the theoretical water

replacement time from 30 to 14 years. Although this estimate varies due to precipitation

(Gray and Jasper, 1982), it is believed that the distillery's release of Okanagan Lake water is

largely responsible for the water quality improvement (Walker et al., 1994a).

Pb-210 dates

-1992
-1988
-1986
-1983
-1976
-1969
-1963
-1958
-1950
-1922
-1876

diatom-inferred TP (g/L)

cyst / diatom ratio (*100)

Stephanodiscus minutulus

Stephanodiscus medius

Fragilaria pinnata

Fragilaria crotonensis

Fragilaria capucina v. mesolepta

Fragilaria pseudostelligera

Cyclotella ocellata

Cyclotella italica v. subarctica

Aulacoseira ambigua

Aulacoseira formosa

Asterionella formosa

Depth (cm)

Relative abundance (%)

Little Round Lake, in southeastern Ontario, experienced similar changes in its recent history (Smol et al., 1983; Christie and Smol, 1993). As anthropogenic nutrient enrichment occurred in Little Round Lake, the relative abundance of small *Stephanodiscus* spp. increased. Subsequently, most of the disturbance in the drainage basin ceased, and seepage from road salt enhanced meromixis, thus preventing the resuspension of hypolimnetic nutrients. An improvement in surface water quality occurred and eutrophic taxa decreased in relative abundance. Water quality recoveries as dramatic as those in Wood and Little Round Lake have rarely been observed.

The lower diatom-inferred [TP] values during the past 25 years in Wood Lake were interrupted at the 4-5 cm interval with a sudden increase to 70 μg/L, which corresponds with a sudden increase in *Fragilaria capucina* var. *mesolepta*. The brief (two or three year) bloom of *F. capucina* var. *mesolepta* may have been caused by an irregularly high nutrient load in Wood Lake at that time. However, a misleading [TP] inference may have occurred at this interval because of poor assemblage fit to [TP] (i.e. a poor analogue). Unfortunately, no historical information is available to explain this incident. Overall, we conclude that Wood Lake was fairly productive before the commencement of human impacts, European development caused a degradation in water quality, and recent unintentional mitigation (water pumping) has resulted in a reduction of TP concentrations.

Kamloops Area

The Pinantan Lake core was varved, and based on the sediment laminations, the core dates to well before A.D. 1850 (Fig. 3.9). Diatom data suggest that Pinantan is naturally eutrophic, and that this has changed little since European settlement. *Stephanodiscus parvus*, a well-known eutrophic indicator (Anderson et al., 1990, Christie and Smol, 1993), occurred
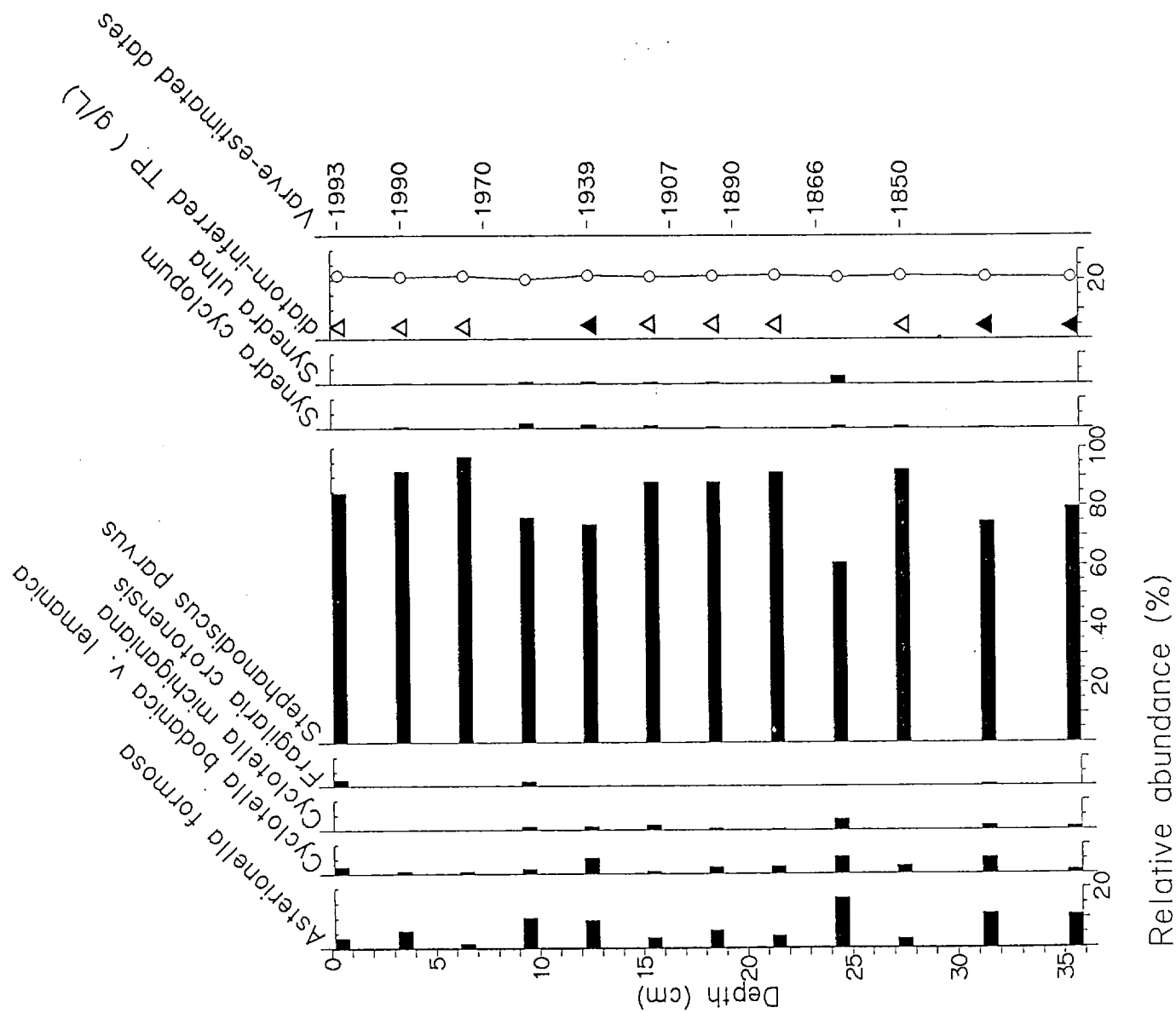
in relative abundances of 60 to 96% throughout the core. In general, the assemblages did not

shift markedly, except for a minor peak of *Asterionella formosa* during the 1850s. Similarly,

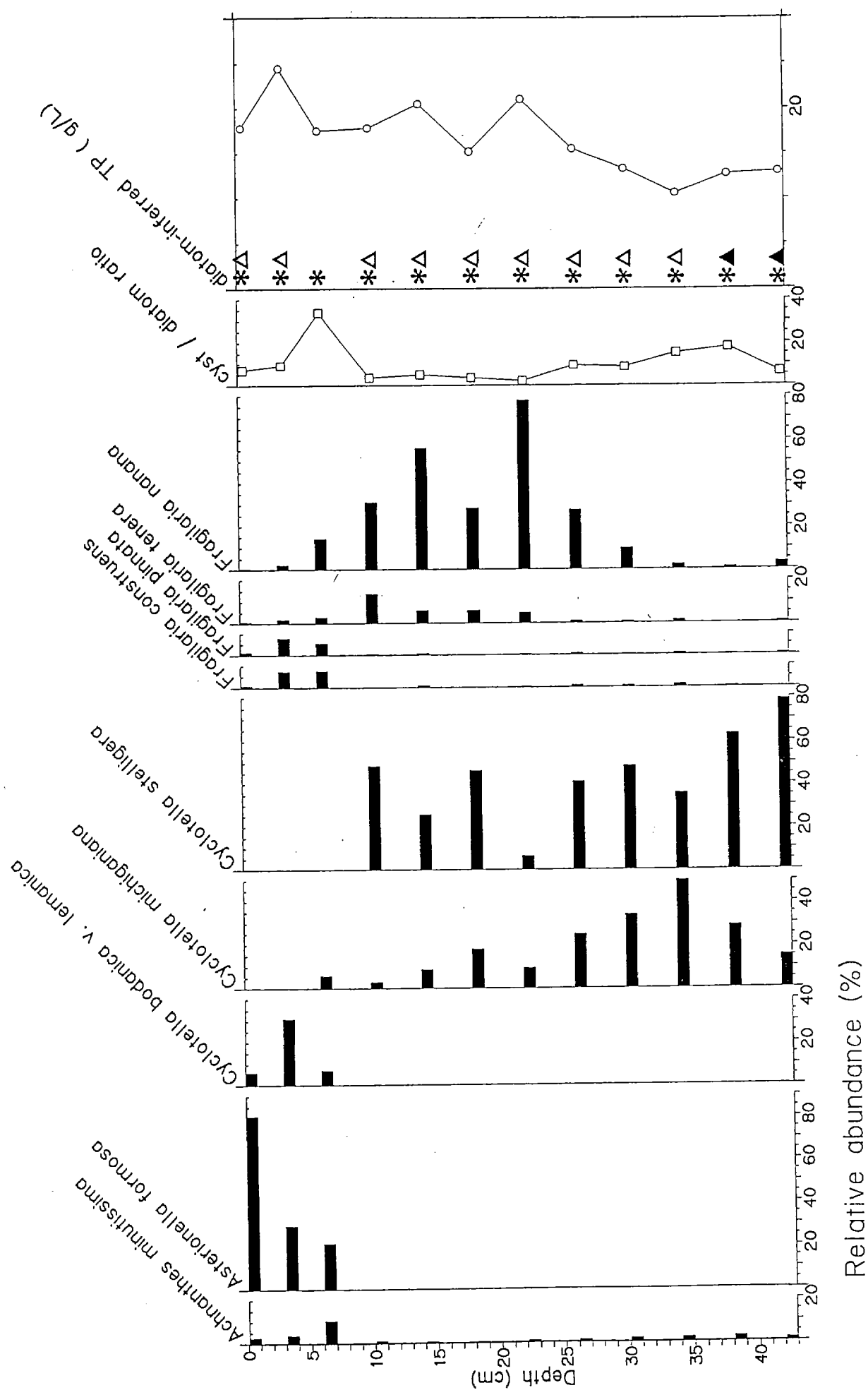the inferred [TP] values were almost uniform (at about 21 μg/L) throughout the core.

However, few analogous assemblages, (with such a high relative abundance of *S. parvus*)

occurred in our calibration set, and therefore many of the intervals show poor fit to TP, so we

suspect that our inference model underestimated TP concentrations. In summary, the diatom

data indicate that there has been little anthropogenic impact on water quality in Pinantan Lake.

The Dutch Lake core recorded five major diatom community shifts (Fig. 3.10).

Cyclotella stelligera occurred in high relative abundance at the bottom of the core.

Cyclotella michiganiana gradually increased in abundance until the 34 cm depth, after which

it gradually tapered off to trace amounts. *Fragilaria nanana* sharply increased from 10 to

78% between the 30 and 22 cm intervals. From 22 to 10 cm, *F. nanana* and *C. stelligera*

were dominant, but above 10 cm both species declined in the sediments. *Asterionella formosa*

and *C. bodanica* var. *lemanica* increased in the most recent sediments, with *As. formosa*

dominating the 0-1 cm assemblage (80% relative abundance). The cyst to diatom ratio

gradually decreased from 0.17 (at the 38-39 cm interval) to about 0.04 (between 23 and 10

cm), but at the 6-7 cm interval, the ratio increased sharply to 0.34, then decreased again to

low levels. Dutch Lake's microfossil history closely mirrors that of other culturally impacted

lakes (O'Sullivan, 1992; Brugam and Vallarino, 1989; Brugam, 1988; Stoermer et al.,

1985b). For example, Brugam and Vallarino (1989) documented similar diatom shifts for

Meridian Lake, Washington: a pre-settlement *Cyclotella* community, followed by *Synedra*

(during deforestation), and a later increase of *Asterionella formosa*. Their observations

confirmed a strong relationship between the onset of human disturbance and an increase in *As.*

formosa. They also discovered a poor relationship between lakewater TP concentration and

Relative abundance (%)

Achnanthes minutissima
Asterionella formosa
Cyclotella bodanica v. lemanica
Cyclotella michiganiana
Cyclotella stelligera
Fragilaria construens
Fragilaria pinnata
Fragilaria tenera
Fragilaria nanana
cyst / diatom ratio
diatom-inferred TP (g/L)

Depth (cm)

patterns in the diatom assemblages. Conversely, Stoermer et al. (1985b) observed that substantial phosphorus loading to Lake Ontario resulted in similar community shifts to those in Dutch Lake (such as an increasing abundance of *As. formosa*), so this matter is still disputable.

Inferred [TP] from the Dutch Lake core fluctuated with the diatom species changes, but an approximate increase from 13 to 20 μg/L is apparent. Even though a variety of assemblages are represented in the Dutch Lake core, many of the assemblages are poorly fit to our [TP] transfer function, and analogues to the calibration set were poor. Dutch Lake has likely been highly affected by human influence, but perhaps other limnological variables (e.g. DOC, total nitrogen) are controlling diatom communities, and hence the diatom-inferred [TP] measurements are somewhat insensitive to changes in the algal communities. But, as discussed in the previous paragraph, the species changes we observed in Dutch Lake probably indicate eutrophication. The correlation of historical events to eutrophication will be attempted when the Dutch Lake core has been dated with ^{210}Pb (in progress).

Prince George Region

Diatom assemblages from the lakes in the vicinity of Prince George indicate naturally eutrophic conditions, but the changes that have occurred since settlement differ between the two lakes (Figs 3.11 and 3.12).

The common taxa in Norman Lake, *Au. italica* var. *subarctica*, *F. pinnata* and *S. minutulus*, maintain relatively constant distributions throughout most of the core (Fig. 3.11). Drastic community changes have not occurred in the ~175 years represented by the core, and the inferred [TP] values indicate that Norman Lake is naturally eutrophic (~24 μg/L pre-disturbance conditions), and has changed little since European settlement. This was expected

Pb-210 dates

1991
1986
1981
1972
1961
1945
1926
1894
1863
1849

Stephanodiscus niagarae
diatom-inferred TP (g/L)

Navicula v. acuminatum
Stephanodiscus hantzschii

Fragilaria pinnata v. acuminatum

Fragilaria construens v. venter
Fragilaria construens
Fragilaria pinnata

Fragilaria brevistriata
Fragilaria ambigua
Aulacoseira

Depth (cm)

Relative abundance (%)
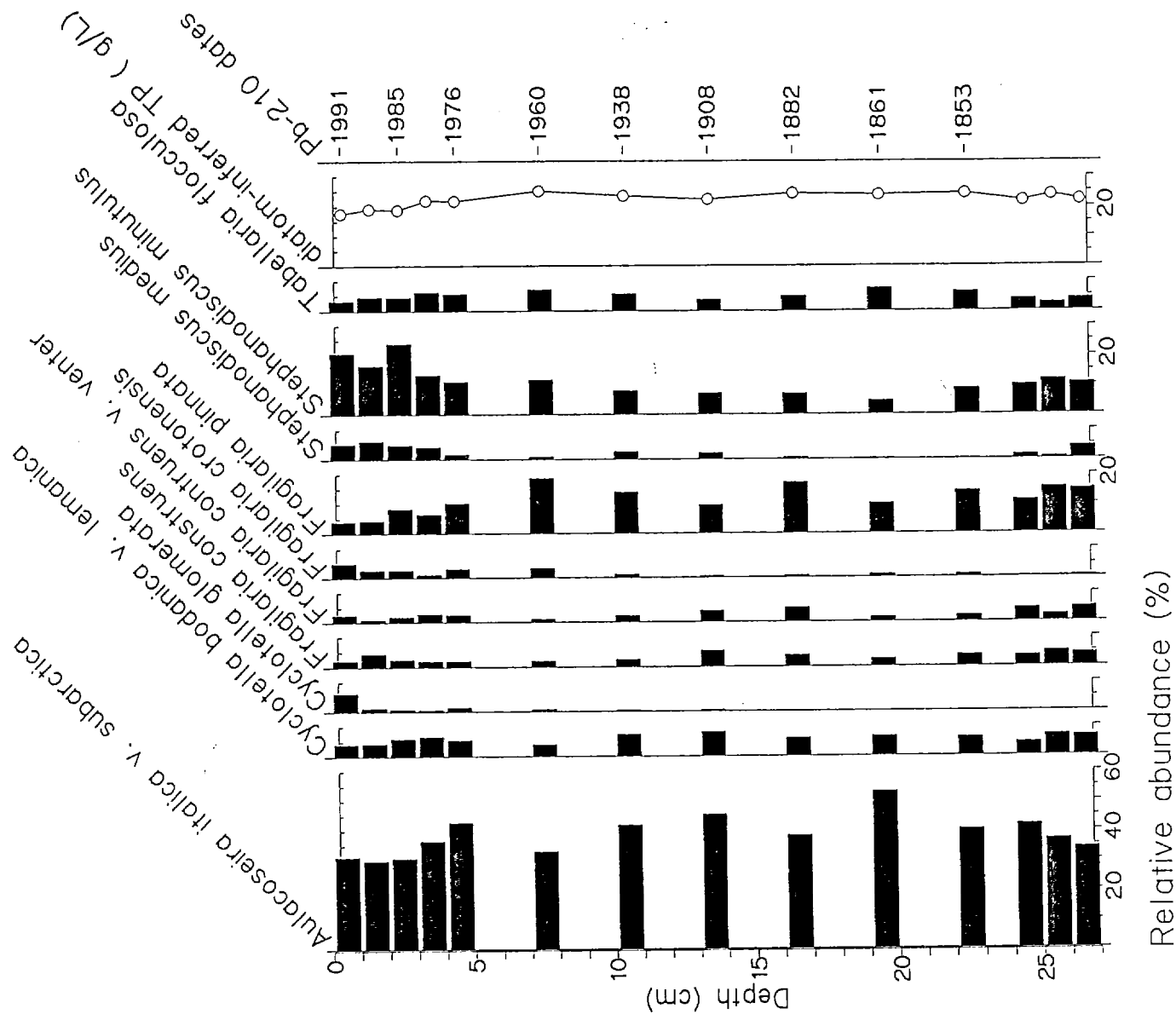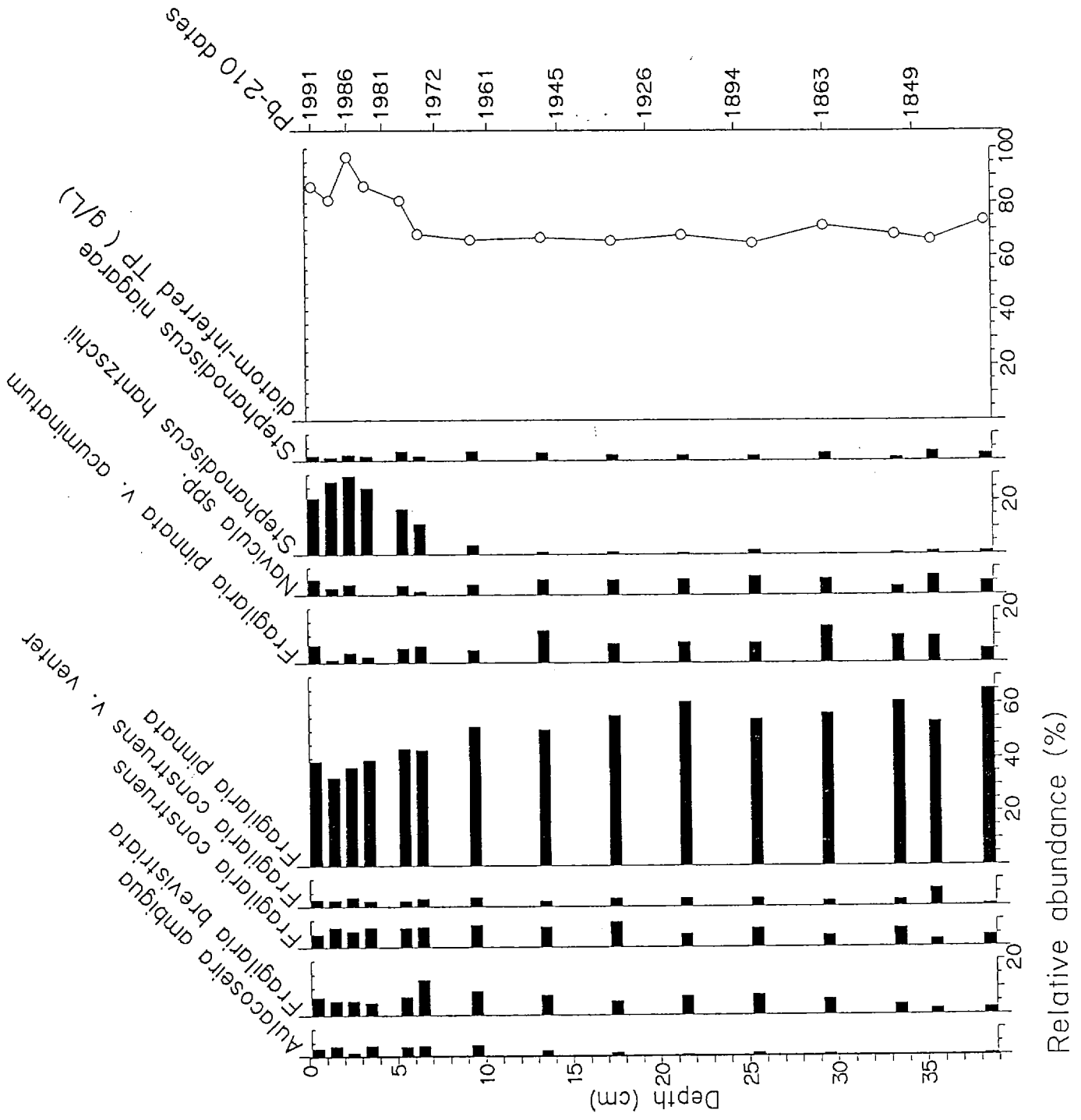
because before A.D. 1970, settlement in the lake's drainage basin was limited (Carmichael, 1992). *Stephanodiscus minutulus* slightly displaces other taxa after approximately A.D. 1976, suggesting a minor increase in nutrient inputs. Carmichael (1992) reports that the sewage disposal facilities (outhouses and rock pits) around the lake are no more than 25 years old, so nutrient inputs from these sources are a possible cause for this slight eutrophication. Our [TP] inference model appears to have been insensitive to the modest community shifts occurring within the last 15 years, and in fact, we infer slight water quality improvement. Although this contradicts our interpretation from the diatom stratigraphy, the shift is relatively minor.

In the Tabor Lake core (Fig. 3.12), *Fragilaria pinnata* is the most common taxon at all intervals, varying between 65% at the bottom of the core to 33% at the 1-2 cm interval. Many other *Fragilaria* taxa are also present, but no obvious shifts are noted. Since the mid-1970s, the eutrophic indicator *S. hantzschii* increased in abundance to peak in ~A.D. 1985. This stratigraphy shows a similar trend in diatom shifts to Norman Lake, with a modest increase in small *Stephanodiscus* species to as high as 30% within the last two decades, indicating minor eutrophication. The consistent dominance of the littoral, generally ubiquitous diatom *F. pinnata* (Patrick and Reimer, 1966) makes this stratigraphy difficult to interpret. *Fragilaria pinnata* is difficult to classify ecologically because it occurs in both high and low nutrient environments (Christie and Smol, 1993; and personal observations). Tabor Lake's relative shallowness (Table 3.1) allows for a largely benthic assemblage to dominate.

The inferred phosphorus history of Tabor Lake illustrates a relatively stable TP concentration (~66 μg/L) until the 1970s, after which some modest eutrophication occurred. Another interesting trend is the apparent nutrient decrease in the uppermost two intervals, indicating that the lake may be recovering.
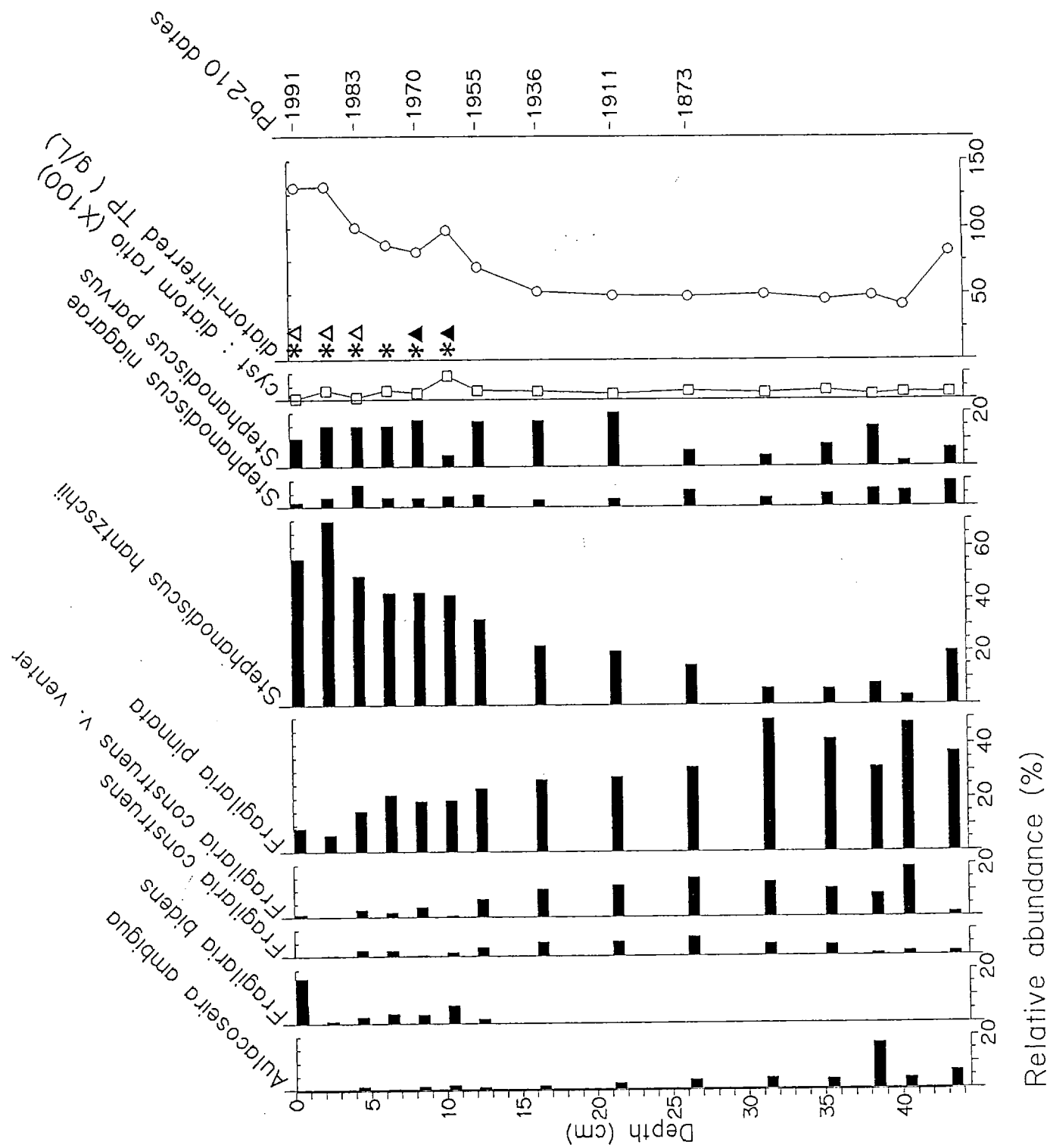
A subsequent analysis of uncleaned sediment from the Tabor Lake core revealed numerous charcoal particles in the 11 cm to 9 cm depths, coinciding with the severe watershed fire in 1961. It was suspected that nutrient inputs to the lake increased following the fire-induced vegetation clearance (N. B. Carmicheal, pers. comm.). Additionally, around 1970, a major inflow stream to Tabor Lake was diverted because of extensive flooding and crop destruction (N. B. Carmicheal, pers. comm.). The resulting decrease in Tabor's flushing rate has likely allowed the lake to maintain a higher nutrient load. Both of these disturbances could be responsible for the recent eutrophication of the lake; however, without analysis of more intervals it is impossible to tease apart these effects.

Fort St. John Area

Aulacoseira ambigua, *F. pinnata* and three *Stephanodiscus* species are the most common taxa in the pre-A.D. 1900 sediments of Charlie Lake (Fig. 3.13), indicating that it was a naturally eutrophic system. The eutrophic indicator *S. hantzschii* increased strikingly between A.D. 1873 and A.D. 1991. *Stephanodiscus parvus* also increased slightly during this time, whereas *F. pinnata* decreased from a relative abundance of 48% to 7%. The impact following settlement was apparently severe; several papers document such dramatic increases in *Stephanodiscus* with eutrophication (e.g. Osborne and Moss, 1977; Anderson et al., 1990). *Fragilaria bidens* (Krammer and Lange-Bertalot, 1991a) is not in our calibration set (Reavie et al., 1994; see Chapter 2), although it appears in the surficial sediments of Charlie Lake. Its autecology is unknown, and because it does not occur in our [TP] transfer function, our inferred [TP] was not influenced by this taxon. As expected, the most recent intervals provided poor analogues to the calibration set and contained assemblages poorly fit to TP, likely due to the absence of *F. bidens* in the calibration set, and the very high relative

Pb-210 dates

—1991
—1983
—1970
—1955
—1936
—1911
—1873

diatom-inferred TP (g/L)
ratio (x100)
cyst : diatom

Cyclotella niagarae
Stephanodiscus parvus
Stephanodiscus hantzschii
Fragilaria construens v. venter
Fragilaria construens v. venter
Fragilaria pinnata
Fragilaria bidens
Aulacoseira ambigua

Depth (cm)

Relative abundance (%)

abundances of *Stephanodiscus* in the top 11 cm of the Charlie Lake core. The naturally high productivity of Charlie Lake probably prevented chrysophytes from occurring in high abundances, as stomatocysts were found only at trace levels throughout the core.

Inferred [TP] measurements indicated an approximate 90 μg/L increase in [TP] (from 40 to 130 μg/L) since the end of the 19th century. Aquatic productivity in Charlie Lake appears to have increased during the period of deforestation, agricultural development and sewage disposal in the region.

Evaluation of the Weighted-Averaging [TP] Inference Model

Because this is the first application of our new [TP] inference model (Reavie et al., 1994) for hindcasting long-term changes in lakewater TP concentration, it is important to assess the reliability and performance of the model. Table 3.2 compares present-day lakewater phosphorus concentrations to [TP] values inferred from the diatoms in the surface sediments (0-1 cm depth) from each core. In some cases, TP concentrations inferred from surface sediment diatom assemblages closely matched available water chemistry measurements (Wood, Charlie, Norman), indicating that our model may be capable of providing accurate quantitative estimates of lakewater TP concentration. However, the most recent [TP] inferences from Pinantan and Dutch lakes underestimated measured values. It was inevitable that the accuracy of our [TP] inferences would be affected by other variables. The absence of analogues and lack-of-fit to [TP] of many fossil diatom assemblages is attributable to the effect of other factors, such as conductivity (Cumming et al., 1994), which exert control over the diatom communities. This could explain the underestimated [TP] reconstructions for Pinantan and Dutch lakes. Inaccurate quantitative reconstructions are not a fatal problem, however. The addition of eutrophic lakes to the original calibration set (Hall and Smol, 1992)

Table 3.2: A comparison of recent [TP] measurements and inferred [TP] in the uppermost sediment intervals of the study lakes.

Lake	actual TP (μg/L)	diatom-inferred TP (μg/L)
Wood	45.4	46
Pinantan	54	22
Dutch	46	18
Charlie	103	128
Tabor	70	86
Norman	23.7	18

allowed us, by providing more analogues, to more realistically infer [TP] in Wood Lake

(Reavie et al., 1994; see Chapter 2). Presumably the addition of more lakes would further

improve our transfer function.

CHAPTER 4

GENERAL DISCUSSION AND CONCLUSIONS

This study provides the first application from western Canada of a diatom-based weighted averaging inference model for hindcasting long-term changes in lakewater TP concentration. Because this work was largely exploratory, and such models are still pioneering in many respects, it is important to assess the reliability and performance of the inference model. Inference reliability is perhaps best assessed by independent cross-validation techniques, such as bootstrapping error estimation (Birks et al., 1990a, b). Because computer programs for such analyses were not available when the original calibration data were published (Hall and Smol, 1992), I provide a synthesis of the old and new calibration sets and their predictive ability with regards to [TP]. It appears, according to RMSE values (Fig. 2.5), that the original data set would provide more reliable inferences of [TP]. However, Hall and Smol's (1992) model is useful only to infer [TP] in oligotrophic, mesotrophic, and mildly eutrophic BC lakes. The new model can infer within Hall and Smol's range, as well as in highly eutrophic systems, so the new model is clearly a more useful tool in inferring [TP] in lakes with serious eutrophication problems.

Further evidence that the new model can provide more realistic [TP] reconstructions is provided by the comparison of known and inferred data (Table 3.2). Although no proof is available to show that the new model more accurately reconstructed the [TP] history of Wood Lake (Fig. 2.8), historical records suggest that nutrient concentrations were much higher than those predicted by the old model (Stockner and Northcote, 1974; Walker et al., 1994a). Furthermore, the new model more accurately inferred present-day [TP].

Problems with inferring [TP] arose when fossil samples provided poor species

analogues to the calibration set, resulting in the underestimation of TP concentrations (Table 3.2). In particular, the model performed most poorly for Dutch Lake (Fig. 3.4) because of a lack of analagous assemblages in the sediment core. With further expansion of the calibration set (particularly in the high lakewater [TP] range), a greater variety of eutrophic diatom assemblages could be added to the data set, thus providing a more robust inference model.

The trends in relative abundance of siliceous microfossils in BC lakes provide several insights on the influence of cultural activities on these lakes. For example, the predominant relationship between small *Stephanodiscus* taxa and high nutrient levels has been known for some time (Hustedt, 1930; Håkansson, 1976), and a considerable increase in *Stephanodiscus* with a concommitant decline in benthic taxa has been seen in several eutrophied lakes (Osborne and Moss, 1977; Engstrom et al., 1985). My results confirm that such a trend appears to coincide with cultural eutrophication in BC lakes. The diatom stratigraphy that occurred in Dutch Lake (i.e. the replacement of *Cyclotella* by *Synedra* and *Asterionella formosa*) is also a common indicator of anthropogenic nutrient loading (e.g. Brugam and Vallarino, 1989).

The nature of some of the events in the six study lakes is not entirely clear. This is because diatom community composition is controlled by multiple variables, making the diatom data for all of the lakes inherently "noisy". For instance, it is clear that some major event occurred around 1986 in Wood Lake, although we cannot point to a specific historical event to explain the *F. capucina* var. *mesolepta* peak. *Fragilaria pinnata* played a major role in the [TP] inference for Tabor Lake; however, it is quite probable that this taxon thrives not only because it is tolerant of high [TP] (Reavie et al., 1994; see Chapter 2), but because its growth is optimal in shallow water (personal observation).

Many interesting eutrophication trends were observed in the six BC lakes. As expected, recent increases in phosphorus levels were inferred in some of the lakes. However, others seem to have maintained their natural trophic state since pre-European settlement of inland BC. In all cases it appears that the lakes were quite productive prior to human settlement.

The underlying assumption of this study was that nutrient inputs increased as human populations grew in the watersheds of every study lake. The response to this loading varied among the six lakes included in this study. The lakes with the largest catchment areas (Wood and Charlie; Table 3.1) appear to have been the most severely affected. Hall and Smol (1993), whose studies involved a paleolimnological analysis of eutrophication following the Ontario hemlock decline ($\sim$4800 BP), also noticed that the degree of eutrophication was related to catchment size. Apparently, forest decimation by natural causes has a similar limnological effect as human exploitation (e.g. deforestation) of the landscape, as would be expected.

Other variables besides catchment size obviously affected the pattern in eutrophication trends that we observed. For instance, Dutch Lake has the smallest watershed area (3.9 km^2) and it eutrophied more notably than Norman Lake (watershed area = 129.4 km^2). The relatively unaffected lakes were probably subject to fewer cultural impacts than the highly degraded systems, so we propose that our quantitative reconstructions can be indirectly related to the amount of human exploitation within the catchment.

Interestingly, a lake such as Pinantan has had some development in its drainage basin, but showed little concommitant evidence of increased eutrophication. Similarly, Norman Lake is also naturally productive, and receives nutrients mainly from natural sources (as was suggested by Carmichael, 1992). We suggest that planned restoration measures for these

systems be postponed until better evidence for anthropogenic eutrophication is produced. The amelioration of present human inputs to such lakes may prevent future water quality problems; however, a significant reduction in [TP] concentrations is probably an unrealistic goal. Conversely, Charlie Lake is an example of a severely culturally impacted lake system, which might be a more plausible candidate for mitigation.

In order to develop a better statistical model for the paleolimnological inference of [TP] in British Columbia, future research could focus on a larger training set. This should provide the diatom species analogues needed to reconstruct past [TP] from a greater variety of fossil diatom assemblages. This study provides evidence for naturally and anthropogenically derived eutrophication in a small proportion of BC's "problem" lakes. Additional paleolimnological research is needed from other regions of BC in order to assess the extent and causes of eutrophication trends.

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