Limnological Aspects of Charlie Lake (Peace River Drainage, British Columbia): A Summary of Data Collected Between 1974 and 1995

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ABSTRACT

The Charlie Lake Conservation Society (CLCS) and its Technical Advisory Committee are in the process of developing a restoration plan for the Charlie Lake watershed (Fort St. John area, British Columbia). Of particular interest to the CLCS is the growing perception that tributary and lake water quality is deteriorating as a result of sediment and nutrient pollution. Sediment pollution to the tributaries appears to be generated in regions of the watershed where cattle are permitted to graze in and around riparian zones, and in regions where riparian vegetation has been disturbed or completely removed. Although the source(s) of nutrient pollution have not been clearly identified, it is generally believed that riparian damage and residential development have decreased the land's ability to buffer nutrient transfer to lake and tributary waters. Anecdotal information and paleolimnological data (sediment coring) suggest that recent increases in nutrient loading rates to the lake has increased the intensity of green and blue-green algal blooms.

Water quality (chemical and biological data) surveys of Charlie Lake and its major tributaries have been undertaken periodically since 1974. The primary objective of this document is to analyze and interpret the 1974 to 1996 water quality database so that the CLCS can develop its restoration plan in view of "what is already known" and "what has been accomplished to date". In addition to the data review, we also summarized the findings of earlier technical documentation, such as the fish parasite and paleolimnological studies conducted by Bangham and Adams (1954) and Reavie *et al.* (1995b), respectively, and the preliminary limnological investigation conducted by Nordin and Pommin (1985).

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1.0 INTRODUCTION

The Charlie Lake Conservation Society (CLCS) and its Technical Advisory Committee are in the processes of developing a restoration plan for the Charlie Lake watershed (Fort St. John area, British Columbia). Of particular interest to the CLCS is the growing perception that tributary and lake water quality is deteriorating as a result of sediment (erosional) and nutrient (N and P) pollution. Anecdotal information indicates that sediment pollution to the tributaries and lake comes from disturbed riparian areas where bare or loose soils have been exposed and eroded by water. The source(s) of nutrient pollution are unverified; however, it is generally believed that residential and agricultural land clearing, both in and around riparian habitats, has decreased the nutrient-buffering capacity of the land-water interface. Loads from fertilized lands, residential sewage and livestock arenas are also thought to be contributing factors. The dense algal blooms (up to 412 $\mu g \cdot L^{-1}$ chlorophyll *a*) observed in Charlie Lake during the summer and autumn months are believed to be triggered by excessive mid-summer nutrient loading events. While major fish kills have not been documented to date, there is a concern that fish populations and recreational opportunities may be jeopardized if summer and autumn algal production increases.

The purpose of this report is to summarize existing documentation on Charlie Lake and its watershed, and to analyze and interpret the vast limnological ("water quality") database (1974 to 1996) that exists for the watershed so that the CLCS can develop its restoration plan in view of "what is already known" and "what has been accomplished to date". More specifically, this document contains quantitative information on:

(1) licensed water withdrawals (water export volumes);
(2) agricultural pesticide residues in water, sediments and fish flesh;
(3) fish parasites (discussion based on very old data (1952));
(4) historical trends in algal production (paleolimnological investigations);
(5) lake morphometry and basic hydrology;
(6) phytoplankton community dynamics and correlates of algal abundance;
(7) nutrient loading and exports (internal versus external P loading); and
(8) effects of watershed development (in terms of P loading).

It is hoped that the technical information presented in this report will help the CLCS develop successful conservation and restoration plans for the Charlie Lake watershed.

2.0 LIST OF ABBREVIATIONS

Standard abbreviations and symbols were used whenever possible to minimize the length of this report; these are listed in Table 1. Readers not familiar with the limnological significance of the tabulated terms are referred to general texts, such as Wetzel (1983) and Wetzel and Likens (1991).

TABLE 1. LIST OF ACRONYMS AND ABBREVIATIONS.

Abbreviation	Definition	Abbreviation	Definition
A_o	Lake surface area	ng	10^{-9}g ("nanogram")
Al	Aluminum	Ni	Nickel
As	Arsenic	NTU	Nephelometric turbidity units
Ba	Barium	ρ	Flushing rate of lake
BCMELP	BC Ministrv of Environment, Lands and Parks	*P*	Probability
B.O.D.	Biological oxygen demand	P	Phosphorus
C	Carbon	Pb	Lead
Ca	Calcium	Po	Polonium
Cd	Cadmium	*R*	Retention rate of lake
$CFU \cdot cL^{-1}$	Bacterial colonies/100 mL	r^2	r-square
Cl	Chloride	s	Second
Co	Cobalt	Sb	Antimony
Cr	Chromium	Si	Silica
Cu	Copper	S.E.	Standard error of mean
d	Day	sp.	Species (singular)
dam^3	1000 m^3	spp.	Species (plural)
Δ	delta ("change in")	SS	Suspended solids
D.O.	Dissolved oxygen	TAC colour	Total absorbable colour
EMS	Environmental Monitorina System	TC colour	True colour
Fe	Iron	TDP	Total dissolved phosphorus
g	Gram	TDS	Total dissolved solids
hr(s)	Hour(s)	TKN	Total Kjeldahl nitrogen
K	Potassium	TN	Total nitrogen
L	Liter	TP	Total phosphorus
M	Mega (10^6)	TS	Total solids
μ	Micro (10^{-6})	U	Uranium
Mg	Magnesium	V	Vanadium
Mn	Manganese	VR	Very rare
Mo	Molybdenum	$\overline{X}$	Average
MPN	Most probable number	yr(s)	Year(s)
n	Samole size of a discrete unit	$\overline{Z}$	Average depth of lake
N	Total sample size	Z_m	Maximum lake depth
N	Nitrogen	Zn	Zinc
Na	Sodium		

3.0 STUDY AREA

3.1 LOCATION AND TERRAIN

Charlie Lake is located approximately 9 km northwest of the City of Fort St. John, with the center point of the lake having the coordinates 55°20'00" N/120°59'25" W (Figure 1). The watershed lies entirely within the Boreal-White-and-Black-Spruce biogeoclimatic zone (BC Ministry of Forests 1992). Perhaps due to the frequency of historical forest fires and current land-management practices, forests in the Charlie Lake area are dominated by early seral species, with the most-dominant tree species being trembling aspen (*Populus tremuloides*) (BC Ministry of Forests 1992).

Soils surrounding Charlie Lake are somewhat variable (see Lord and Green (1986)). The west side of the watershed is dominated by Alcan soils, whereas soils on the east side can be described as patchworks of Alcan-Buick, Alcan-Murdale, Murdale and Murdale-Esher soils (Table 2). While the soils surrounding the northern drainage valleys (i.e., Coffee and East and West Stoddard creeks) are primarily of the Alcan-Murdale type, soils within the creek valleys themselves are of the Kenzie (upper reaches of East and West Stoddart creeks), Goose (lower reaches of Coffee and East and West Stoddart creeks) and Eaglesham (mouth of Coffee Creek) types.

With the exception of the northeastern shoreline of Charlie Lake which is steeply faced (sheer cliffs), the relief of the watershed is, in general, low to moderate (Lord and Green 1986) (Plates 1 and 2). The west side of the watershed is gently to moderately sloped (6-15%) with some strong slopes (16-30%) occurring south of Charlie Lake Provincial Park. The east side of the watershed is "gently rolling" (0.5-2.5%) with some strong slopes (up to 30%) occurring near drainage valleys. While the area around the mouth of Coffee Creek is more-or-less flat (0-2.5%), slopes along the lower reaches of Coffee and East and West Stoddart creeks can be as high as 5% (gentle slope). Gradients surrounding the northern drainage valleys range from gently sloping (6%) to strongly sloping (20%).

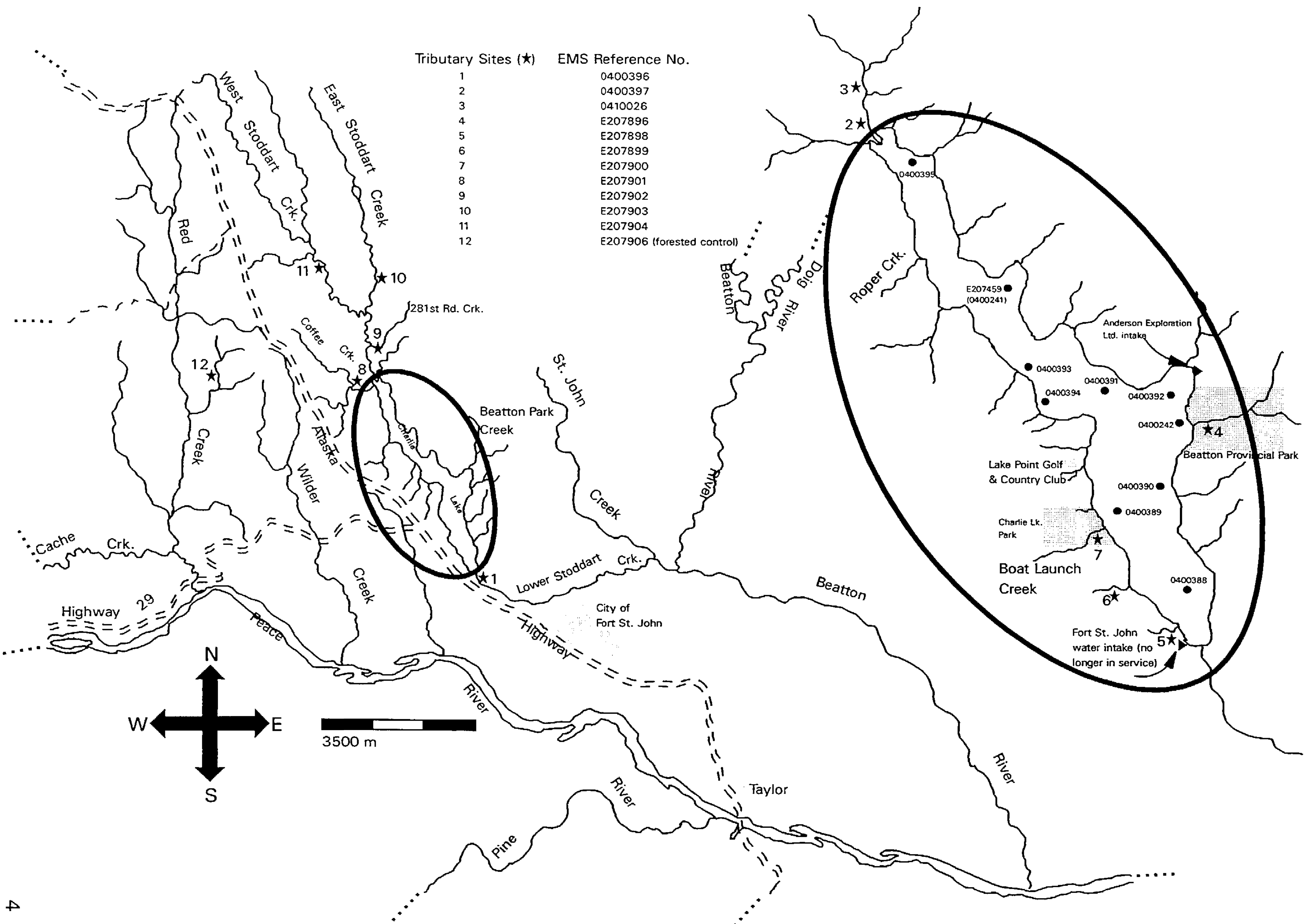

FIGURE 1. CHARLIE LAKE WATERSHED (NORTH EASTERN BRITISH COLUMBIA, CANADA) SHOWING THE LOCATION OF SAMPLING SITES.

TABLE 2. CHARLIE LAKE WATERSHED SOILS (MODIFIED FROM LORD AND GREEN (1986)).

Soil name	Landform	Description and distribution within British Columbia	Permeability	Soil composition
Alcan	loamy and clayey morainal; weakly calcareous and saline	occurs west of Charlie Lake and near Goodlow; inclusions of gleyed soils and Gleysols are < 30%; strongly acidic to depths < 150 cm	moderately well drained/ moderately pervious	Orthic Gray Luvisols (Alcan); some gleyed soils and Gleysols
Alcan-Buick	loamy and clayey morainal; weakly acidic, calcareous and saline	occurs on high plateau lands west of Charlie Lake and east of Cecil Lake; Buick soils make up about 40% of total composition	moderately well drained/ moderately pervious (Alcan); poorly drained/ slowly pervious (Buick)	Orthic Gray Luvisols (Alcan); Orthic Luvic Gleysol (Buick)
Alcan-Murdale	loamy and clayey morainal; weakly calcareous and saline	occurs near Montney and Cache Creek; Murdale soils make up 30-40% of total composition; found mainly on steep slopes having southern aspects	well drained/ moderately pervious	Orthic Gray Luvisol (Alcan); Dark Gray Solod (Murdale)
Eaglesham	moderately decomposed sedge fen peat	associated primarily with fluvial and lacustrine materials; 40-60% organic and Gleysol composition; peaty materials are usually 3-5 cm thick over mineral soils	very poorly drained/ slowly pervious	Terric Mesisol (Eaglesham) with Typic Mesisol and Gleysols
Goose	clayey lacustrotill and glaciolacustrine	primarily found in the valleys of major drainage systems; > 30% is comprised of other poorly drained soils and organics	poorly drained/ moderately pervious	Orthic Humic Gleysol (Goose) and peaty phase Gleysols
Kenzie	moderately decomposed moss bog peat	widespread distribution with main concentrations in the southwest and northeast regions; sphagnic phase bog soils (Kenzie) predominant, but variable amounts of fen peats, typic subgroups and Gleysols occur	very poorly drained/ slowly pervious	Terric Mesisol, sphagnic phase
Murdale	loamy and clayey morainal; weakly calcareous and saline	primarily found on the south-facing slopes of ridges north of Dawson Creek and Fort St. John	well drained/ moderately pervious	Dark Gray Solod (Murdale) with Black Solods and Gray Luvisols
Murdale-Esher	loamy and clayey morainal (Murdale); clayey and loamy lacustrotill (Esher); weakly calcareous and saline	primarily found on ridges north of Dawson Creek and Fort St. John; this unit generally lies beneith the Murdale unit on gentler slopes; Esher soils make up about 40% of the total composition	well drained/ moderately pervious (Murdale); moderately well drained/ moderately to slowly pervious (Esher)	Dark Gray Solods (Murdale, Esher) with Gray Luvisols

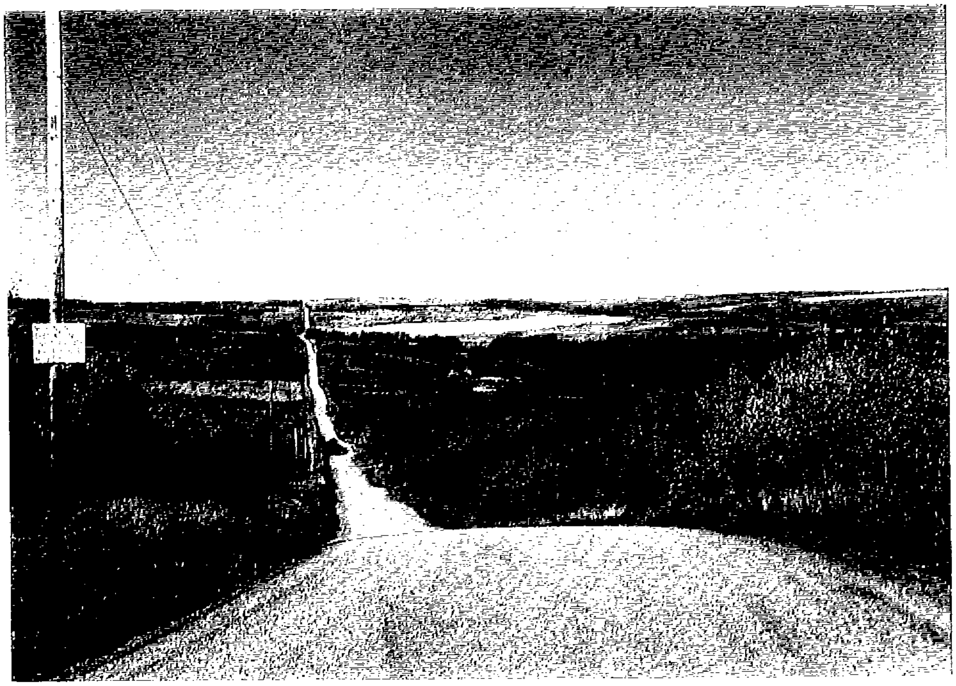

Plate 1. Upper Charlie Lake watershed, 1996, showing low-relief agricultural landscape.

Plate 2. Upper Charlie Lake watershed, 1996, showing cleared agricultural landscape.

3.2 CLIMATE

Climatological data collected between 1961 and 1990 indicate that air temperatures in the Fort St. John area are typically < 0 °C November to March, inclusively, with January being the coolest month (daily average = -10.8 °C; minimums < -40 °C) (Table 3). The warmest months are typically June ($\overline{X}$ = 19.4 °C), July ($\overline{X}$ = 21.5 °C) and August ($\overline{X}$ = 20.3 °C). During extremely hot periods, air temperatures can get as high as 30 °C.

On average, the area receives about 295 mm of rain annually, with June and July being the rainiest months (Table 3). Although small amounts of snow can fall during summer cold-snaps, the rate of snow accumulation tends to be greatest in December and January (Table 3).

Due to its exposed terrain and proximity to the Rocky Mountain foothills, the Fort St. John area is subject to heavy winds (Table 3). Winds tend to flow from the north December to March, at an average rate of 15 $km \cdot hr^{-1}$. While winter winds typically flow from the north, they are frequently interrupted by southwesterly gusts that approach 100 $km \cdot hr^{-1}$. Wind flows primarily from the southwest ($\overline{X}$ = 15 $km \cdot hr^{-1}$) during the spring, summer and autumn months.

TABLE 3. CLIMATIC SUMMARY FOR THE CHARLIE LAKE REGION, AS MEASURED AT THE FORT ST. JOHN AIRPORT (1961 TO 1990). DATA FROM ENVIRONMENT CANADA (1993).

Month	Mean daily temperature (°C)	Total monthly rainfall (mm)	Total monthly snowfall (cm)	Mean daily windspeed ($km \cdot hr^{-1}$)/most frequent direction	Maximum hourly windspeed ($km \cdot hr^{-1}$)/most frequent direction	Total monthly bright sunshine (hours)
January	-10.8	0.7	35.9	16/N	89/SW	75.9
February	-6.5	0.2	28.4	16/N	84/SW	112.9
March	-0.6	0.8	28.3	14/N	68/SW	168.0
April	8.8	7.3	14.0	16/SW	77/SW	235.5
May	15.5	34.1	6.6	16/SW	77/SW	279.9
June	19.4	66.5	0.4	15/SW	64/SW	295.8
July	21.5	73.7	0.0	13/SW	80/SW	309.8
August	20.3	56.7	0.8	13/SW	58/W	273.5
September	14.8	39.7	4.3	14/SW	64/SW	170.0
October	8.7	12.8	15.1	17/SW	80/S	144.5
November	-3.3	2.6	29.6	15/SW	74/SW	85.2
December	-9.1	0.7	34.8	15/N	97/SW	61.9

3.3 LAND USE AND HUMAN SETTLEMENT

The region's economy is based on, in order of relative importance, oil and gas, forestry (e.g., aspen pulping, sawmilling) and agriculture (Fort St. John Chamber of Commerce, verb. comm. to T.D. French, March 1997). The City of Fort St. John (population 15,000) and District of Taylor (population 1,000) are the only major population centers located in the vicinity of Charlie Lake. Although these settlements add to the human usage of the Charlie Lake watershed, they are located outside of the lake's drainage network and, thus, they do not contribute urban runoff to the system.

Charlie Lake has a shoreline community consisting of 1500 to 2000 individuals. Although there are several well-dispersed country residences in the watershed, residential development is concentrated along the southern shorelines, with one residential corridor extending from the area surrounding Lake Point Golf and Country Club to the south end of the lake and another from the south edge of Beatton Park to the south end of the lake (Figure 1). Many of the lakeshore properties have lawns, which are presumably fertilized from time to time, that extend to the waters edge and patches of cleared land (exposed soil), indicating that lakeshore development has proceeded with few ecological considerations (Plates 3 and 4).

To the best of our knowledge, land utilization patterns in the Charlie Lake watershed have not been quantified in the recent past. However, an unpublished report commissioned by the Peace River-Liard Regional District in 1980 concluded the following about land utilization in the watershed from ground surveys and air photos taken in the mid-1970s:

(1) private land accounts for > 80% of the watershed;
(2) vacant Crown land accounts for 16% of the watershed;
(3) 2-3% of the land is reserved as parkland (Charlie Lake Provincial Park and Beatton Park);
(4) of the 35 km of shoreline, 22 km are privately owned, 2.5 km are reserved as parkland and 10.5 km are vacant Crown land;
(5) approximately 40% of the watershed has been cleared for agricultural or other reasons;
(6) the shorelines of Coffee and East and West Stoddart creeks have been cleared for much of their length;
(7) there is evidence of sediment erosion in the Stoddart Creek valley; and
(8) approximately 600 cows, 300 sheep, and 100 horses live in the watershed.

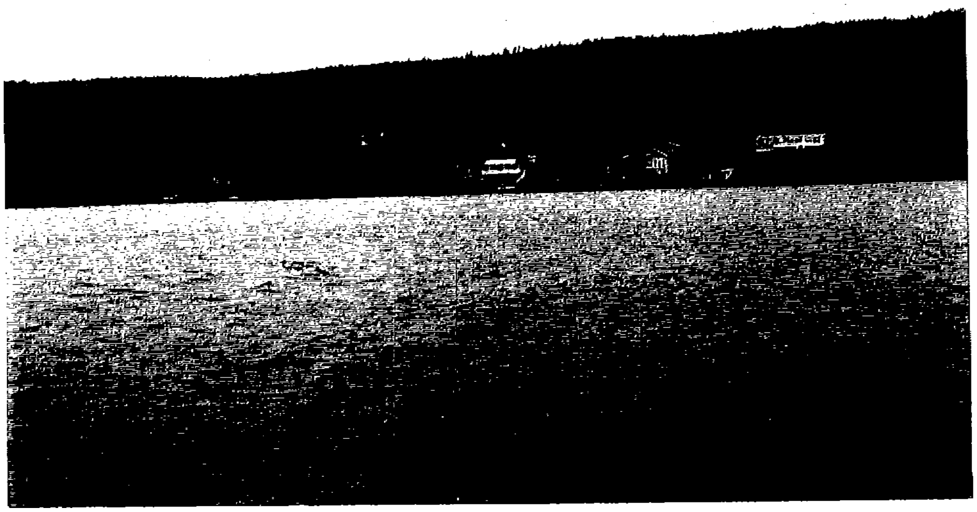

Plate 3. Residential development on shoreline, 1996.

Plate 4. Land cleared right to lake shore, 1996. Note exposed soils.

3.4 POLLUTION SOURCES

There are no licensed point-source polluters in the Charlie Lake watershed. Although diffuse-source contaminant loading to Charlie Lake has not been quantified to date, such pollution may enter the lake via a number of avenues. Sediment pollution in the watershed appears to be generated from streamshore and lakeshore erosion. Streamshore erosion is particularly problematic in cleared drainage valleys where cattle are permitted to graze along shorelines (e.g., the northern tributaries of East Stoddart Creek; T.D. French and N.B. Carmichael, pers. obs., September 1996) (Plates 5 and 6). Lakeshore erosion is most evident along the west shoreline (T.D. French and N.B. Carmichael, pers. obs., September 1996). While lakeshore erosion is a natural process, anecdotal information suggests that past increases in lake level (Section 5.3) have accelerated erosional processes (Mr. B. Ohland, verb. comm. to T.D. French, November 1996) (Plates 7 and 8). Sediment pollution to Charlie Lake may also come from ground drilling (oil and gas exploration), agricultural runoff (Nordin and Pommen 1985) and, in particular, the roads that service the region. Given that sediment pollution in other systems has had measurable effects on aquatic biota (e.g., Cordone and Kelley 1961; Hausle and Coble 1976; Barton 1977; Auld and Schubel 1978; Rosenberg and Wiens 1978; Lenat *et al.* 1981; Fudge and Bodaly 1984; Berg and Northcote 1985; Lisle and Lewis 1992; Anderson *et al.* 1996; Newcombe and Jensen 1996), it is likely that sediment generation in the Charlie Lake watershed is affecting aquatic ecosystems to some degree. Since biologically-active exchangeable salts (e.g., nutrients, metals, pesticide and herbicide residues, etc.) are bound to the surface of almost all natural soils (e.g., Brady 1974), sediment entering Charlie Lake likely carries a significant chemical load with it.

Up until the early 1990s, most lakeshore residences were serviced by evaporative lagoons or mounded septic fields. According to Urban Systems Ltd. (undated report), these waste disposal systems did not adequately treat or contain domestic sewage. Thus, it is generally believed that residential development in the Charlie Lake watershed has resulted in the production of nutrient-rich and fecal-coliform-contaminated pollution. Runoff from livestock arenas may also contribute nutrients and fecal coliforms to the system.

It has long been known that precipitation can remove contaminants from the atmosphere (Radke *et al.* 1980). In fact, it has been estimated that 70-80% of the total mass of aerosol particles removed from the trophospere of temperate latitudes is scrubbed out by precipitation (SMIC, 1971). Contaminants removed from the atmosphere by precipitation can end up in lakes and streams (e.g., Peters 1977; Scheider *et al.* 1979; Galloway *et al.* 1982; Munger 1982; Verry 1983; Likens *et al.* 1984; Krug *et al.* 1985; Brunskill and Wilkinson 1987; Linsey *et al.* 1987; Clair *et al.* 1995). Given that there are several industrial operations in the Fort St. John area that release particulate matter, inorganic and volatile organic carbon, nitrous oxides and sulfurous compounds into the atmosphere (BCMELP, unpubl. data), it is conceivable that pollutants enter Charlie Lake with atmospheric fallout.

Plate 5. Cows grazing and watering in riparian zone (Stoddart Creek sub-basin, 1996).

Plate 6. Hoof prints and erosion in riparian zone (Stoddart Creek sub-basin, 1996).

Plate 7. Lakeshore erosion resulting from fluctuating water levels.

Plate 8. Several lakeshore properties have stabilization walls to slow shoreline erosion.

3.5 LICENSED WATER WITHDRAWALS

There are 15 licensed water withdrawals in the Charlie Lake watershed that have a combined permitted withdrawal of 8123 $dam^3 \cdot yr^{-1}$ (Table 4). Of the 15 licenses, six are held by the City of Fort St. John, two by Lake Point Golf & Country Club Inc. and seven by single-license users. With the exception of license C034416 which permits a withdrawal from Lower Stoddart Creek, all withdrawals are taken from either the lake itself or from tributaries (Table 4).

The City of Fort St. John (Σ licensed withdrawals = 6,637 $dam^3 \cdot yr^{-1}$) and Anderson Exploration Ltd. (licensed withdrawal = 1,197 $dam^3 \cdot yr^{-1}$) have, by far, the largest permitted withdrawals. In fact, their combined allowable withdrawals account for 97% of all licensed water consumption (Table 4). The City of Fort St. John and Anderson Exploration Ltd. are the only license holders in the watershed that are required, by the terms of their licenses, to meter their withdrawals (Table 5). Thus, little is known about the water withdrawal patterns of low-volume users.

According to schematic diagrams in UMA Engineering Ltd. (1991), the City of Fort St. John and Anderson Exploration Ltd. take their water from the 689.92-m (geodetic) and 687.40-m depth contours, respectively. Based on the lake's high-water mark (692.10 m) (BCMELP 1986), these geodetic depth contours translate to water depths of about 2.18 m and 4.70 m, respectively. Daily measurements (1993-1996) of water-intake volumes taken by the City of Fort St. John indicate that the City has maintained a fairly constant rate of water withdrawal over the past four years (Table 5). By comparison, monthly water-intake data (1992-1996) collected by Anderson Exploration Ltd. indicate that the Company is gradually reducing its water consumption rate (Table 5). Thus, the average monthly rate of water withdrawal was 70 ± 3 (average ± 1S.E.) dam^3 in 1992, 63 ± 3 dam^3 in 1993, 58 ± 1 dam^3 in 1994, 44 ± 3 dam^3 in 1995 and 29 ± 1 dam^3 in 1996. According to UMA Engineering Ltd. (1991), Anderson Exploration Ltd. will stop taking water from Charlie Lake in the year 2009.

TABLE 4. LICENSED WATER WITHDRAWALS IN THE CHARLIE LAKE WATERSHED (FROM THE BCMELP INTERNET SITE HTTP://WWW.ENV.GOV.BC.CA/WAT/WRS/QUERY/LICENCES/LICENCES.HTM).

License holder	License No.	Purpose	Quantity ($m^3 \cdot yr^{-1}$)	Estimated effect on lake level (± mm)[a]	Water source
Hugh & Earl Creighton, Site 12, Comp 34, SS2, Fort St. John, BC, V1J 4M7	C027420	Domestic	1,659	-0.1	Roper Creek
James & Marion Sodergren, Site 5, Comp 18, SS2, Fort St. John, BC, V1J 4M7	C034416	Irrigation	8,018	-	Lower Stoddart Creek[b]
Herbert Just, c/o Callison & Company Ltd., 10419-100th Street, Fort St. John, BC, V1J 3Z3	C039877	Irrigation	93,746	-6.2	Stoddart Creek
BC Ministry of Transportation & Highways, 10716 100th Avenue, Fort St. John, BC, V1J 1Z3	C103660	Road spreading (dust control)	3,319	-0.2	Stoddart Creek
Kurt & Heidi Schwarz, PO Box 6852, Fort St. John, BC, V1J 4J3	C106025	Stock watering	1,659	-0.1	Stoddart Creek
Lake Point Golf & Country Club, Inc., PO Box 6208, Fort St. John, BC, V1J 4H7	C024526 Z105289	Irrigation Pond storage/watering	55,508 123,350	-3.7 -8.2	Charlie Lake Charlie Lake
Citv of Fort St. John. 10631-100th Street. Fort St. John. BC. V1J 3Z5	C025342 C025343 C025344 C029426 C033894 C033895 F059430	Waterworks Waterworks Waterworks Waterworks Waterworks Waterworks In-lake storage	165,932 248,898 1.410.424 829,661 248,898 3.733.476 7.401.000	-438.6 (sum of waterworks) +489.0 (storage)	Charlie Lake Charlie Lake Charlie Lake Charlie Lake Charlie Lake Charlie Lake Charlie Lake
Anderson Exploration Ltd., 10504-87th Avenue, Fort St. John, BC, V1J 5P7	C068852	Oil-field injection	1.196.618	-79.1	Charlie Lake
Peace Block Enterprises Ltd., 9016 101st Avenue, Fort St. John, BC, V1J 2A6	C070688	Truck washing	1,659	-0.1	Charlie Lake

[a] computed under the assumption that users consume a volume equal to that permitted by their license.
[b] Lower Stoddart Creek drains Charlie Lake; therefore, this withdrawal does not have a significant effect on lake levels.

TABLE 5. CITY OF FORT ST. JOHN AND ANDERSON EXPLORATION LTD. WATER WITHDRAWAL VOLUMES. NOTE: CITY OF FORT ST. JOHN STOPPED DRAWING WATER FROM CHARLIE LAKE ON OR ABOUT FEBRUARY 29, 1998 (THE CITY NOW DRAWS ITS DRINKING WATER FROM THE PEACE RIVER).

Month	Average monthly withdrawal by Fort St. John, 1993 to 1996 (dam^3)[b]	Total withdrawal by Anderson Exploration Ltd., 1992 to 1996 (dam^3)[a]				
		1992	1993	1994	1995	1996
January	207	69	61	54	59	31
February	194	76	53	52	53	30
March	204	86	66	61	60	32
April	194	77	55	61	57	32
May	216	78	na	58	56	33
June	215	76	56	57	40	36
July	209	72	74	59	36	33
August	205	70	73	62	35	25
September	195	62	74	58	34	26
October	196	66	63	57	37	24
November	190	52	52	53	31	22
December	194	61	na	60	30	22

[a] based on monthly data provided by Anderson Exploration Ltd. (1992 to 1996).
[b] based on daily withdrawal volumes provided by the City of Fort St. John (1993 to 1996).

4.0 METHODS

4.1 REVIEW OF EXISTING DOCUMENTATION

4.1.1 Public Interests and Concerns (PRRD 1996)

The Peace River Regional District (PRRD) held a townhall meeting for Charlie Lake and Area residents in late 1996. The primary objective of this meeting was to inform the public of factors that affect the productivity and "quality" of eutrophic lakes, such as Charlie Lake. To identify public interests and concerns regarding the uses and quality of Charlie Lake and its watershed, the PRRD distributed a questionnaire (sample size not reported) asking participants to rank their greatest interests (choices were motorized and non-motorized pleasure boating, summer fishing, ice fishing, seadoing, snowmobiling, x-country skiing, SCUBA diving, swimming, family picnicking and recreation, residential development, wildlife and bird viewing and water skiing) and concerns (choices were algal growth, fishing quality, lakeshore development, weekend noise levels, wildlife and bird habitat destruction, wildlife and bird harassment by pets and inconsiderate boaters and water craft users). Public interests and concerns were ranked with a simple scoring system: 3 points for first choice, 2 for second, 1 for third and 0.5 for any other non-prioritized choices. The PRRD plans to use the questionnaire responses to prioritize where efforts should be directed in planning for the lake's future.

4.1.2 Agricultural Pesticide Residues (BCMELP 1992)

Fish tissue (n = 5 walleye), bottom sediment (n = 1 composite sample) and water (n = 1 grab sample) samples were analyzed for a diverse array of organophosphate, organochlorine and solvent soluble pesticides at Zenon Laboratories, Burnaby, BC, with methods having detection limits given in Tables 6-8 (Diagnostic Pesticide Scan).

The walleye (5) were gill netted from two pelagic sites on May 11 and 15, 1992: near Tributary 19 embayment and near site 0400394 (Figure 1). Their bodies were wrapped twice in tinfoil and shipped frozen and in the dark to the laboratory. Bottom sediments were collected at sites 0400390, 0400395 and 0400391 with an Eckman dredge. The samples were homogenized into a single-sample composite, and shipped to the laboratory in an acid-washed amber bottle. The water sample was collected (acid-washed amber bottle) from Stoddart Creek about half way between the 281st Road Creek inflow and the juncture of East and West Stoddart creeks (Figure 1).

TABLE 6. MINIMUM DETECTABLE CONCENTRATIONS FOR SOLVENT SOLUBLE HERBICIDES (BCMELP, 1992, UNPUBL. DATA).

Compound	Minimum detectable concentrations		
	1L Water ($mg \cdot L^{-1}$)	10g Sediment ($\mu g \cdot g^{-1}$)	10g Fish flesh ($\mu g \cdot g^{-1}$)
Alachlor	0.001	0.1	0.1
Allidochlor	0.0005	0.05	0.05
Atrazine	0.0002	0.02	0.02
Bromacil	0.0001	0.01	0.01
Chlorpropham	0.0005	0.05	0.05
Chlorthal, dimethyl-	0.0001	0.01	0.01
Dichlobenil	0.00005	0.005	0.005
Dichlofop, methyl-	0.0001	0.01	0.01
Dinitramine	0.0003	0.03	0.03
Diuron	0.0005	0.05	0.05
Eptam	0.001	0.1	0.1
Flamprop, methyl	0.0001	0.01	0.01
Hexazinone	0.0005	0.05	0.05
Metribuzin	0.0001	0.01	0.01
Metobromuron	0.0005	0.05	0.05
Monuron	0.0005	0.05	0.05
Nitrofen	0.0001	0.01	0.01
Oxyfluorfen	0.0001	0.01	0.01
Prometryne	0.0002	0.02	0.02
Pronamide	0.0001	0.01	0.01
Propanil	0.0002	0.02	0.02
Propazine	0.0002	0.02	0.02
Simazine	0.0002	0.02	0.02
Terbacil	0.0005	0.05	0.05
Terbutylazine	0.0005	0.05	0.05
Triallate	0.0001	0.01	0.01
Trifluralin	0.0001	0.01	0.01
Vernolate	0.0002	0.02	0.02

TABLE 7. MINIMUM DETECTABLE CONCENTRATIONS FOR ORGANOPHOSPHATE-BASED PESTICIDES (BCMELP, 1992, UNPUBL. DATA).

Compound	Minimum detectable concentrations		
	1L Water ($mg \cdot L^{-1}$)	10g Sediment ($\mu g \cdot g^{-1}$)	10g Fish flesh ($\mu g \cdot g^{-1}$)
Acephate	0.0005	0.05	0.05
Azinphos, methyl	0.0005	0.05	0.05
Bromphos	0.0001	0.01	0.01
Carbophenthion	0.0001	0.01	0.01
Chlorfenvinfos	0.0001	0.01	0.01
Chlorpyrifos	0.0001	0.01	0.01
Demeton	0.0002	0.02	0.02
Diazinon	0.0002	0.02	0.02
Dichlorvos	0.0001	0.01	0.01
Dimethoate	0.0002	0.02	0.02
Dimethoate, O	0.002	0.02	0.02
Ethion	0.0005	0.05	0.05
Fenitrothion	0.0002	0.02	0.02
Fensulfothion	0.0001	0.01	0.01
Fenthion	0.0002	0.02	0.02
Fonofos	0.0002	0.02	0.02
Fonofos-O	0.0005	0.05	0.05
Iodofenphos	0.0001	0.01	0.01
Malathion	0.0001	0.01	0.01
Methamidophos	0.0005	0.05	0.05
Methidathion	0.0002	0.02	0.02
Mevinphos	0.0005	0.05	0.05
Naled	0.0001	0.01	0.01
Parathion	0.0001	0.01	0.01
Parathion, methyl	0.0002	0.02	0.02
Phorate	0.0002	0.02	0.02
Phosalone	0.0005	0.05	0.05
Phosmet	0.0003	0.03	0.03
Phosphamidon	0.0005	0.05	0.05
Sulfotep	0.0002	0.02	0.02
Tetrachlorvinphos	0.0002	0.02	0.02

TABLE 8. MINIMUM DETECTABLE CONCENTRATIONS FOR ORGANOCHLORINE-BASED PESTICIDES (BCMELP, 1992, UNPUBL. DATA).

Compound	Minimum detectable concentrations		
	1L Water ($mg \cdot L^{-1}$)	10g Sediment ($\mu g \cdot g^{-1}$)	10g Fish flesh ($\mu g \cdot g^{-1}$)
Aldrin	0.00001	0.002	0.002
BHC, alpha-	0.00001	0.002	0.002
BHC, beta-	0.00001	0.002	0.002
BHC, delta-	0.00001	0.002	0.002
Chlordane, alpha-	0.00005	0.01	0.01
Chlordane, gamma-	0.00005	0.01	0.01
DDD, p,p-	0.00005	0.01	0.01
DDD, p,p'-	0.00005	0.01	0.01
DDE, p,p'-	0.00005	0.005	0.005
DDT, o,p-	0.00005	0.01	0.01
DDT, p,p'-	0.00005	0.01	0.01
Dieldrin	0.00005	0.01	0.01
Endosulfan I	0.00005	0.01	0.01
Endosulfan II	0.00005	0.01	0.01
Endosulfan sulphate	0.0001	0.02	0.02
Endrin	0.00005	0.01	0.01
Hexachlorbenzene	0.000005	0.001	0.001
Heptachlor	0.00001	0.002	0.002
Heptachlor epoxide	0.00002	0.004	0.004
Lindane (BHC, gamma-)	0.00001	0.002	0.002
Methoxychlor	0.0001	0.02	0.02
Mirex	0.0001	0.02	0.02
Nonachlor, trans-	0.00005	0.01	0.01
Oxychlordane	0.00005	0.01	0.01
PCB	0.0004	0.05	0.05

4.1.3 Fish Parasites (Bangham and Adams 1954)

Twenty-eight white suckers (*Catostomus commersoni*) and 16 brook sticklebacks (*Eucalia inconstans*) were gill netted or seined, respectively, from Charlie Lake during the late summer 1952. External and internal eye, gill and gut tissues/contents were examined for parasites under a wide-field binocular microscope. Gut tissues and contents were examined first in an untreated state and then again after being soaked in a 1% sodium bicarbonate solution (dissolves mucous). Reference parasites were relaxed in water, preserved in 5% formalin and archived for historical record.

4.1.4 Paleolimnological Investigations (Reavie *et al.* 1995b)

The eutrophication history (mid-1800s to 1991) of Charlie Lake was inferred from changes in diatom community structure with a sediment coring technique. A 45-cm deep sediment core was collected from a depth of 12 m (Z_m) with a Kajak-Brinkhurst gravity corer equipped with a 6.35-cm (inside diameter) core tube (Glew 1989) in late summer 1991. The core was sectioned immediately after collection into 1-cm slices with a close-interval extruder (Glew 1988); each slice was stored in Whirlpak® bag until analysis.

Subsamples (approx. 30 g) of selected sediment intervals were weighed, oven dried for 24 hr at 110 °C and ground to a fine dust. Dried samples were sent to Atomic Energy of Canada Limited (Chalk River, Ontario) for ^{210}Pb dating. ^{210}Pb dates were computed from alpha spectroscopy determinations of ^{210}Po, a decay product of ^{210}Pb (Cornett *et al.* 1984). Unsupported ^{210}Pb concentrations were computed by subtracting supported ^{210}Pb concentrations (baseline ^{210}Pb activity naturally present in sediments) from total ^{210}Pb activity. Dates were determined from unsupported isotopes using the constant rate of supply model (Appleby and Oldfield 1978; Binford 1990).

Diatom analysis (species composition and enumeration) was performed on twelve evenly spaced wet-sediment subsamples (0.3-1.0 g). Subsamples were heated for 1 hr in a potassium dichromate-sulphuric acid mixture to digest organic matter. They were then washed repeatedly with distilled water and centrifuged until they were cleared of residual acid. Siliceous remains were settled onto cover slips and mounted onto glass slides using Hyrax®. At least 500 diatom valves were identified (Patrick and Reimer 1966; Krammer and Lange-Bertalot 1986, 1988, 1991a,b; Camburn *et al.* 1984-1986) and counted on each slide under oil immersion at 1000X magnification. Chrysophyte cysts were enumerated and expressed relative to the total diatom count (Smol 1985).

Historical total lake-water P concentrations were inferred by canonical relationships (canonical correspondence analysis) between modern diatom assemblages and lake-water TP concentrations developed from data (diatom and lake-water phosphorus) collected from 59 British Columbia lakes (see Ter Braak 1990a,b; Line 1994; Reavie *et al.* 1995a).

4.1.5 Preliminary Limnological Investigations (Nordin and Pommen 1985)

Nordin and Pommen (1985) summarized post-freshet water quality data collected (procedures not specified) at several Charlie Lake sites and two sites on the primary inlet (Stoddart Creek):

Charlie Lake (1974 to 1976)

Sites (Figure 1): 0400241 (re-named to E207459), 0400242, 0400388, 0400389, 0400390, 0400391, 0400392, 0400393, 0400394 and 0400395

Parameters (*N*): TC colour (70), pH (203), TS (109), TDS (70), SS (181), specific conductance (203), turbidity (184), Secchi depth (22), TAC colour (70), total alkalinity (203), organic C (202), hardness (187), ammonia N (203), nitrate+nitrite N (95), nitrate N (125), organic N (187), TKN (185), total N (76), ortho-P (68), TDP (14), TP (203), inorganic C (90), chlorophyll *a* (163), Ca (203), Mg (203) and volatile solids (1)

Stoddart Creek (site 0400397 in 1974 and 1975; site 0410026 in 1982)

Sites (Figure 1): 0400397 and 0410026

Parameters (*N*): pH (9), TS (8), SS (9), specific conductance (9), turbidity (6), total alkalinity (6), organic C (6), hardness (6), ammonia N (9), organic N (9), TP (9), ortho-P (5), Ca (6) and Mg (6)

Several recommendations were made based on conclusions drawn from these data in view of regional land (e.g., agriculture, residential, oil and gas developments) and water (e.g., irrigation and potable water supplies, recreation and fisheries) use issues. These recommendations are summarized in Section 7.0 of the present report.

4.2 MORPHOMETRY

As will become evident in subsequent sections of this report, a detailed morphometric assessment of Charlie Lake was required before the objectives of this study could be fully addressed. In particular, it was necessary to develop a precise relationship between depth contour elevation (geodetic) and lake volume (i.e., the volume of water below a specified contour elevation). The first step in developing this relationship was to determine the surface area of each of the geodetic depth contours presented in BCMELP (1986). While these surface areas were tabulated by Mr. L.J. Barr (1989 unpubl. BCMELP Memorandum), we determined them again by planimetry (PLACOM digital planimeter, model KP-90N) to confirm their accuracy. The second step was to determine the volume (V) of each successive depth strata as follows (Wetzel and Likens 1991):

$$V=\frac{h}{3}\left(A_1+A_2+\sqrt{A_1A_2}\right) \qquad (1)$$

where h is the vertical thickness of the stratum, A_1 is the area of the upper stratum, and A_2 is the area of the lower stratum. The third step was to compute the lake volume below each geodetic depth contour. This was done by summing the strata volumes below the contour of interest. The relationship between depth contour elevation and volume was then determined by regressing volume ("cumulative storage") against elevation. Although the relationship between these variables is, when the entire elevation range is considered, non-linear, we felt that it was appropriate to perform the regression analysis as a series of linear step-functions (Zar 1984). The rationale behind this decision was that the relationship was almost perfectly linear between the surface contour (692.1 m) and the 686.0-m contour, but curvilinear below the 686.0-m contour (Figure 6).

We also estimated the total lake volume (Vt), surface area (A_o), maximum depth (Z_m), average depth ($\bar{Z}$), relative depth (Z_r), shoreline length, maximum length and watershed area. Shoreline length and surface area were measured by planimetry. Total length was measured from a 1:10,000 map with a ruler. Maximum depth was read directly from contours presented in BCMELP (1986). Catchment area was measured by Mr. Sarge Bhatti (BCMELP, GIS Coordinator) with GIS software. Total lake volume, mean depth, and relative depth were computed as follows (Wetzel and Likens 1991):

$$V_t = \frac{h}{3}\left(A_1 + A_2 + \sqrt{A_1 A_2}\right) \qquad (2)$$

where the equation variables are coded as in equation 1;

$$\bar{Z} = \frac{V_t}{A_o} \qquad (3)$$

and;

$$Z_r = \frac{50 Z_m \sqrt{\pi}}{A_o} \qquad (4)$$

4.3 BASIC HYDROLOGY

The number of tributaries draining directly into Charlie Lake was determined by counting tributary inflows resolved by 1:50,000 N.T.S. contour maps. Watershed boundaries and watershed areas were estimated by contour interpretation and planimetry, respectively (Wetzel and Likens 1991). There were not enough measures of discharge to adequately quantify the flow characteristics of these tributaries.

Evaporative water losses were modeled from long-term climatic data collected at the Fort St. John airport (Mr. R.L. Davis, BCMELP, Air Management Branch, unpubl. data, 1979). The model used in this computation was that presented in Priestley and Taylor (1972), which argues that:

$$LE = 1.26\left(\frac{S}{S+\gamma}\right)\left(Q^* - G\right) \qquad (5)$$

where LE is the evaporative flux ($MJ \cdot m^{-2} \cdot month^{-1}$), Q^* is the net radiation flux ($MJ \cdot m^{-2} \cdot month^{-1}$), G is the heat flux into either the soil or water body ($MJ \cdot m^{-2} \cdot month^{-1}$), γ is the psychrometric constant (0.66 mb $^{\circ}C^{-1}$), and S is the slope of the saturation of vapor pressure versus temperature curve at the mean monthly air temperature (0.66 mb $^{\circ}C^{-1}$). While modeled evaporation rates are probably not as reliable as measured evaporation rates, Stewart and Rouse (1976) reported that the Priestley-Taylor model can predict evaporation rates with reasonable accuracy. It was assumed that G = 0 in the Fort St. John area. The value of S was estimated using the regression equation (Dilley 1968):

$$S=\left(\frac{25029}{(T+237.3)^2}\right)\left(e\left(\frac{17.269}{T+237.3}\right)\right) \qquad (6)$$

where T is the mean monthly air temperature (°C) and e is the base of natural logarithms ($\cong$2.71828). Q* was estimated using the equation (Linacre 1968):

$$Q^*=K\downarrow(1-r)-2.988\left(0.2+0.8\frac{n}{N}\right)(100-T) \qquad (7)$$

where $K\downarrow$ is the solar radiation flux ($MJ \cdot m^{-2} \cdot month^{-1}$), r is the albedo (reflectivity of the surface), n is the number of hours of bright sunshine, and N is maximum possible hours of bright sunshine (available from meteorological tables). $K\downarrow$ was estimated from an Angstrom regression equation:

$$K\downarrow=Q_0\left(0.21-0.565\frac{n}{N}\right) \qquad (8)$$

where Q_0 is the solar radiation flux at the top of the earth's surface ($MJ \cdot m^{-2} \cdot month^{-1}$) (available from meteorological tables). The ratio $\frac{n}{N}$ was developed using solar radiation and hours of sunshine data from several British Columbia climate stations. The computed evaporation flux ($MJ \cdot m^{-2} \cdot month^{-1}$) was then converted to $mm \cdot month^{-1}$ with the equation:

$$1\ MJ\ m^{-2} \cdot month^{-1} = 0.423\ mm \cdot month^{-1} \qquad (9)$$

Monthly evaporate volume was then calculated as follows:

$$V=\frac{\left(\frac{mm}{1000}\right)(19{,}000{,}000)}{1000} \qquad (10)$$

where V is in dam^3 (cubic decameters) and the term "19,000,000" is A_o in square meters. In their study of Charlie Lake, UMA Engineering Ltd. (1991) used evaporation data collected at Hudson's Hope (~ 70 km southwest of Charlie Lake) as a surrogate for the evaporation rate of Charlie Lake. UMA Engineering Ltd.'s use of Hudson's Hope evaporation data was reasonable and justified because Hudson's Hope is quite close to Charlie Lake and the data are "real" (i.e., measured as opposed to modeled). However, since the terrain surrounding Hudson's Hope is much less exposed (i.e., steeper topography) than the terrain surrounding Charlie Lake it could be argued that evaporation rates based on data collected at Hudson's Hope would underestimate the evaporation rate of Charlie Lake. Although we may not have been totally justified in doing so, we chose to use our modeled estimates of evaporation rate in all water-budget computations rather than UMA Engineering Ltd.'s measured values.

The volume of water exported from Charlie Lake via Lower Stoddart Creek was estimated from daily discharge data obtained from Environment Canada (1994) for the years 1988-1992. Clearly, discharge in Lower Stoddart Creek varies between seasons, years and even days. With this in mind, the data were analyzed in terms of daily medians, daily minimums and daily maximums to provide a range of discharge possibilities. Flows were determined with a method based on the correlation between lake level and discharge (i.e., a stage-discharge relationship); thus, the discharge data presented in this report are estimates rather than actual measurements.

The stage hydrograph of the lake was estimated from daily measures of lake level collected over the years 1968 to 1979, 1987 to 1990 (Environment Canada 1994), and the years 1991 to 1994 (City of Fort St. John, unpubl. data). To quantify the sum of all water inputs to the lake (sum of tributary flows, groundwater flows and precipitation inputs), the water level data were first converted to geodetic elevation with the knowledge that 2.384 m (gauge level) is the equivalent of 692.100 m (geodetic elevation). Total water input (TWI) to Charlie Lake over any given time period was then estimated with the equation:

$$TWI = \Delta S + \sum outflows \qquad (11)$$

where ΔS is the change in lake volume over the time period of interest and $\sum outflows$ is the total outflow volume over the time period of interest. ΔS was computed with the equation:

$$\Delta S = V_{t1} - V_{t2} \qquad (12)$$

where V_{t1} is the lake volume at the beginning of the time period of interest and V_{t2} is the volume of the lake at the end of the time period of interest. V_{t1} and V_{t2} were computed with either equation 24 or 25 (depending on the geodetic elevation of the lake's surface). $\sum outflows$ were estimated by summing the water volumes lost through Lower Stoddart Creek, evaporation and withdrawals by Anderson Exploration Ltd. and the City of Fort St. John over the time period of interest. The proportion of TWI contributed by rain and snow was computed by multiplying daily water-equivalent measures (Environment Canada, Atmospheric Services, Fort St. John Station, unpubl. data) by A_o.

The lake's flushing rate (ρ, yr^{-1}) was computed as (Dillon 1975):

$$\rho = \frac{\sum outflows}{136{,}839\ dam^3} \qquad (13)$$

where $\sum outflows$ was calculated as described in the preceding paragraph and 136,839 dam^3 is lake volume at full capacity. Retention rate (*R*, yr), or the time it takes for the lake to replace its volume, was computed as the reciprocal of ρ:

$$R = \frac{1}{\rho} \qquad (14)$$

4.4 PHYTOPLANKTON COMMUNITY DYNAMICS

Charlie Lake's phytoplankton community was described from 1-L surface (about 1-m depth) samples collected at the north (E207459) and south (0400390) deep stations in 1984 (September), 1988 (April and May), 1990 (May, August to October, inclusive), 1991 (March, May to October, inclusive) and 1992 (March and May). Samples were preserved with Lugol's solution and sent to BCMELP Environmental Laboratory (Vancouver, BC) for taxonomic (species or genus resolution) and numeric (cell counts per mL) analyses (name and qualifications of taxonomists not specified).

Taxon-specific abundance was described in terms of peak abundance (highest observed density) and within-order relative abundance (highest observed density divided by sum of all peak cell counts within respective order). Bloom periodicity was described at the taxonomic level of order (classified as per Lee (1989)) from density versus date time-series plots, under the assumption that density variations were a function of seasonal phenology as opposed to year-to-year variation (i.e., samples were not all collected during the same year).

4.5 LIMNOLOGICAL SAMPLING

The lake (1974 to 1995), tributary (1974 to 1993) and sewage trunk system (1990 to 1995) data were collected by BCMELP staff (Prince George and Fort St. John). Lower Stoddart Creek data were collected by BCMELP (1974 to 1976, 1985) and Anderson Exploration Ltd. (1992 to 1994). Anderson Exploration Ltd. sampled the chemistry of their water intake (1992 to 1994) and the City of Fort St. John's water intake (1992 to 1994).

All BCMELP data were extracted from the provincial EMS database. Anderson Exploration Ltd. provided their data to the authors in MS Excel® spreadsheets. Neither database specified collection or analytical methods; thus, the methods described in this section are limited to descriptions of sampling location, sampling schedule and parameter selection.

4.5.1 BCMELP Site Descriptors

BCMELP sampled 10 sites on Charlie Lake, 11 tributary sites and one site on Lower Stoddart Creek. Table 9 gives the BCMELP EMS descriptor information for each site, site coordinates and site-specific sampling period. Readers should keep in mind that the EMS database is being updated continually, and that interested individuals can retrieve the raw data through the BCMELP EMS Helpdesk (EMSHELP@epdiv1.env.gov.bc.ca).

TABLE 9. BCMELP SITE DESCRIPTORS WITH SITE COORDINATES (MEASURED OFF 1:50,000 N.T.S. MAPS AND SAMPLING PERIOD).

BCMELP site no.	BCMELP descriptor	Site coordinates		Sampling period (mm/dd/yy)	
		Latitude	Longitude	First date	Last date
(a) Lake sites (data collected by BCMELP)					
0400241 (site 0400241 re-named to E207459; see E207459 row for site information)					
0400242	Charlie Lake off Beatton Park Creek inlet	56.3290	120.9566	05/29/74	03/23/76
0400388	Charlie Lake south arm. 1.5 km north of outlet	56.2847	120.9544	05/29/74	06/08/92
0400389	Charlie Lake south arm, 500 m east of Charlie Lake Park	56.3092	120.9758	05/05/74	03/23/76
0400390	Charlie Lake south deep station. 1.2 km east of Charlie Lake Park	56.3125	120.9642	05/29/74	08/17/98
0400391	Charlie Lake center off golf course	56.3331	120.9861	05/29/74	09/15/92
0400392	Charlie Lake bay mouth, north of Beatton Park	56.3328	120.9611	05/29/74	08/08/92
0400393	Charlie Lake north arm, north of golf course	56.3406	121.0081	05/29/74	08/22/84
0400394	Charlie Lake, center west shore	56.3333	121.0036	05/29/74	08/22/84
0400395	Charlie Lake north arm, 0.8 km from Stoddart Creek inlet	56.3858	121.0467	05/29/74	06/08/92
E207459	Charlie Lake north deep station	56.3569	121.0169	04/28/88	03/20/95
(b) Tributary and outlet sites (data collected by BCMELP)					
0400396	Charlie Lake outlet, near lake mouth	56.2747	120.9469	10/24/74	03/20/85
0400397	Stoddart Creek inflow, 2 km upstream Charlie Lake	56.4192	121.0586	09/12/74	03/23/76
0410026	Stoddart Creek inlet at 64 Road, upstream Charlie Lake	56.4203	121.0600	07/21/82	07/12/88
E207896	Beatton Park Creek, in Beatton Park	56.3285	120.9563	04/04/89	06/29/93
E207898	Residential Ditch #5	56.2825	120.9429	04/04/89	06/29/93
E207899	Residential Ditch #3	56.2908	120.9760	04/04/89	06/29/93
E207900	Boat Launch Creek	56.3053	120.9849	15/15/89	06/29/93
E207901	Coffee Creek at 114 Road	56.3829	121.0651	04/04/89	06/29/93
E207902	281st Road Creek	56.4066	121.0651	04/04/89	06/26/93
E207903	East Stoddart Creek, upstream Charlie Lake	56.4417	121.0656	04/04/89	06/26/93
E207904	West Stoddart Creek, upstream Charlie Lake	56.4469	121.1183	04/04/89	06/26/93
E207906	Unnamed Creek at mile 63.5 Road, Alaska Highway (Forested Control Site)	56.3810	121.2286	04/19/89	05/04/93
(c) Industrial water intakes (data collected by Anderson Exploration Ltd.)					
na	Anderson Exploration Ltd. pumphouse	56.3390	120.9429	04/22/92	08/10/94
na	City of Fort St. John intake water	56.2790	120.9760	04/22/92	08/10/94
(d) Sewage trunk system (data collected by BCMELP)					
E209658	Sewage export from Charlie Lake residential area	na	na	09/13/90	09/01/95

4.5.2 Tributaries (1974 to 1993)

BCMELP sampled selected Charlie Lake tributaries on the following schedule (by sampling decade):

Years 1974 to 1976

Sites (Figure 1): Stoddart Creek near mouth (0400396)

Schedule: 1 occasion April, 1974; 1-2 occasions/month April and June to August, 1975; 1 occasion March, 1976

Parameters (*N*): dissolved Ca (6); organic C (6); TAC colour (2); specific conductance (6); dissolved hardness (6); dissolved Mg (6); ammonia N (6); NO_2+NO_3 N (1); TKN (6); organic N (6); D.O. (2); pH (6); ortho-P (2); TDP (1); TP (6); filterable residues (6); non-filterable residues (6); temperature (3); NTU turbidity (5)

Years 1982 to 1989

Sites (Figure 1): Stoddart Creek at 64 Road (0410026) during period 1982 to 1989; West Stoddart Creek (E207904), East Stoddart Creek (E207903), 281^{st} Creek Road (E207902), Beatton Park Creek (E207896), Residential Ditch #5 (E207898), Residential Ditch #3 (E207899), Boat Launch Creek (E207900) and Coffee Creek (E207901) in 1989 only

Schedule: 1 occasion/month July, August and October, 1982; 1-2 occasions/month March, April, July, August and September, 1984; March (once), April (9 occasions) and May (9 occasions), 1985; March (once) and April (4 occasions), 1986; 1-2 occasions/month March to July, 1988; April (8 occasions), May (twice), June (3 occasions) and August (once), 1989

Parameters (*N*): Total C (1); organic C (1); dissolved Cl (1); *Enterococcus* coliforms (20); *E. coli* coliforms (30); fecal coliforms (55); *Streptococcus* coliforms (3); TAC colour (2); specific conductance (10); ammonia N (73); NO_2+NO_3 N (41); total N (9); TKN (10); organic N (9); BOD (1); pH (10); ortho-P (42); TDP (118); TP (108); dissolved K (1); filterable residues (5); non-filterable residues (7); dissolved Si (1); NTU turbidity (1)

Years 1990 to 1993

Sites (Figure 1): East Stoddart Creek (E207903), West Stoddart Creek (E207904), 281st Road Creek (E207902), Beatton Park Creek (E207896), Residential Ditch #5 (E207898), Residential Ditch #3 (E207899), Boat Launch Creek (E207900), Coffee Creek at 114 Road crossing (E207901)

Schedule: 1-2 occasions/month May to October, 1990; March (once), April (3 occasions), May (twice) and June (5 occasions), 1991; February (3 occasions), March (5 occasions) and April (4 occasions), 1992; January (twice), March (twice), April (4 occasions), May (once) and June (5 occasions) 1993

Parameters (*N*): Fecal coliforms (213); TDP (297); TP (273)

4.5.3 Outlet (1974 to 1994)

Lower Stoddart Creek (lake outlet) was sampled by BCMELP in 1974 to 1976 and 1985, and by Anderson Exploration Ltd. between 1992 and 1994 on the following schedule:

Years 1974 to 1976 and 1985 (BCMELP)

Sites (Figure 1): Lower Stoddart Creek at outlet of Charlie Lake (0400396)

Schedule: Once in October, 1974; 2 occasions/month April to August, 1975; twice in March, 1976; once in March, 1985

Parameters (*N*): Alkalinity to pH 4.5 (7); dissolved Ca (7); inorganic C (5); organic C 7); total C (1); dissolved Cl (1); fecal coliforms (1); TAC colour (4); dissolved hardness (7); dissolved Mg (7); ammonia N (8); NO_2+NO_3 N; organic N (7); TKN (8); total N (2); D.O. (3); pH (8); ortho-P (5); TDP (3); TP (8); filterable residues (8); non-filterable residues (8); total residues (8); specific conductance (8); temperature (4); NTU turbidity (7)

Years 1992 to 1994 (Anderson Exploration Ltd.)

Sites (Figure 1): Lower Stoddart Creek near outlet of Charlie Lake (near 0400396)

Schedule: 2 occasions/month April, May, July and August, 1992; 2 occasions/month January, February, April, May, July and August, 1993/1994

Parameters (N): total alkalinity (35); specific conductance (36); D.O. (35); ammonia N (36); NO_2+NO_3 N (35); temperature (22); ortho-P (36); TP (36)

4.5.4 Charlie Lake (1974 to 1995)

BCMELP sampled Charlie Lake on the following schedule (by sampling decade):

Years 1974 to 1976

Sites (Figure 1): 0400390, 0400388, 0400389, 0400391, 0400392, 0400393, 0400394 and 0400395

Schedule: 1-2 occasions/month May to October, 1974; 1-2 occasions/month April and June to November, 1975; 1 occasion March 1976

Parameters (*N*): Ammonia N (201), NO_2+NO_3 N (123), nitrate N (123), nitrite N (123), TKN (184), organic N (185), D.O. (132), ortho-P (67), TDP (14), TP (202), filterable residues (201), non-filterable residues (201), dissolved Mg (201), dissolved Ca (201), organic C (200), inorganic C (90), alkalinity to pH 4.5 (201), specific conductance (201), dissolved hardness (185), pH (201), temperature (179), NTU turbidity (182), TCU colour (60) and chlorophyll *a* (352)

Field parameters (D.O. and temperature) were measured near the surface (at an arms reach), mid-depth and near the bottom. Laboratory measured parameters, with the exception of chlorophyll *a* concentrations which were measured for surface waters only, were measured on surface and near-bottom waters.

Years 1984 to 1989

Sites (Figure 1): 0400392 and 0400395 (1984 only), 0400388 (1984 and 1985 only), E207459 (1988 and 1989 only), 0400390 (1984, 1985, 1986, 1987, 1988 and 1989)

Schedule: 1 occasion/month July to September, 1984; 1-2 occasions/month March and May, 1985, 1986; one occasion May 1987; 1-2 occasions/month April, May, July and August to October, 1988; 1-2 occasions/month March, May, June, August and September, 1989

Parameters (*N*): Alkalinity to pH 4.5 (2); total Al (4); total Sb (2); total As (4); total Ba (2); inorganic C (10); organic C (3); total Cd (4); dissolved Cl (4); total Cr (4); total Co (4); MPN fecal coliforms (5); $CFU \cdot cL^{-1}$ fecal coliforms (3); total Cu (4); chlorophyll *a* (23); total Fe (4); total Mn (4); total Mo (4), total Ni (4); ammonia N (109); nitrate N (18); nitrite N (33); NO_2+NO_3 N (72); TKN (25); organic N (14); D.O. (188); pH (117); ortho-P (144); TDP (126), TP (123); specific conductance (118); dissolved K (4); filterable residues (11); non-filterable residues (9); dissolved Si (47); temperature (181); dissolved Na (4); sulfate (4); total U (2); total V; total Zn (4)

Field measurements between 1984 and 1987 were taken from near the surface, mid-depth and near bottom, and laboratory parameters near the surface and bottom. In 1989, temperature and D.O. measurements were taken at 1-m intervals from the water's surface to the bottom; laboratory analyses were done on water samples collected from near the surface, mid-depth and near the bottom.

Years 1990 to 1995

Sites (Figure 1): E207459 and 0400390 (1990 to 1995), 0400388, 0400391 and 0400395 (1989 to 1992 only)

Schedule: 1 – 2 occasions/month May to October, 1990; March and May to October, 1991; January, March and May to September 1992; March and May to November, 1993; January, March, May to July, September and December, 1994; January to March, 1995

Parameters (*N*): Alkalinity to pH 4.5 (30); chlorophyll *a* (53); specific conductance (241); ammonia N (235); nitrate N (12); nitrite N (12); nitrate+nitrite N (231); TKN (87); D.O. (880); pH (241); ortho-P (415); TDP (416); TP (289); filterable residues (12); non-filterable residues (58); dissolved Si (95); temperature (901)

Temperature and D.O. measurements were taken at 1-m intervals from the water's surface to the bottom; laboratory analyses were done on water samples collected from near the surface, mid-depth and near the bottom.

4.5.5 Major Water Withdrawals (Anderson Exploration Ltd. and City of Fort St. John)

Anderson Exploration Ltd. sampled the chemistry of their and the City of Fort St. John's intake water over the years 1992 to 1994. The data were collected to assess whether the Company and/or the City were benefiting the lake by drawing nutrient rich water (i.e., increasing nutrient export and, thereby, slowing eutrophication processes), or concentrating nutrients in the lake by drawing relatively dilute water. The Company sampled the withdrawal water on the following schedule:

Years 1992 to 1994

Sites (Figure 1): Anderson Exploration Ltd. and City of Fort St. John pump houses

Schedule: 2 occasions/month April, May, July and August, 1992; 2 occasions/month January, February, April, May, July and August 1993, 1994

Parameters (*N*): Alkalinity to pH 4.5 (32); specific conductance (32); flow* (Anderson Exploration Ltd., monthly; City of Fort St. John, continually); ammonia N (32); NO_2+NO_3 (32); D.O. (32); ortho-P (32); TP (32); temperature (32)

*Fort St. John's withdrawal volumes were measured by the municipal Public Works department

While the City of Fort St. John stopped drawing water from Charlie Lake on or about February 28, 1998 (water supply now drawn from the Peace River), we suggest that the pumping facility may be used in the future to pump nutrient-rich hypolimnetic waters out of the lake to slow and/or help reverse eutrophication rates (Section 7.0).

4.5.6 Sewage Trunk System (1990 to 1995)

Up until the early 1990s, most lakeshore residents were serviced by evaporative lagoons or mounded septic fields. As reported by Urban Systems Ltd. (undated report), these sewage disposal systems were "leaky" and, as such, they permitted untreated sewage to pond and flow within residential ditches (e.g., Residential Ditches #3 and #5) (Figure 1). In an effort to sanitize the runoff ditches and reduce the rate of contaminant loading to Charlie Lake, most (not all) lakeshore residences were connected to a "trunk" system that pipes domestic waste waters out of the Charlie Lake watershed.

The wastewater pipe was sampled by BCMELP on the following schedule:

Years 1990 to 1995

Site: Charlie Lake sewage trunk pipe

Schedule: Once in September, 1990; 2 occasions/month June to October, 1993; once in June and August, 1994; 2 occasions/month July, September and October, 1994; once in June and September, 1995; 2 occasions/month July and August, 1995

Parameters (*N*): BOD (10); flow (14); non-filterable residues (10); TP (5)

4.6 ANALYSIS OF LIMNOLOGICAL DATA

4.6.1 Nutrient Limitation (1974 to 1992 data)

Nutrient limitation (N and/or P) was assessed by comparing surface water (0.5- to 1.0-m depth) TN:TP ratios (*n* = 112) observed at nine lake sites (0400388, -389, -390, -391, -392, -393, -394, -395 and E207459) to typical cellular TN:TP ratios (Nordin 1985). Using this model, ratios of less than 5:1 were assumed to indicate N-limitation, those of 5-15:1 no or co-limitation, and those greater than 15:1 P-limitation. TN concentrations were determined by summing TKN and NO_2+NO_3 concentrations. TP concentrations were taken directly from the EMS database.

4.6.2 Effects of Watershed Development

The tributary streams were not sampled for nutrient chemistry when the watershed was in a forested state. Therefore, there is no direct way to determine how land clearing, and associated road building, fertilization and livestock husbandry, has affected the nutrient chemistry of the tributaries. Given the lack of long-term historical data, we had to use an inferential (*a posteriori*) approach to deduce how the nutrient chemistry (TP and TDP concentrations) of the tributary streams *may* have been affected by land clearing.

To test the hypothesis that TP and TDP concentrations in the tributary streams (mostly cleared watersheds) were different than those in a forested counterpart ("control" stream), TP and TDP concentrations observed in the tributary streams were compared to those observed in a forested tributary to Red Creek, located about 10 km northwest of the Charlie Lake watershed (Figure 1). The TP and TDP data were paired by sampling date to control for temporal hydrological and meteorological variations, and between-stream differences assessed with Wilcoxin paired-sample tests (Zar 1984, pp. 153-156). Significant differences ($P<0.05$) in TP and TDP concentrations observed between tributary and control streams were attributed to land-clearing affects under the assumption that between-stream differences in water chemistry were negligible when the watershed was in a forested state.

4.6.3 Phosphorus Budgets

The relative rates of internal versus external P-loading to Charlie Lake were assessed with two approaches: correlative and mass-balance.

Correlative Approach (1988 to 1993 Data)

The premise of this approach was that the P content of Charlie Lake (total mass of P within the water column) and P concentrations would be greatest during the hydrograph peak (maximum lake volume) if tributaries were the primary source of P (external loading). Conversely, it was reasoned that the P content of the lake would be greatest during periods of hypolimnetic anoxia (post-freshet, low lake volume) if the rate of internal P loading exceeded the rate of external P loading.

When the lake was thermally and chemically mixed (e.g., May 6, August 20 and September 15, 1992) (see Figures 2-4), the TP and TDP content of the lake was computed by multiplying the average concentration (from all sites and all depths) by lake volume; where lake volume was computed with equation 24 (Section 5.2). When P concentrations varied with depth, as occurred during periods of prolonged stratification (e.g., March 12, June 8 and July 14, 1992; Figures 2-4), the P content of the lake was computed by summing the P content of low-oxygen strata (hypolimnion) with the P content of aerobic strata (typically metalimnion and epilimnion). Hypolimnetic P content was estimated by multiplying the average TP and TDP concentrations observed in the strata by hypolimnetic volume. Hypolimnetic volume was determined by first estimating the vertical extent of oxygen depletion from chemical/thermal profiles (Figures 2-4); for example the extent of oxygen depletion was from the bottom to about 7.5-m depth on March 12, 1992 (Figure 2b) and from the bottom to about 7.0-m depth on March 17, 1993 (Figure 3b). The depth of the upper-most extent of the low-oxygen zone was then converted to a geodetic elevation by subtracting the depth from the geodetic elevation of the lake's surface on the respective sampling date (lake surface elevation measured daily). Volume was then computed with one of equations 24-28. The P content of the aerobic strata was determined by multiplying the average TP and TDP concentrations by volume; where volume was computed by subtracting hypolimnetic volume from total lake volume.

The relationship between hydrograph stage (as lake volume) and lake P content/concentration was described from plot overlays of lake volume and lake P content/concentration versus date versus date (January to December).

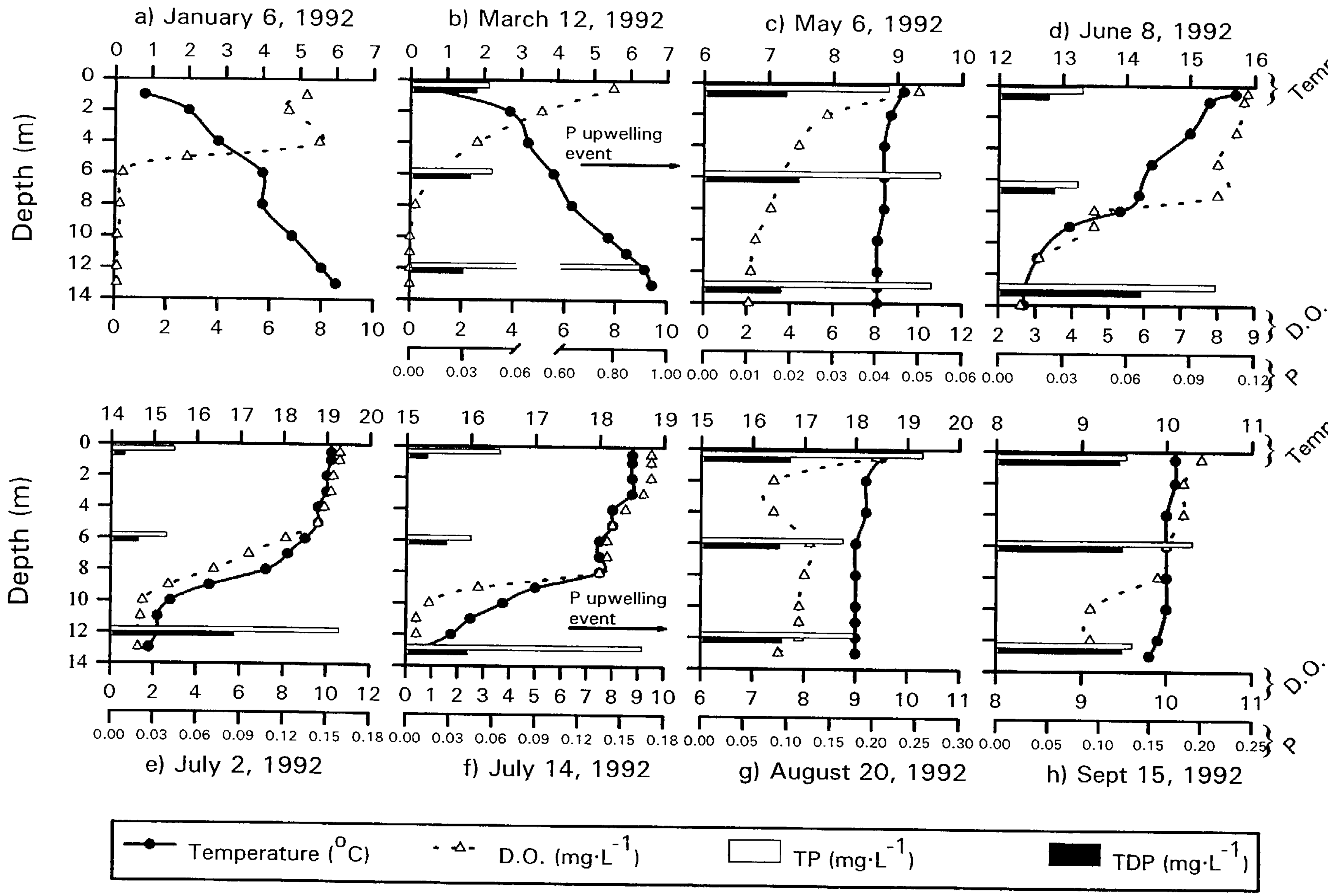

FIGURE 2. TEMPERATURE, D.O., TP AND TDP PROFILES OBSERVED IN 1992 (SOUTH DEEP STATION, EMS SITE 0400390). SPRING (MARCH 12) AND LATE-SUMMER (JULY 14) P UPWELLING EVENTS MARKED WITH ARROWS.

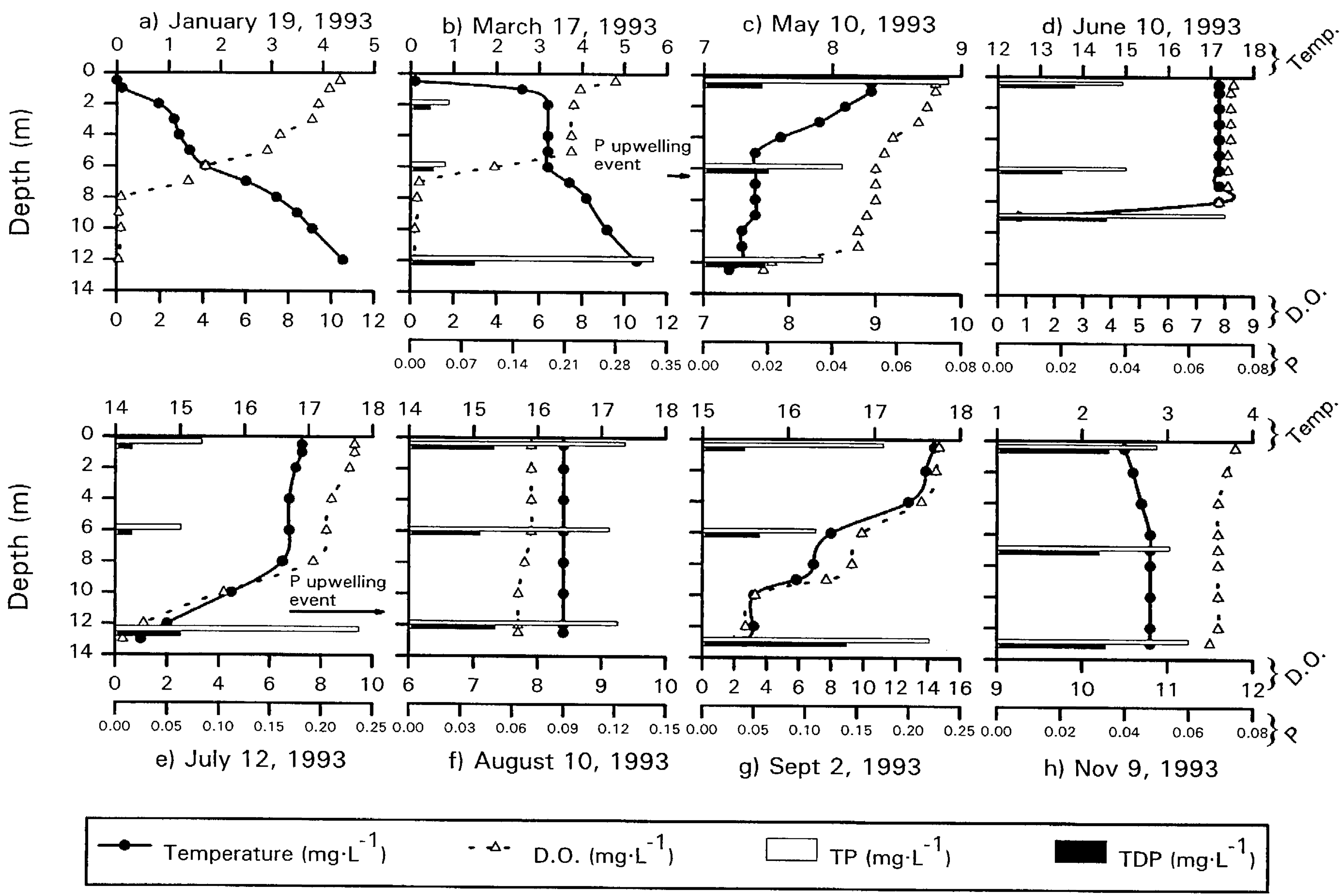

FIGURE 3. TEMPERATURE, D.O., TP AND TDP PROFILES OBSERVED IN 1993 (SOUTH DEEP STATION, EMS SITE 0400390). SPRING (MARCH 17) AND SUMMER (JULY 12) P UPWELLING EVENTS MARKED WITH ARROWS.

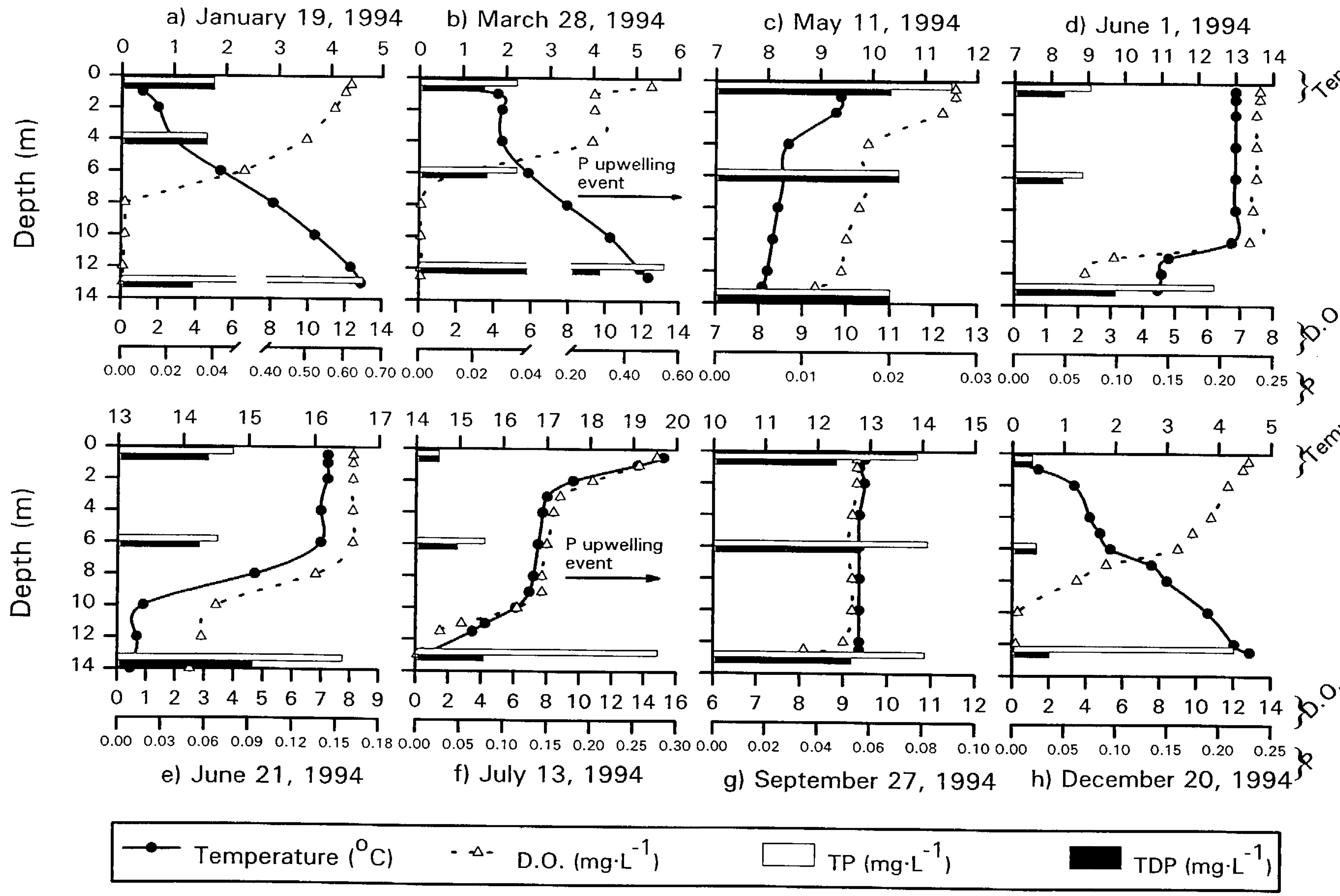

FIGURE 4. TEMPERATURE, D.O., TP AND TDP PROFILES OBSERVED IN 1994 (SOUTH DEEP STATION, EMS SITE 0400390). SPRING (MARCH 28) AND SUMMER (JULY 13) P UPWELLING EVENTS MARKED WITH ARROWS.

Mass-Balance Approach (1988 to 1991 Data)

In this approach, external P loading (tributaries plus groundwater) and external P export (Lower Stoddart Creek and the Anderson Exploration Ltd. and City of Fort St. John withdrawals) rate estimates were used to algebraically compute the relative contribution of internal versus external P dynamics to the total P budget of the lake. Since allochthonous P is carried to lakes with inflowing waters, our first step was to estimate a water budget. The volume of water exported via Lower Stoddart Creek during each sampling period (time between two sampling dates) was calculated from daily discharge data (continual gauging) obtained from Environment Canada (1994). Evaporation losses were estimated as described in Section 4.3. Water loss through major water withdrawals was estimated from daily (City of Fort St. John) and monthly (Anderson Exploration Ltd.) measures of in-pipe flow volumes. Rainwater volumes deposited on the surface of Charlie Lake were estimated, on a daily basis, by multiplying rain quantities (measured in mm and converted to m) by A_o (m^2). Rainwater quantities were measured (by Environment Canada staff) with an MSC rain gauge (graduated in mm; 35-cm opening) positioned near the City of Fort St. John. Tributary and groundwater inflows were not measured on a regular basis; thus, we had to algebraically compute tributary and groundwater inputs as a combined flow. The first step in the computation of tributary plus groundwater inputs was to determine the total water input (TWI) to the lake over each sampling period with the equation:

$$TWI = \Delta S + \Sigma outflows \qquad (15)$$

where ΔS is the change in lake volume over the sampling period of interest and $\Sigma outflows$ is the sum of Lower Stoddart Creek, evaporation and major withdrawal flows over the period of interest. ΔS was computed with the equation:

$$\Delta S = Vt1 - Vt2 \qquad (16)$$

where Vt1 is the lake volume at the beginning of the sampling period of interest and Vt2 is the volume of the lake at the end of the time period of interest. Vt1 and Vt2 were computed with one of equations 24 to 28, depending on the geodetic elevation of the lake's surface. Tributary plus groundwater inflows for each sampling period were then computed by subtracting precipitation inputs from TWI.

Once the water budget was established, P loads and exports were estimated by multiplying flow volumes by P concentration. Readers should be aware that the P budget computed by the mass-balance approach is only a coarse estimate because of the limited availability of P concentration data. P concentrations in rainwater and snow were assumed to be 30 and 5 $\mu g \cdot L^{-1}$, respectively (derived from Shaw *et al.* (1989) and Wetzel (1983, p. 280)). A fallout rate of 70 $\mu g\ P \cdot m^{-2} \cdot d^{-1}$ (Shaw *et al.* 1989) was used to estimate P loads from dry atmospheric sources (e.g., dust, pollen, aerosols and microscopic biota). Median P concentrations observed at the surface of site 0400388 (Figure 1) and in Lower Stoddart Creek were used to compute P exports via Lower Stoddart Creek: 79 $\mu g \cdot L^{-1}$, January to April; 47 $\mu g \cdot L^{-1}$, May to June; and 116 $\mu g \cdot L^{-1}$, July to December (monthly trends are not statistically significant). Annual median TP concentrations of 80 $\mu g \cdot L^{-1}$ (City of Fort St. John) and 67 $\mu g \cdot L^{-1}$ (Anderson Exploration Ltd.) were used to estimate P exports by major water withdrawals. A total of 410 TP measurements were available for all tributaries combined. P contributed to the lake by tributaries plus groundwater was estimated under three scenarios to assess the range of loading possibilities: tributary and groundwater P concentrations = 0.12 $mg \cdot L^{-1}$ (25^{th} percentile, Scenario 1); tributary and groundwater P concentrations = 0.19 $mg \cdot L^{-1}$ (50^{th} percentile, Scenario 2); and tributary and groundwater P concentrations = 0.36 $mg \cdot L^{-1}$ (75^{th} percentile, Scenario 3). Internal P consumption and internal P loading rates were computed algebraically under each of scenarios 1 to 3. It was reasoned that:

$$\text{Tributary plus groundwater P loads + atmospheric P loads - outlet P loads} \quad (17)$$

would equal, or at least approximate, ΔP (change in lake P content over any given sampling period, where lake P content is computed as described in "Correlative Approach", above) if internal (within-lake) processes did not contribute P to the water column (internal P loading) or remove P (internal P consumption) from the water column. The lake was classified as being either a "net P producer" (P released to the water column faster than it is removed) or a "net P consumer" (P removed from the water column faster that it is released) over each sampling period with the model:

$$\text{Net P consumer if } \Delta P < \text{tributary plus groundwater P loads + atmospheric P loads - outlet P loads} \quad (18)$$

$$\text{Net P producer if } \Delta P > \text{tributary plus groundwater P loads + atmospheric P loads - outlet P loads} \quad (19)$$

When the lake was a net P producer, the total P input (TPI) to the lake was computed as:

$$TPI = \Delta P + \Sigma \text{outlet P load} \quad (20)$$

and internal P load (IPL) by:

$$IPL = TPI - \text{tributary plus groundwater loads + atmospheric P loads} \quad (21)$$

When the lake was a net P consumer, TPI was computed by summing tributary and atmospheric loads. Internal P consumption rates (IPC) were estimated by subtracting outlet loads from the total P output (TPO), where TPO was estimated by:

$$TPO = \text{tributary plus groundwater loads + atmospheric loads} - \Delta P \quad (22)$$

4.6.4 Correlates of Algal Abundance

One of the most widely cited algal correlates is that developed by Dillon and Rigler (1974). They showed that average summer algal biomass (measured as chlorophyll *a* concentration) in Precambrian Shield lakes is tightly linked to spring overturn TP concentrations, such that:

$$\text{Chlorophyll } a = 1.449 \times TP - 1.136 \quad (23)$$
$$n = 46;\ r^2 = 0.95$$

where chlorophyll *a* and TP concentrations are log $\mu g \cdot L^{-1}$. This relationship was subsequently found to hold for lakes off the Precambrian Shield as well (e.g., Prepas and Trew 1983). The spring overturn TP and average summer chlorophyll *a* concentrations observed (surface grab samples) in Charlie Lake (1988, 1989 and 1991 to 1994 data) were plotted with spring-overturn TP-chlorophyll *a* data extracted from Dillon and Rigler (1974) and Prepas and Trew (1983) to test the hypothesis that algal biomass in Charlie Lake is similar to that predicted by the Dillon-Rigler spring-overturn P concept.

The premise of the Dillon-Rigler model is that summer algal production is controlled by spring overturn TP concentrations. Although the model predicts summer algal biomass reliably, it does not consider within-lake variations that occur after spring overturn, even though these variations may be more influential than spring-overturn conditions. We plotted Charlie Lake chlorophyll *a* concentrations with surface TP and TDP concentrations and water temperature (1-m depth samples) to test the hypothesis that post-overturn variations in algal biomass are related to variations in nutrient availability and temperature. Under the assumptions that algal production in the lakes is limited by P availability and that metabolic reaction rates increase with increasing thermal energy (at least over the range observed in the lake), we predicted that algal biomass would be positively correlated with post-overturn P concentration and temperature. While it can be safely assumed that post-overturn algal production is functionally dependent on nutrient availability and, perhaps, thermal energy, we, conversely, hypothesized that algal production should increase and decrease pH (photosynthesis consumes carbon) and water clarity, respectively (i.e., pH and water clarity may be, at least partially, functionally dependent upon algal production). We plotted 1997-1998 chlorophyll *a* concentrations against concurrent measures of pH and water clarity to see if changes in these factors tracked changes in algal biomass.

4.6.5 Water Quality Criteria (Provincial Standards)

To assess the overall "quality" of tributary and lake waters, the data were compared to drinking, recreational, aquatic life, irrigation, livestock watering and food processing (dairy processing, food washing, etc.) standards set by BCMELP (Nagpal *et al.* 1995). The tributary ammonia data could not be directly compared to BCMELP standards because temperature and pH were not measured during the 1989 to 1993 stream sampling program (i.e., ammonia standards are temperature- and pH-dependent). We compared the tributary ammonia data to the standards assuming that the pH of tributary waters fluctuated between 7.2 and 7.9 (values observed in Stoddart Creek, 1982 to 1988) and that temperatures fluctuated between 0 and 5 °C in the autumn, winter and spring, and between 5 and 15 °C in the summer (worst-case scenarios). Lake ammonia data were paired with temperature and pH data prior to comparing them to the standards (392 pairs in all). Spring overturn TP concentrations observed in the lake were compared to the standards for the years 1988, 1989 and 1991 to 1994 (i.e., TP standards are based on spring-overturn concentrations for lakes having epilimnetic residence rates > six months). Dissolved oxygen concentrations were compared to standards set for "non-salmonid" waters.

4.6.6 Nutrient Export by Major Water Withdrawals

As stated in Section 3.5, the City of Fort St. John withdrew water from the 689.92-m geodetic (~2.2-m water depth). Anderson Exploration Ltd., by comparison, withdraws water from a deeper stratum, taking water from the 687.20-m geodetic (~4.7-m water depth). Although both withdrawals can be considered to be "deep-water" withdrawals in that the intake ports are located near the bottom, they are, in a functional sense, "shallow-water" withdrawals in that they take water from depths 3 m or more above the upper-most extent of the 8- to 15-m depth hypolimnion (Figures 2-4).

There is concern that the withdrawals may be concentrating P in the lake and, thereby, contributing to the perceived algal problem. To address this issue, we estimated the total mass (kg) of P removed from the lake on a monthly basis (City of Fort St. John, 1992 to 1994; Anderson Exploration Ltd., 1992 to 1994) by multiplying average monthly withdrawal volumes (Table 10) by the average monthly TP concentration observed at the surface or bottom of site 0400390 (Table 11). P export (kg per month) by the water withdrawals was computed under the scenario that the intake ports were located above the depth hypolimnion formation (actual situation; surface TP concentrations used) and the scenario that the intake ports were located within the zone of hypolimnion formation (hypothetical situation; near-bottom TP concentrations used) to compare the P-export rates of the surface withdrawal to a hypothetical deep-water withdrawal.

The P export rates were used to quantify the withdrawals' monthly affect on lake water P concentration. This computation required that we first estimate average lake volume for each month from daily measures of lake surface elevation (equations 24-28; Table 11). Monthly withdrawal volumes (Table 10) were then subtracted from average lake volume to estimate lake volume after the monthly withdrawal. The average total mass of P in the lake for each month was then estimated from measures of P concentration at site 0400390 using an algorithm similar to that described in Section 4.6.3 (Table 11). This mass was then converted to an average lake concentration, under the assumption of complete mixing. The mass of P in the lake after each monthly withdrawal was then estimated by subtracting the P export mass from the total mass of P in the lake; this mass was then converted to an average lake concentration using the estimate of lake volume after withdrawal. The change in P concentration resulting from the withdrawals was then computed by subtracting the average lake P concentration after withdrawal from the average lake P concentration before withdrawal.

TABLE 10. CITY OF FORT ST. JOHN (1992 TO 1994) AND ANDERSON EXPLORATION LTD. (1992) AVERAGE MONTHLY WITHDRAWAL VOLUMES.

Month	Average monthly withdrawal volume (dam^3, (*n*))	
	City of Fort St. John	Anderson Exploration Ltd.
January	207 (93)	69 (4)
February	194 (84)	76 (4)
March	204 (93)	86 (4)
April	194 (90)	77 (4)
May	216 (93)	78 (4)
June	215 (90)	76 (4)
July	209 (93)	72 (4)
August	205 (93)	70 (4)
September	195 (90)	62 (4)
October	196 (93)	66 (4)
November	190 (90)	52 (4)
December	194 (93)	61 (4)

TABLE 11. AVERAGE LAKE VOLUME (1991 TO 1995), TP CONCENTRATION (1984 TO 1995) AND LAKE P CONTENT (1988 TO 1995) FOR EACH MONTH.

Month	Average lake volume, 1991 to 1995 (dam^3, (*n*))	Average TP concentration ($mg \cdot L^{-1}$, (*n*))		Average lake P content	
		Near surface	Near bottom	Total mass (kg, (*n*))	Average concentration ($mg \cdot L^{-1}$, (*n*))[a]
January	109,436 (155)	0.084 (4)	0.524 (4)	10,649 (4)	0.097 (4)
February	109,566 (140)	0.048 (1)	0.274 (1)	7,203 (1)	0.066 (1)
March	112,906 (155)	0.036 (6)	0.343 (6)	7,597 (6)	0.067 (6)
April	114,508 (150)	0.041 (3)	0.193 (3)	6,126 (3)	0.053 (3)
May	115,393 (155)	0.052 (8)	0.058 (8)	6,429 (8)	0.056 (8)
June	114,430 (150)	0.054 (9)	0.121 (9)	9,851 (9)	0.086 (9)
July	113,135 (155)	0.063 (8)	0.192 (8)	9,408 (8)	0.083 (8)
August	111,859 (155)	0.180 (6)	0.178 (6)	17,388 (6)	0.155 (6)
September	110,058 (150)	0.103 (7)	0.121 (7)	14,023 (7)	0.127 (7)
October	109,558 (155)	0.074 (4)	0.068 (4)	7,615 (4)	0.070 (4)
November	109,438 (150)	0.050 (1)	0.060 (1)	6,973 (1)	0.064 (1)
December	109,404 (155)	0.016 (1)	0.215 (1)	8,111 (1)	0.074 (1)

[a] average P concentration in lake, assuming complete mixing.

4.6.7 Nutrient Export by Sewage Trunk System

Analyses of runoff water collected from Residential Ditch # 3 suggest that the residential ditches were loading Charlie Lake with, among other things, fecal coliforms (<2-1500 CFU·cL^{-1}; $\overline{X}$ = 202 CFU·cL^{-1}), P (89-977 μg·L^{-1} TP; $\overline{X}$ = 334 μg·L^{-1} TP) and NH_4-N (6-22 μg·L^{-1}; $\overline{X}$ = 14 μg·L^{-1}). To quantify the rate at which the trunk system removes N and P from the watershed, in-pipe flow volumes were measured periodically in conjunction with analytical measures of TP and TN; and export computed as the product of median concentration and flow volume.

5.0 RESULTS AND DISCUSSION

5.1 REVIEW OF EXISTING DOCUMENTATION

5.1.1 Public Interests and Concerns (PRRD 1996)

Summer fishing (46.5/212 points), motorized pleasure boating (39.5/212 points) and residential living (25.5/212) were the top-three public uses of Charlie Lake (Table 12a). By comparison, x-country skiing (5.5/212 points), seadoing (3.5/212 points), water skiing (2/212 points) and scuba diving (1.5/212 points) were the lowest-ranking activities. The public ranked algal abundance (eutrophication) as their greatest environmental concern (79/194 points) (Plates 9-12), with fishing quality (41/194 points) and lakeshore development (23/194 points) ranking second and third, respectively (Table 12b).

Given the noted public interests and concerns, any environmental programs implemented to "improve" Charlie Lake should focus on reducing nutrient loading rates and controlling the nature and amount of near-shore residential development. Nutrient load reductions should, over time, decrease the intensity of summer algal blooms and, in turn, increase the aesthetic appeal of the lake to recreational and residential users. Reducing nutrient loading rates may also reduce hypolimnetic and under-ice oxygen consumption rates, decreasing the potential for summer and winter fish mortality.

5.1.2 Agricultural Pesticide Residues (BCMELP 1992)

BCMELP (1992) concluded that there were no measurable organophosphate, organochlorine or solvent soluble pesticide residues in walleye flesh, lake-bottom sediments or Stoddart Creek water. Readers should keep in mind that this conclusion was based on chemical analyses made on a very small number of samples (Section 4.1.2), so it may not be completely reliable.

5.1.3 Fish Parasites (Bangham and Adams 1954)

Twenty seven of the 28 white suckers examined had parasites, with six individuals being infected with *Diplostomulum* sp. (Trematoda – the "flukeworms), two with *Eustrongylides* sp. (Nematoda, the "roundworms"), 15 with *Glaridacris catostomi* (Cestoda, the "tapeworms"), five with *Myxosporidia* (Protozoa, the "unicells") and 18 with *Pomphorhynchus bulbocolli* (Acanthocephala, a copepod parasite). By comparison, parasites were observed in less than half of the examined brook sticklebacks: four fish had *Bunoderina eucaliae* (Trematoda), one had Gyrodactyloidea (Trematoda), three had *Schistocephalus* (Cestoda) and three had *Tetracotyle* (Trematoda). While it is clear that white suckers had a higher infection rate than brook sticklebacks, it is also notable that white suckers (Trematoda, Nematoda, Cestoda, Protozoa and Acanthocephala) supported a considerably more diverse parasite community than brook sticklebacks (only Trematoda and Cestoda).

TABLE 12. RESULTS OF QUESTIONNAIRE USED TO SURVEY LOCAL INTERESTS AND CONCERNS REGARDING THE USE AND "QUALITY" OF CHARLIE LAKE (PEACE RIVER REGIONAL DISTRICT, 1996, UNPUBL. DATA).

Parameter	Total awarded points	Overall ranking of community relevance
a) Greatest activity interests		
Summer fishing	46.5	1
Motorized pleasure boating	39.5	2
Residential living	25.5	3
Family picnics	25	4
Ice fishing	17	5
Swimming	16.5	6
Wildlife and bird viewing	14	7
Non-motorized pleasure boating	9	8
Snowmobiling	7	9
X-country skiing	5.5	10
Seadoing	3.5	11
Water Skiing	2	12
Scuba diving	1.5	13
TOTAL POINTS	212	na
b) Greatest concerns		
Algal abundance	79	1
Quality of fishing	41	2
Lakeshore development	23	3
Destruction of wildlife and bird habitat	20	4
Noise levels from weekend recreational users	16	5
Harassment of wildlife and birds by pets	9	6
Inconsiderate boaters and watercraft users	6	7
TOTAL POINTS	194	na

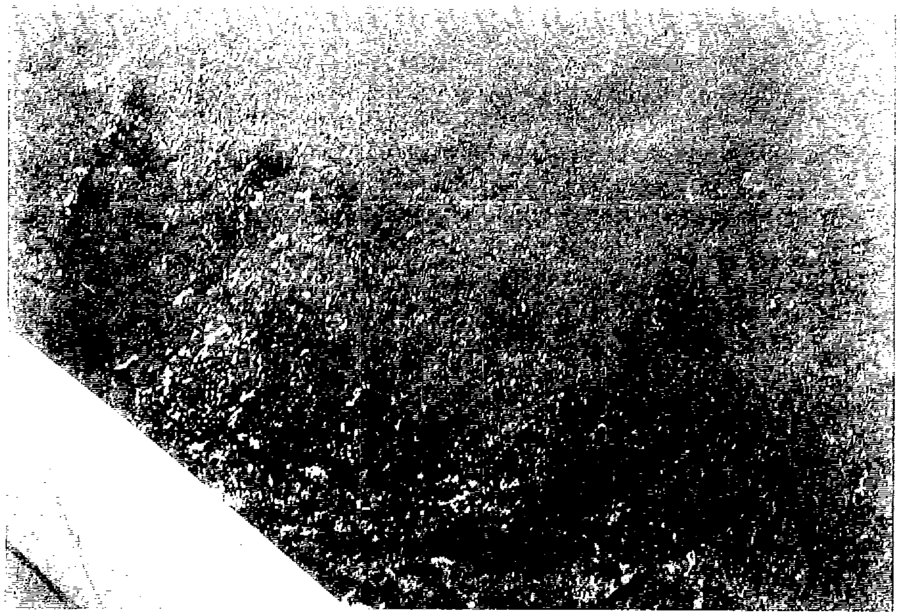

Plate 9. Thick algal scum on Charlie Lake, 1974.

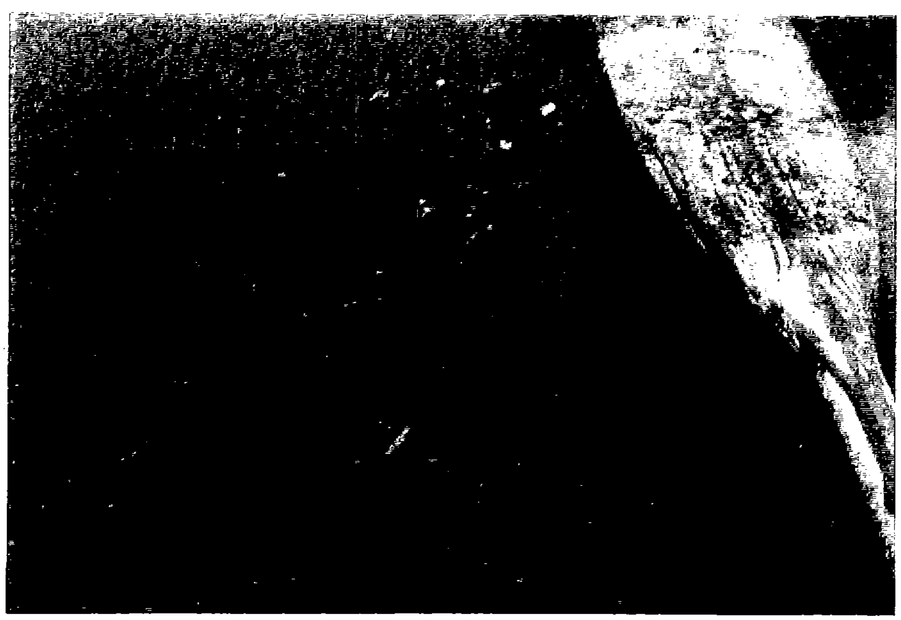

Plate 10. Algal scum on Charlie Lake, 1974. Note green stain on paddle after it was dipped in the water.

Plate 11. Blue-green scum washed into shoreline embayment in early autumn, 1996.

Plate 12. Blue-green scum in Charlie Lake embayment area, autumn 1996

5.1.4 Paleolimnological Investigations (Reavie *et al.* 1995b)

Aulacoseira ambigua, *Fragilaria pinnata* and three *Stephanodiscus* species were the most common taxa in pre-1900 sediment, suggesting that Charlie Lake is naturally eutrophic. The abundance of the eutrophic indicator species *Stephanodiscus hantzschii* increased strikingly between the mid 1900s and 1991, and that of *Stephanodiscus parvus* slightly. While the abundance of *S. hantzschii* and *S. parvus* increased, the *F. pinnata* population shrank from a relative abundance of 48% to 7% over the time period.

The increase in *Stephanodiscus* spp. abundance starting in the early 1900s, suggests that human settlement, and associated land clearing and farming, hastened eutrophication rates (see Osborne and Moss 1977; Andersen *et al.* 1990). In further support of this conclusion, inferred measures of TP concentration suggest that TP concentrations have increased from an average of 22 to 60 $\mu g \cdot L^{-1}$ (almost 3-fold) since the end of the 19th century.

5.1.5 Preliminary Limnological Investigations (Nordin and Pommen 1985)

Nordin and Pommen (1985) drew a number of preliminary conclusions from their limited database, including:

<u>Stoddart Creek inflow</u>

(1) average organic C, ammonia N and P concentrations in Stoddart Creek were greater than those observed in Charlie Lake, suggesting that tributary waters concentrate the lake water, as opposed to diluting it;
(2) average specific conductance and turbidity values in Stoddart Creek were greater than those in Charlie Lake, indicating that tributary waters carry more dissolved and particulate matter than the lake water; and
(3) agricultural practices in the watershed have likely affected the chemistry and particulate load of Stoddart Creek.

Charlie Lake

(1) the overall water quality of Charlie Lake was considered to be "very poor"; the lake was in an advanced state of eutrophication;

(2) in terms of algal biomass ($\overline{X}$ =31.4 $\mu g \cdot L^{-1}$ chlorophyll *a*) and nutrient chemistry ($\overline{X}$ =0.078 $mg \cdot L^{-1}$ ammonia N, 1.09 $mg \cdot L^{-1}$ organic N and 0.096 $mg \cdot L^{-1}$ TP), Charlie Lake was one of the most eutrophic lakes in BC (see Figure 5 for a comparison of several BC lakes);

(3) Charlie Lake had low water clarity ($\overline{X}$ =1.5 m Secchi depth), and high colour ($\overline{X}$ =27.5 TC and 21.9 TAC) and turbidity ($\overline{X}$ =4.6 NTU);

(4) Cyanophytes (blue-green algae) dominated the algal community;

(5) the average chlorophyll *a* concentration observed in Charlie Lake (31.4 $\mu g \cdot L^{-1}$) far exceeded the Province of Ontario's recreational criteria of 5 $\mu g \cdot L^{-1}$, suggesting that Charlie Lake may not be suitable for contact-based recreation;

(6) based on an observed spring-overturn N:P ratio of 15.6 (1.03 $mg \cdot L^{-1}$ N and 0.066 $mg \cdot L^{-1}$ P) , primary production in Charlie Lake was likely limited by P availability;

(7) Charlie Lake is typically isothermal (little or no stable stratification observed); however, if stratification were to persist for more than a few days, there could be significant hypolimnetic D.O. consumption rates;

(8) the median coliform concentration (not specified whether concentrations refer to total coliform count or fecal coliform count) at the lake outlet was 452/100 mL^{-1} in 1971, 135/100 mL in 1972 and 95/100 mL in 1973; and 48/100 mL at the City of Fort St. John water intake in 1973;

(9) in terms of Provincial fecal coliform criteria, Charlie Lake was suitable for recreational use; however, waters withdrawn for domestic consumption require disinfection and some treatment; and

(10) the primary water quality concerns for Charlie Lake and Stoddart Creek are eutrophication, fecal contamination and possibly insecticide and herbicide pollution;

Based on their observations and above-noted conclusions, Nordin and Pommen (1985) gave several monitoring recommendations. They are listed in Section 7.0 of the present report.

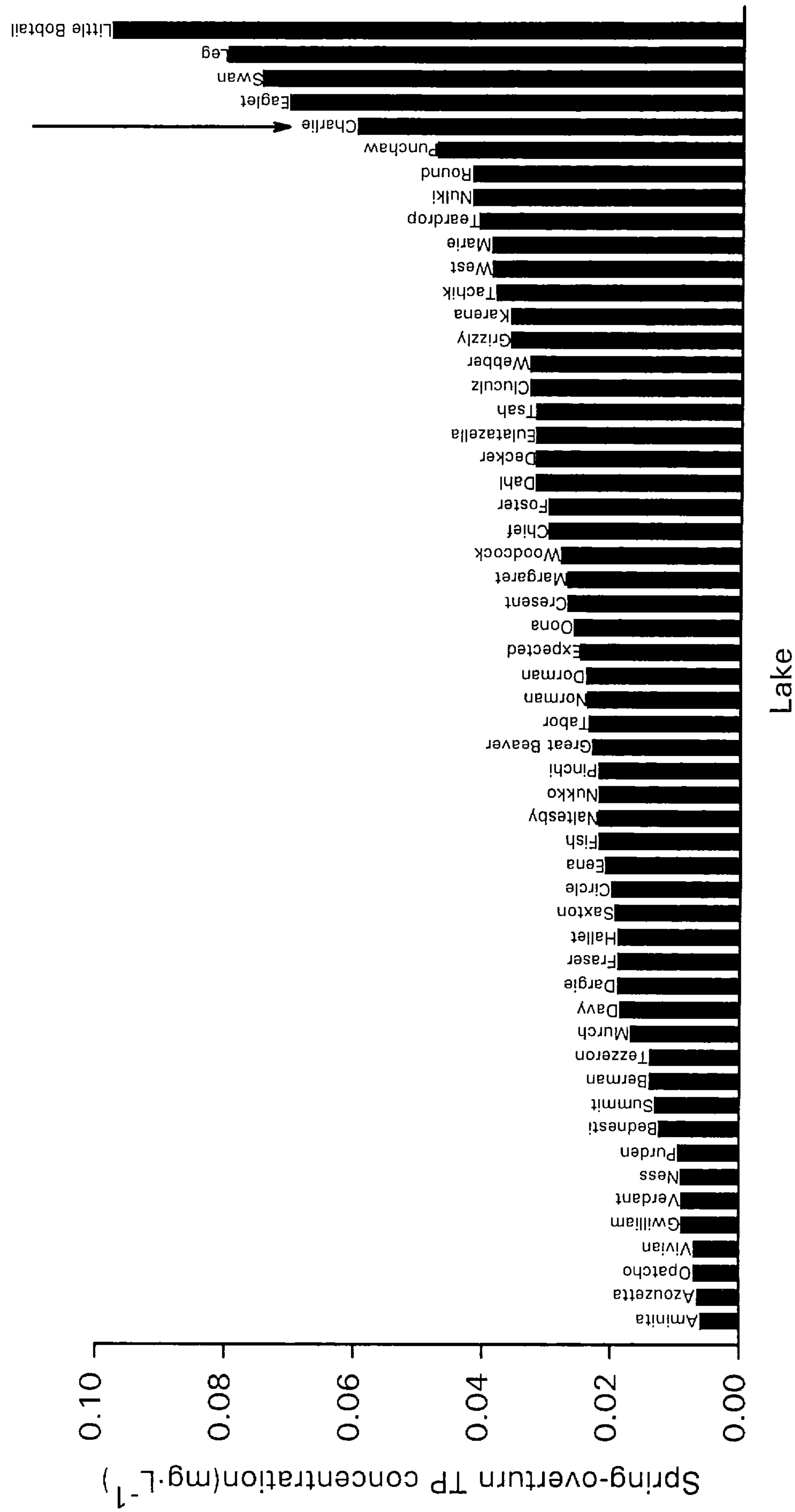

FIGURE 5. SPRING-OVERTURN TP CONCENTRATIONS IN SEVERAL NORTHERN BRITISH COLUMBIA LAKES (BCMELP UNPUBL. DATA). CHARLIE LAKE MARKED WITH AN ARROW.

5.2 MORPHOMETRY

The water volume (storage, *S*) below any given depth contour (geodetic elevation, *GE*) can be computed with the following set of regression equations (Figure 6):

When *GE* is > 686 m $S\,[dam^3] = 15{,}133.9 \times GE\,[m] - 10{,}357{,}379.3$ (24)
When *GE* is 684-686 m $S\,[dam^3] = 8{,}700.5 \times GE\,[m] - 5{,}940{,}562.6$ (25)
When *GE* is 682-684 m $S\,[dam^3] = 4{,}161.8 \times GE\,[m] - 2{,}836{,}091.8$ (26)
When *GE* is 680-682 m $S\,[dam^3] = 1{,}023.1 \times GE\,[m] - 695{,}464.3$ (27)
When *GE* is 677-680 m $S\,[dam^3] = 69.9 \times GE\,[m] - 47{,}322.3$ (28)

Having a maximum length of 15 km, a shoreline length of 38 km, a surface area of 19 km^2 and a volume of 136,839 dam^3, Charlie Lake is considered to be a medium-sized lake. As are most naturally eutrophic lakes of glacial origin (Rawson 1955; Hutchinson 1957), Charlie Lake is shallow ($\bar{Z}$ = 7 m; Z_m = 15 m). In reference to values given in Wetzel and Likens (1991), the relative depth of 0.3 suggests that the lake has little resistance to vertical mixing. Charlie Lake drains an area of about 281 km^2 (A_o not included).

5.3 BASIC HYDROLOGY

Charlie Lake has 21 tributaries (Table 13). Ground surveys conducted by the authors and others indicate that most of these tributaries are ephemeral. In addition to these tributaries, there are several drainage ditches running through the southwestern shoreline community. These ditches carry water only during rain and snow-melt events. The two largest tributaries (Stoddart Creek, 171 km^2; Coffee Creek, 25 km^2) enter the lake at its north end and, together, drain nearly 80% of the watershed. Charlie Lake is drained at its southernmost tip by Lower Stoddart Creek. After leaving Charlie Lake, Lower Stoddart Creek flows for about 20 km to where it joins the Beatton River. The Beatton River drains into the Peace River, which flows northeastwardly across northern Alberta to where it pours into Lake Athabasca (Figure 1).

According to our modeling results, Charlie Lake loses about 12,038 dam^3 of water annually via evaporative processes (Table 14). By comparison, UMA Engineering Ltd. (1991) estimated that the lake's annual evaporative loss averages about 9,449 dam^3 (Table 14). Given that our evaporative-loss estimate is fairly close to that calculated by UMA Engineering Ltd. (i.e., on an annual basis they differ by only 22%) and that these estimates were arrived at using independent approaches, our modeled evaporation rate may be a reasonable "ball park" estimate of the true evaporation rate. The modeled values suggest that evaporation does not occur when the lake has a solid ice-covering (November to February). Evaporation rates are greatest during the month of July, when air temperatures are highest and cloud cover lowest (Table 3).

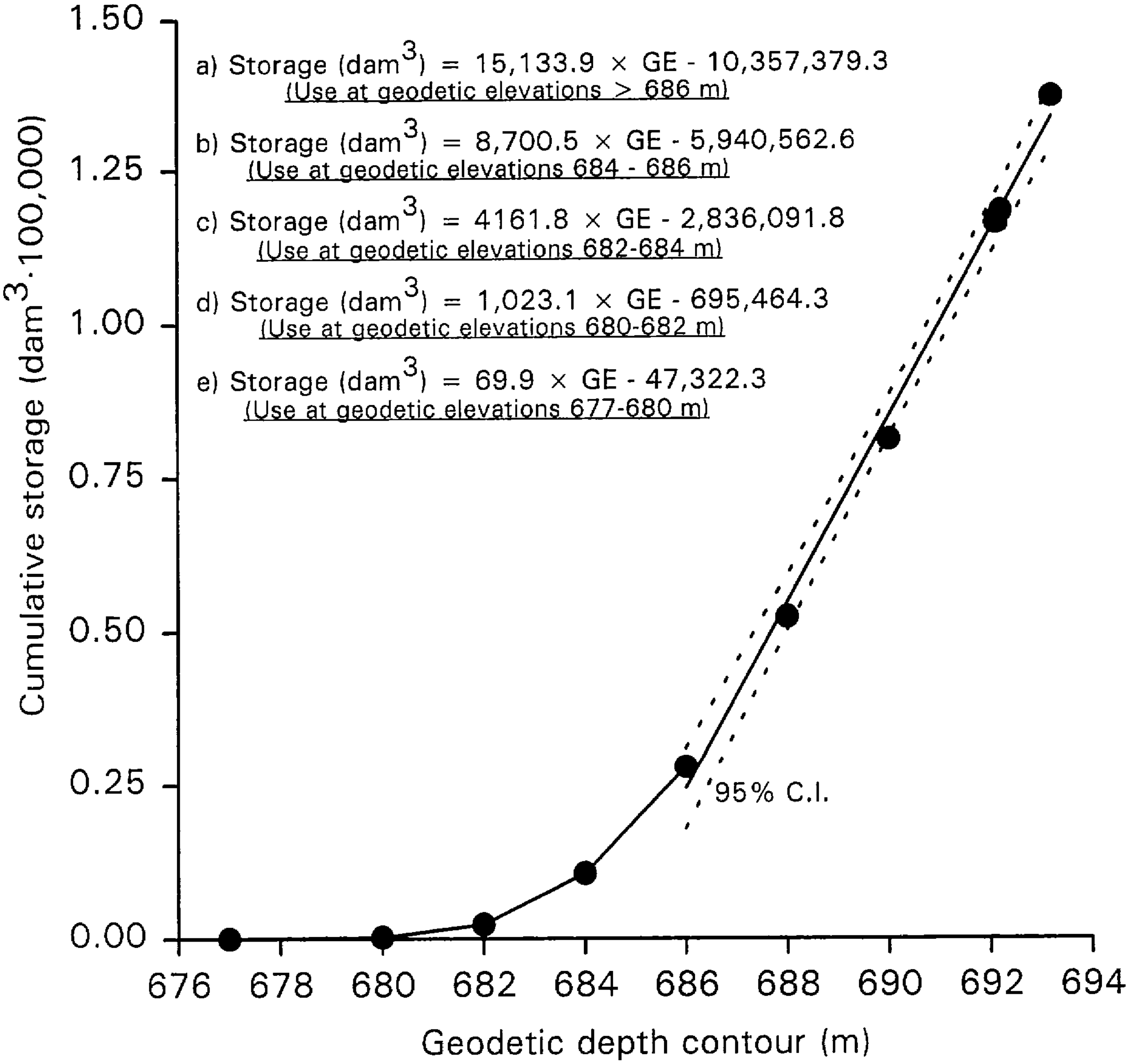

FIGURE 6. RELATIONSHIP BETWEEN DEPTH-CONTOUR ELEVATION (GEODETIC) AND STORAGE (LAKE VOLUME). GE IS "GEODETIC ELEVATION".

TABLE 13. TRIBUTARIES TO CHARLIE LAKE. A TRIBUTARY NUMBER WAS ASSIGNED TO EACH TRIBUTARY AS COUNTED IN A CLOCKWISE DIRECTION AROUND THE LAKE STARTING AT THE NORTHERN-MOST INFLOW (STODDART CREEK). TRIBUTARIES INCLUDE ONLY THOSE THAT WERE RESOLVED BY 1:50,000 N.T.S. MAPS 94 A/7 (NORTH PINE) AND 94 A/6 (BEAR FLAT).

Tributary No.	Local name (if one)	Sub-basin area (km^2)
1	Stoddart Creek	171
2	not named	2.3
3	not named	6.8
4	not named	0.8
5	not named	8.8
6	not named	1.5
7	not named	0.7
8	not named	11.5
9	Beatton Park Creek	7.0
10	not named	1.8
11	not named	1.0
12	Boat Launch Creek	0.3
13	not named	0.1
14	not named	0.8
15	not named	0.4
16	not named	0.9
17	not named	1.0
18	not named	0.5
19	not named	6.5
20	not named	3.7
21	Coffee Creek	25
	TOTAL	approx. 252

TABLE 14. CHARLIE LAKE'S MONTHLY AND ANNUAL EVAPORATIVE LOSSES AS ESTIMATED WITH OUR MODELING APPROACH AND BY UMA ENGINEERING LTD. (1991).

Month	Total evaporation (mm)		Total evaporation (dam^3)	
	Modeled	UMA Engineering Ltd. (1991)	Modeled	UMA Engineering Ltd. (1991)
January	0	na	0	0
February	0	na	0	0
March	14.1	na	267.9	0
April	54.4	na	1,033.6	0
May	114.8	na	2,181.2	775.5
June	135.6	na	2,576.4	2,701.0
July	148.7	na	2,825.3	2,178.8
August	111.4	na	2,116.6	1,930.4
September	46.1	na	875.9	1,210.0
October	8.5	na	161.5	433.2
November	0	na	0	220.2
December	0	na	0	0
TOTAL	633.6	na	12,038.4	9,449.1

Very little water leaves Charlie Lake via Lower Stoddart Creek during the winter months (Figure 7). Spring flows to Lower Stoddart Creek typically begin at the end of March, with peak flows occurring between the end of April and the middle of May. While flows in Lower Stoddart Creek typically crest at 1 $m^3 \cdot s^{-1}$, they can get as high as 5 $m^3 \cdot s^{-1}$ (Figure 7). During dry years, Lower Stoddart Creek may crest at only 0.09 $m^3 \cdot s^{-1}$. Spring flows typically begin to subside during the latter half of May. However, during wet years (e.g., 1990) flows can remain high until the end of August (Figure 7). As stated in Section 4.3, the discharge data used in our analyses were based on a correlation between lake level and discharge. Thus, in order for the data to be accurate there must be a tight relationship between lake level (stage) and discharge. Indeed, lake level and discharge were tightly correlated in 1988, 1990 and 1991, suggesting that the discharge data are reasonably accurate for these years (Figure 8a,c,d). However, there was a very poor relationship, if any relationship at all, between lake level and discharge in 1989 (Figure 8b). Therefore, there is reason to believe that the 1989 discharge data are unreliable.

Basal lake-levels typically occur during the autumn and winter months (September to March), suggesting that inflow and outflow rates are approximately equal during these seasons (Figure 9). Inflow rates typically begin to exceed outflow rates at the end of March, as indicated by rising lake levels. Lake levels typically increase until the end of May or the middle of June, at which time the outflow rates begin to exceed the inflow rates and lake levels, consequently, begin their decline to basal levels.

As shown on panel m of Figure 9, peak lake levels appear to have increased by about 1.2 m following the construction of the new weir in 1981. UMA Engineering Ltd. (1991, p. 2.1), however, reported that the newly constructed weir did not increase peak lake levels. Thus, there is an apparent disagreement between the lake-level data and UMA Engineering Ltd.'s claims. In an attempt to resolve this disagreement we contacted officials from Environment Canada (Water Survey Division) and the City of Fort St. John (Public Works) and questioned them about this apparent rise in lake level. We were informed that the water level gauge was relocated on a couple of occasions in the early 1980s. We then asked if the gauge levels were "tied in" to one another following the relocations. Nobody could answer this question with confidence. Thus, it is possible that lake levels did not rise in the early 1980s and that the water-level data collected at the various locations are not, therefore, directly comparable. In another effort to resolve this problem we contacted Mr. Bob Ohland (a long-time resident of Fort St. John) in October 1997. Without informing him that we suspected that there was a substantial rise in water level in the early 1980s, we asked him if he could remember if there were any substantial changes in water level in the past. In his response he stated that the lake rose by about 3 feet in the early 1980s and that many lakeshore residents were "upset" with the City of Fort St. John and Anderson Exploration Ltd. because their properties were partially inundated (Plates 7 and 8). We then asked Dr. Ted Downs (Fisheries Biologist, BCMELP) if he believed that water levels rose in the early 1980s. Although he could not provide us with any quantitative estimates, he suggested that Mr. Ohland's recollection of a 3-foot rise was reasonable. Thus, there is anecdotal information supporting our finding that peak water levels rose by about 1 m in the early 1980s.

For the years 1989 to 1992, the annual inflow to Charlie Lake ranged from 18,588 dam^3 (1991) to 41,828 dam^3 (1990) (Table 15). Our calculated average annual inflow (27,159 dam^3) is close to that calculated by Nordin and Pommen (1985, p.2) (23,400 dam^3) and UMA Engineering Ltd. (1991, p. 2.2) (26,000 dam^3). According to our estimates, Charlie Lake has an average flushing rate of 0.21 yr^{-1} (Table 11). This translates to an average retention rate of about 5 yr., an estimate reasonably close to that reported in Nordin and Pommen (1985) (5.9-7.4 yr). According to Dr. Rick Nordin (BCMELP, Victoria; pers. comm. to T.D. French, March 1997), retention rates are not considered slow unless they are greater than 10 yr. He suggests that Charlie Lake's retention rate is typical of inland BC lakes.

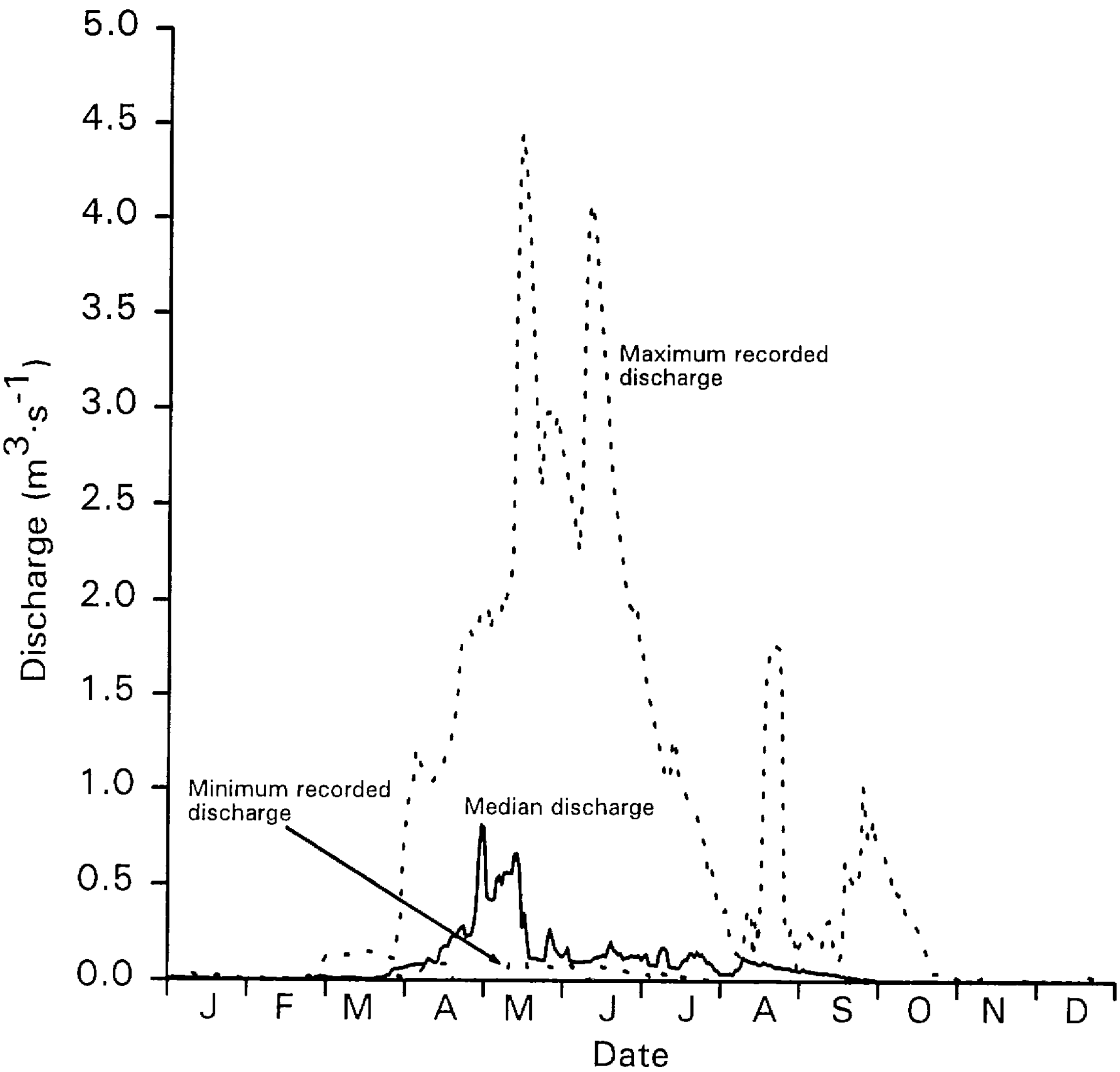

FIGURE 7. DISCHARGE HYDROGRAPH OF LOWER STODDART CREEK (CHARLIE LAKE OUTLET) (1988 TO 1991).

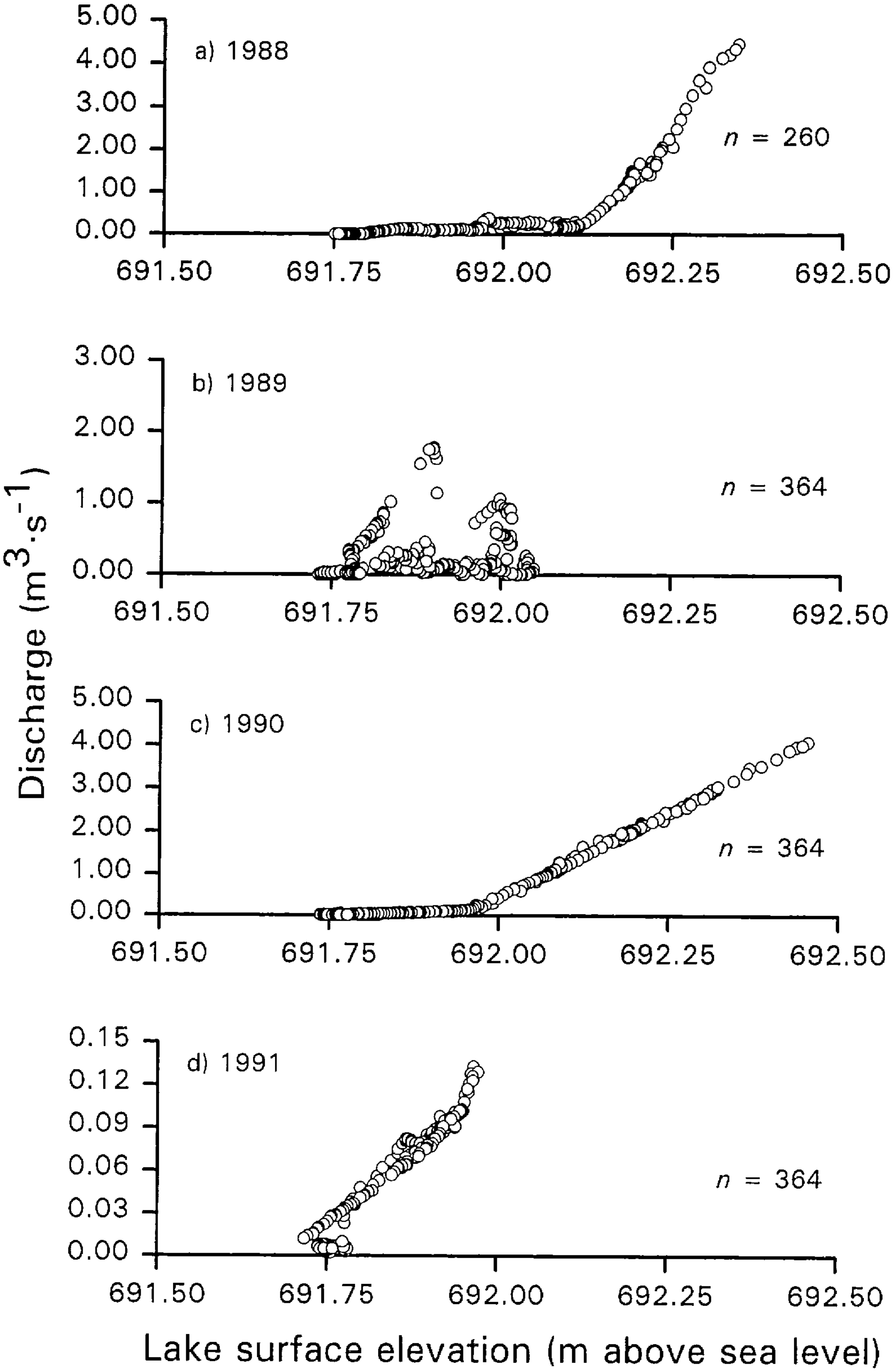

FIGURE 8. LOWER STODDART CREEK (CHARLIE LAKE OUTLET) DISCHARGE IN RELATION TO LAKE SURFACE ELEVATION (ABOVE SEA LEVEL) (1988 TO 1991).

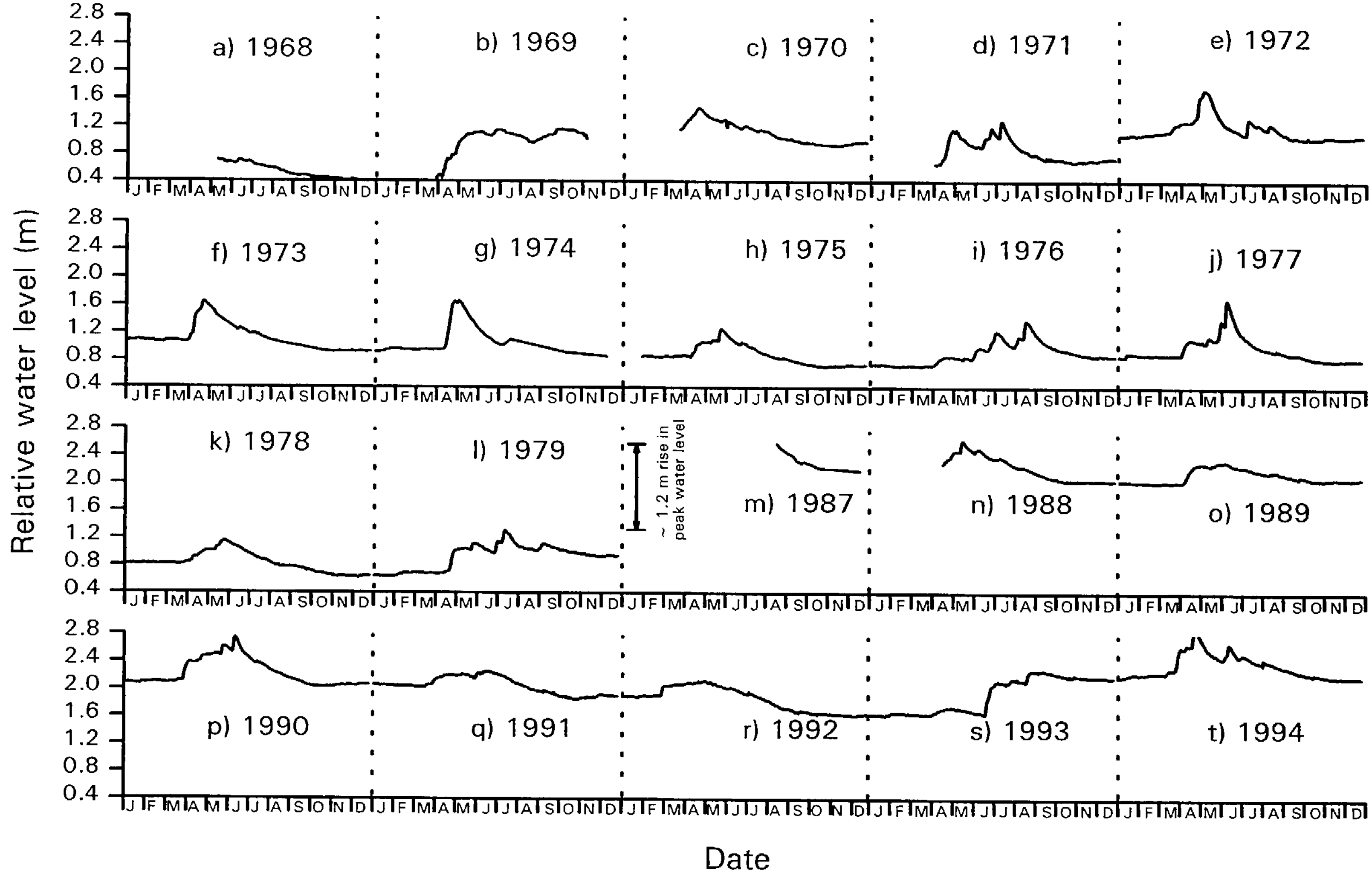

FIGURE 9. CHARLIE LAKE WATER LEVELS (1968 TO 1994). 1968 TO 1990 DATA FROM ENVIRONMENT CANADA (1994). 1991 TO 1994 DATA PROVIDED BY THE CITY OF FORT ST. JOHN.

TABLE 15. HYDROLOGIC CHARACTERISTICS (TOTAL ANNUAL INFLOW, FLUSHING AND RETENTION RATES) OF CHARLIE LAKE (1989-1992).

Year	Δ lake volume (dam^3)	Total outflow via Lower Stoddart Creek (dam^3)	City of Fort St. John withdrawal (dam^3)	Anderson Exploration Ltd. withdrawal (dam^3)	Evaporation (dam^3)	Total inflow to lake (dam^3)	ρ (yr^{-1})	$\frac{1}{\rho}$ (yr)
1989	+560	5,912	2,419	845	12,038	28,081	0.20	5
1990	-257	20,475	2,419	845	12,038	41,828	0.31	3
1991	-2,497	1,058	2,419	845	12,038	20,171	0.17	6
1992	-4,041	1,019	2,419	845	12,038	18,588	0.17	6

5.4 PHYTOPLANKTON COMMUNITY DYNAMICS

The phytoplankton community was composed of species from seven phyla: Phylum Cyanophyta (the blue-green algae), Phylum Bacillariophyta (the diatoms), Phylum Chlorophyta (the green algae), Phylum Cryptophyta, Phylum Chrysophyta (the golden-brown algae), Phylum Dinophyta (the dinoflagellates) and Phylum Euglenophyta (the euglenas). The Cyanophyta were represented by the orders Nostocales (unbranched filamentous blue-greens), Oscillatoriales (branched filamentous blue-greens) and Chroococcales (unicellular greens); the Bacillariophyta by the orders Pennales (pen-shaped diatoms) and Centrales (spherical/circle-shaped diatoms); the Chlorophyta by the orders Chlorococcales, Volvocales, Tetrasporales, and Zygnematales; the Cryptophyta by the order Cryptomonadales; the Chrysophyta by the order Ochromonadales; the Dinophyta by the order Dinokontae; and the Euglenophyta by the order Euglenales.

In total, 52 algal genera were found in Charlie Lake (Table 16). The Cyanophyta were dominated by *Aphanizomenon* spp. (up to 365,841 cells·mL^{-1}), *Oscillatoria* spp. (71,464 cells·mL^{-1}), *Anacystis* spp. (20,229 cells·mL^{-1}), *Gomphosphaerium* spp. (6,036 cells·mL^{-1}), *Coelosphaerium* spp. (980 cells·mL^{-1}) and *Anabaena* spp. (717 cells·mL^{-1}); the Bacillariophyta by *Fragilaria* spp. (3,235 cells·mL^{-1}), *Asterionella* sp. (639 cells·mL^{-1}), *Stephanodiscus* spp. (340 cells·mL^{-1}) and *Cyclotella* spp. (273 cells·mL^{-1}); the Chlorophyta by *Sphaerocystis* sp. (1,333 cells·mL^{-1}), *Schroederia* spp. (488 cells·mL^{-1}), *Eudorina* spp. (472 cells·mL^{-1}) and *Quadrigula* spp. (263 cells·mL^{-1}); the Cryptophyta by *Chroomonas* sp. (2,018 cells·mL^{-1}) and *Cryptomonas* spp. (992 cells·mL^{-1}); the Chrysophyta by *Dinobryon* spp. (184 cells·mL^{-1}) and *Mallomonas* spp. (1,289 cells·mL^{-1}); the Dinophyta by Dinoflagellate spp. (125 cells·mL^{-1}) and *Ceratium* sp. (2.5 cells·mL^{-1}); and the Euglenophyta by *Trachelemonas* spp. (11.7 cells·mL^{-1}) and *Euglena* spp. (8 cells·mL^{-1}). Thus, the structure of Charlie Lake's phytoplankton community (i.e., domination by *Aphanizomenon* spp., *Anacystis* spp., *Anabaena* sp., *Fragilaria* spp., *Asterionella* sp., and *Stephanodiscus* spp.) is typical of eutrophic lakes (Hutchinson 1967).

Chrysophyte, Cryptophyte and dinoflagellate abundance peaked in early April, when surface waters were cold (< 3 °C) and ice-covered (Figure 10 d,e,f). The Chrysophyte and dinoflagellate populations collapsed by the end of April, where the Cryptophyte population maintained its bloom density until the middle of May. Our finding that the Cryptophyte population bloomed under ice, is consistent with observations made by Wright (1964) and Pechlauer (1971). Some authors have suggested that the phenological patterns of Chrysophyte populations are linked to the potassium cycle (e.g., Lehman 1976). We were unable to make this association because potassium concentrations were measured on only four occasions.

The diatom populations bloomed soon after spring (Centrales) and autumn (Pennales) overturns (Figure 10b), when surface-water silica (an important constituent of diatom frustrules (Lewin 1957)) concentrations were high (Figures 11 and 12). The apparent relationship between surface-water silica concentration and diatom abundance has also been described by Lund (1949, 1950, 1954, 1959), Knudson (1957), Heron (1961), Conway *et al.* (1977) and many others. The mechanism by which silica concentrates in surface waters during the spring and autumn was not investigated; however, it appears that silica-rich hypolimnetic waters upwell to the surface when thermal stratification is lost in the spring and autumn (Figure 11), increasing surface water silica concentration 4-fold (Figure 12).

While Bacillariophyte, Cryptophyte, Chrysophyte, Dinophyte, and Euglenophyte blooms are inconspicuous to most human observers, Chlorophyte and Cyanophyte blooms can develop into dense "scums" that are bothersome to many water users. Chlorophyte populations bloomed in May, June (Chlorococcales and Tetrasporales) and July (Volvocales) (Figure 10c). The Cyanophyte population bloomed in August and September (Figure 10a). Since *Aphanizomenon* spp. (Kotak *et al.* 1993), *Anacystis* spp. (Botes *et al.* 1985; Watanabe *et al.* 1986; Namikoshi *et al.* 1992a; Sivonen *et al.* 1992), *Anabaena* spp. (Krishnamurthy *et al.* 1986; Harada *et al.* 1991; Namikoshi *et al.* 1992b,c), and *Oscillatoria* spp. (Meriluoto *et al.* 1989; Bruno *et al.* 1992) can produce highly toxic compounds, water users should be advised to keep livestock and pets away from Charlie Lake during and after late-summer algal blooms.

TABLE 16. ALGAL SPECIES OBSERVED IN CHARLIE LAKE, WITH THEIR PEAK AND RELATIVE ABUNDANCES. DATA COLLECTED AT THE NORTH (E207459) AND SOUTH (0400390) DEEP STATIONS.

Taxon	Peak abundance (cells·mL^{-1})	Within-order relative abundance (% total abundance)
a) Cyanophyta (blue-greens)		
Order Nostocales		
Anabaena sp.	135	0.04
Anabaena affinis	VR	negligible
Anabaena flos-aquae	582	0.2
Aphanizomenon sp.	2.5	0.0007
Aphanizomenon flos-aquae	365,838	99.8
Order Oscillatoriales		
Lyngbya sp.	VR	negligible
Oscillatoria limnetica	233	0.3
Oscillatoria tenuis	71,231	99.6
Pseudoanabaena catanata	73	0.1
Order Chroococcales		
Agmenellum sp.	VR	negligible
Agmenellum tenuissima	VR	negligible
Anacystis sp.	10	0.04
Anacystis aeruginosa	20,229	74.3
Anacystis limneticus	VR	negligible
Aphanocapsa sp.	VR	negligible
Coelosphaerium sp.	VR	negligible
Coelosphaerium naegelianum	980	3.4
Dactylococcopsis sp.	VR	negligible
Gomphosphaerium sp.	50	0.2
Gomphosphaerium naegelianum	5,986	22.0
Gomphosphaerium pallidum	VR	negligible
b) Bacillariophyta (diatoms)		
Order Pennales		
Achnanthes minutissima	VR	negligible
Amphora sp.	2.6	0.07
Asterionella formosa	639	16.1
Cocconeis sp.	VR	negligible
Cyamatopleura sp.	3	0.08
Cyamatopleura solea	VR	negligible
Cymbella sp.	5	0.1
Cymbella affinis	VR	negligible
Diatoma sp.	3	0.1
Epithema sp.	VR	negligible
Epithema sorex	VR	negligible
Eunotia sp.	2.9	0.1
Fragilaria sp.	907	22.9
Fragilaria capucina	1,884	47.6
Fragilaria construens	434	11.0
Fragilaria crotonensis	9.6	0.2

Taxon	Peak abundance (cells·mL^{-1})	Within-order relative abundance (% total abundance)
Frustrulia sp.	VR	negligible
Gomphonema sp.	VR	negligible
Navicula sp.	25.1	0.6
Nitzchia sp.	3.4	0.1
Pleurosigma sp.	VR	negligible
Synedra sp.	0.5	0.01
Synedra ulna	5	0.1
Tabellaria sp.	32.4	0.8
Tabellaria fenestrata	5	0.1
Order Centrales		
Coscinodiscus sp.	VR	negligible
Cyclotella sp.	189.8	29.0
Cyclotella bodanica	2.6	0.4
Cyclotella glomerata	81	12.4
Melosira sp.	41.3	6.3
Stephanodiscus sp.	104	15.9
Stephanodiscus astera	177	27.1
Stephanodiscus niagarae	58.5	8.9
c) Chlorophyta (greens)		
Order Chlorococcales		
Ankistrodesmus sp.	VR	negligible
Ankistrodesmus falcatus	5.1	0.2
Ankyra sp.	2.7	0.1
Botryococcus braunii	VR	negligible
Dictosphaerium sp.	VR	negligible
Elaktothrix sp.	VR	negligible
Elaktothrix gelatinosa	16	0.8
Nephrocytium sp.	VR	negligible
Oocystis sp.	10	0.5
Oocystis borgei	VR	negligible
Pediastrum sp.	1.2	0.1
Quadrigula sp.	3	0.1
Quadrigula lacustris	260	12.2
Scendesmus sp.	VR	negligible
Scendesmus quadricauda	10	0.5
Schroederia sp.	237.5	11.2
Schroederia judayi	245	11.5
Schroederia setigera	5.1	0.2
Sphaerocystis schroeteri	1,333	62.6
Selanastrum sp.	VR	negligible
Tetraedron minimum	VR	negligible
Order Volvocales		
Chlorogonium sp.	5.1	0.9
Eudorina elegans	34	6.1
Eudorina sp.	438	78.3
Volvox sp.	31.2	5.6

Taxon	Peak abundance (cells·mL^{-1})	Within-order relative abundance (% total abundance)
Pandorina morum	51	9.1
Order Tetrasporales		
Gloeocystis sp.	12.9	100
Order Zygnematales		
Closterium sp.	VR	negligible
Cosmarium sp.	2.6	51.0
Staurastrum sp.	2.5	49.0
Staurastrum natator	VR	negligible
Staurastrum paradoxum	VR	negligible
d) Cryptophyta		
Order Cryptomonadales		
Chroomonas acuta	2,018	67.0
Cryptomonas sp.	175	5.8
Cryptomonas acuta	335.8	11.2
Cryptomonas marssonii	8	0.3
Cryptomonas ovata	473	15.7
e) Chrysophyta (golden-browns)		
Order Ochromonadales		
Dinobryon sp.	11.7	0.8
Dinobryon divergens	172	11.7
Mallomonas sp.	26.3	1.8
Mallomonas akromos	1,263	85.7
f) Dinophyta (dinoflagellates)		
Order Dinokontae		
Dinoflagellate spp.	125	98.0
Ceratium hirundinella	2.5	2.0
g) Euglenophyta (euglenas)		
Order Euglenales		
Euglena spp.	8	40.6
Trachelemonas spp.	11.7	59.4

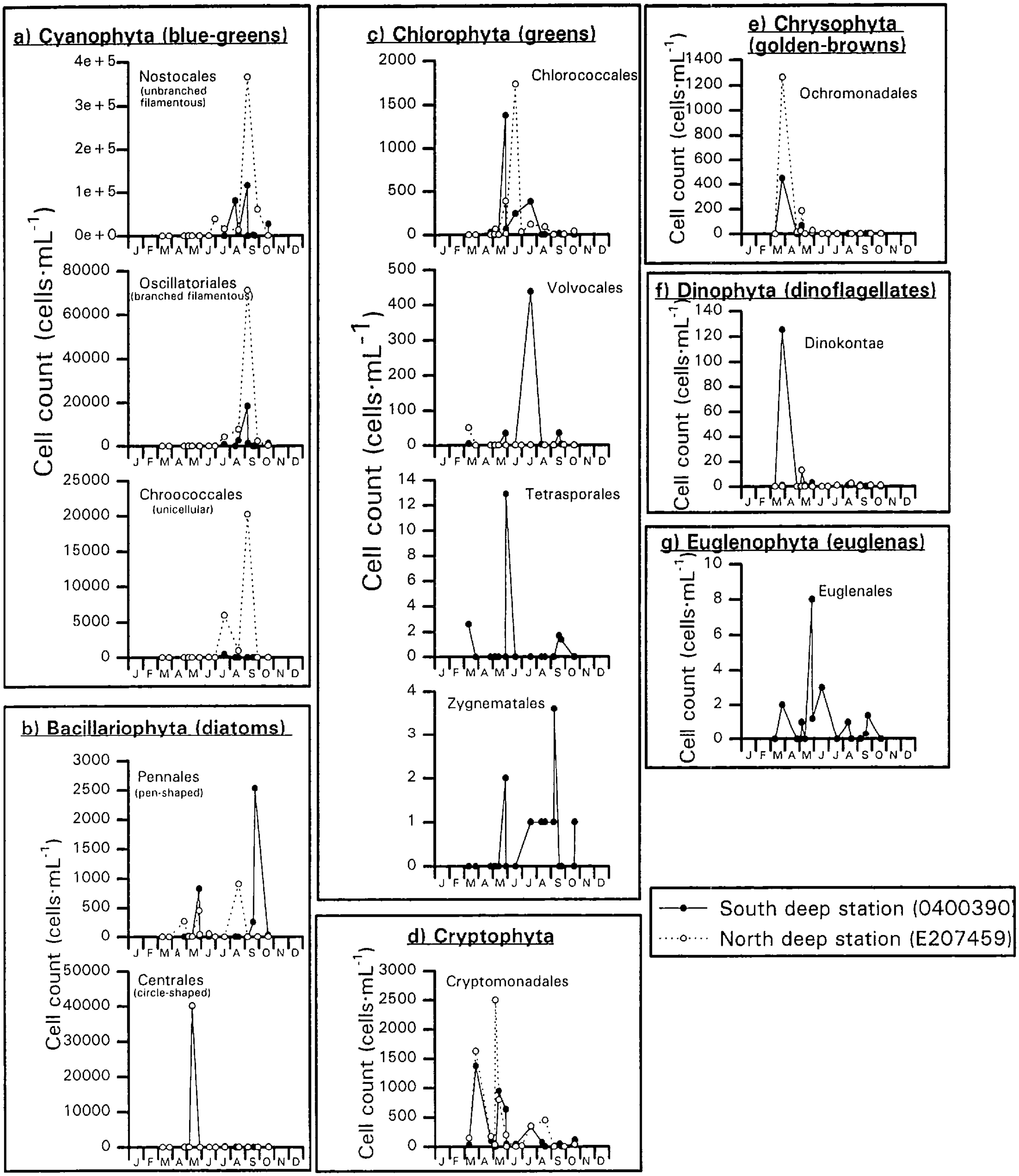

FIGURE 10. THE PHENOLOGY OF CHARLIE LAKE'S ALGAL COMMUNITY: A) CYANOPHYTES, B) BACILLARIOPHYTES, C) CHLOROPHYTES, D) CRYPTOPHYTES, E) CHRYSOPHYTES, F) DINOPHYTES, G) EUGLENOPHYTES.

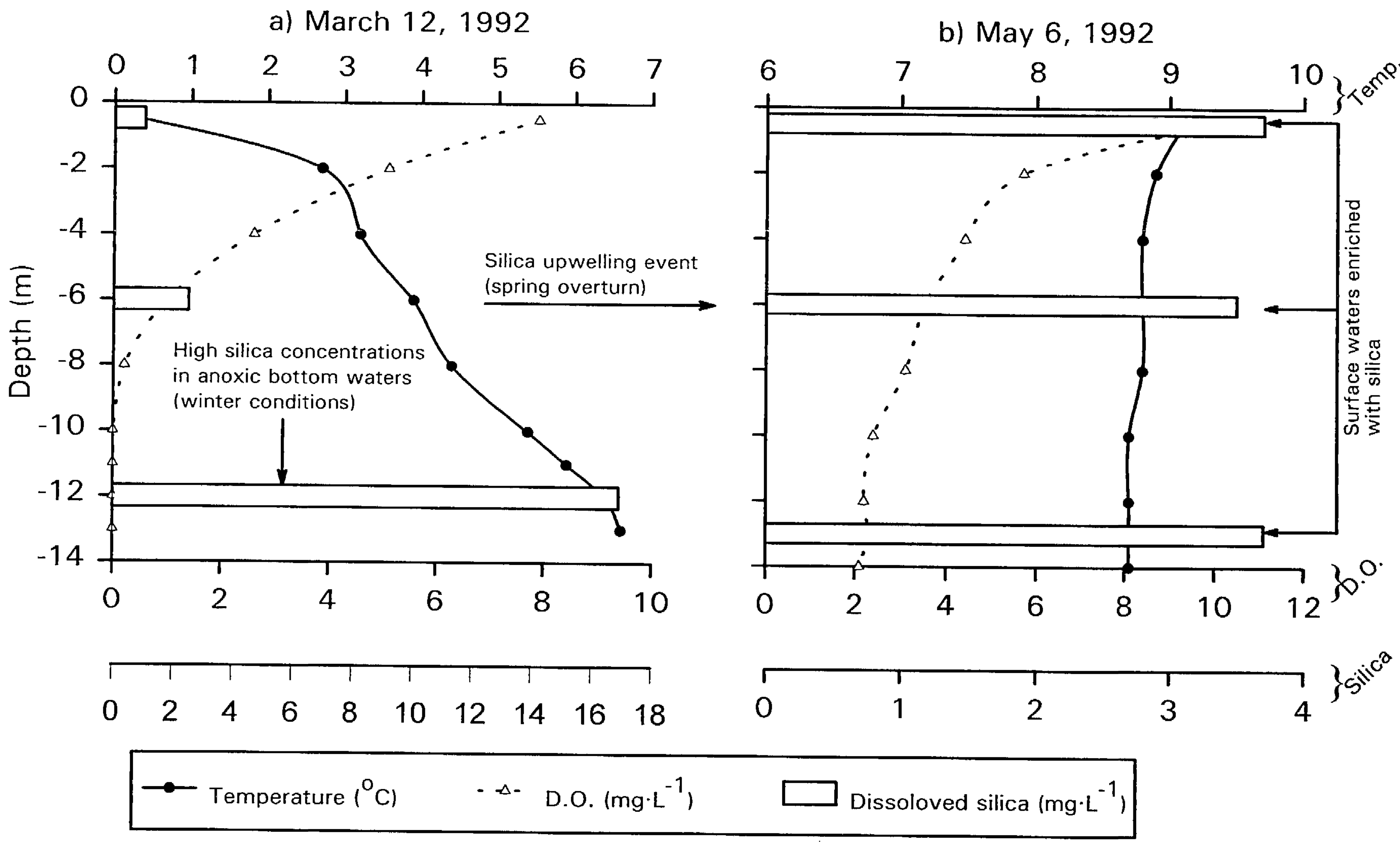

FIGURE 11. EXAMPLE SPRING SILICA UPWELLING EVENT (SOUTH DEEP STATION, EMS SITE 0400390, 1992. SPRING UPWELLING EVENTS MAY TRIGGER SPRING CENTRIC DIATOM BLOOMS.

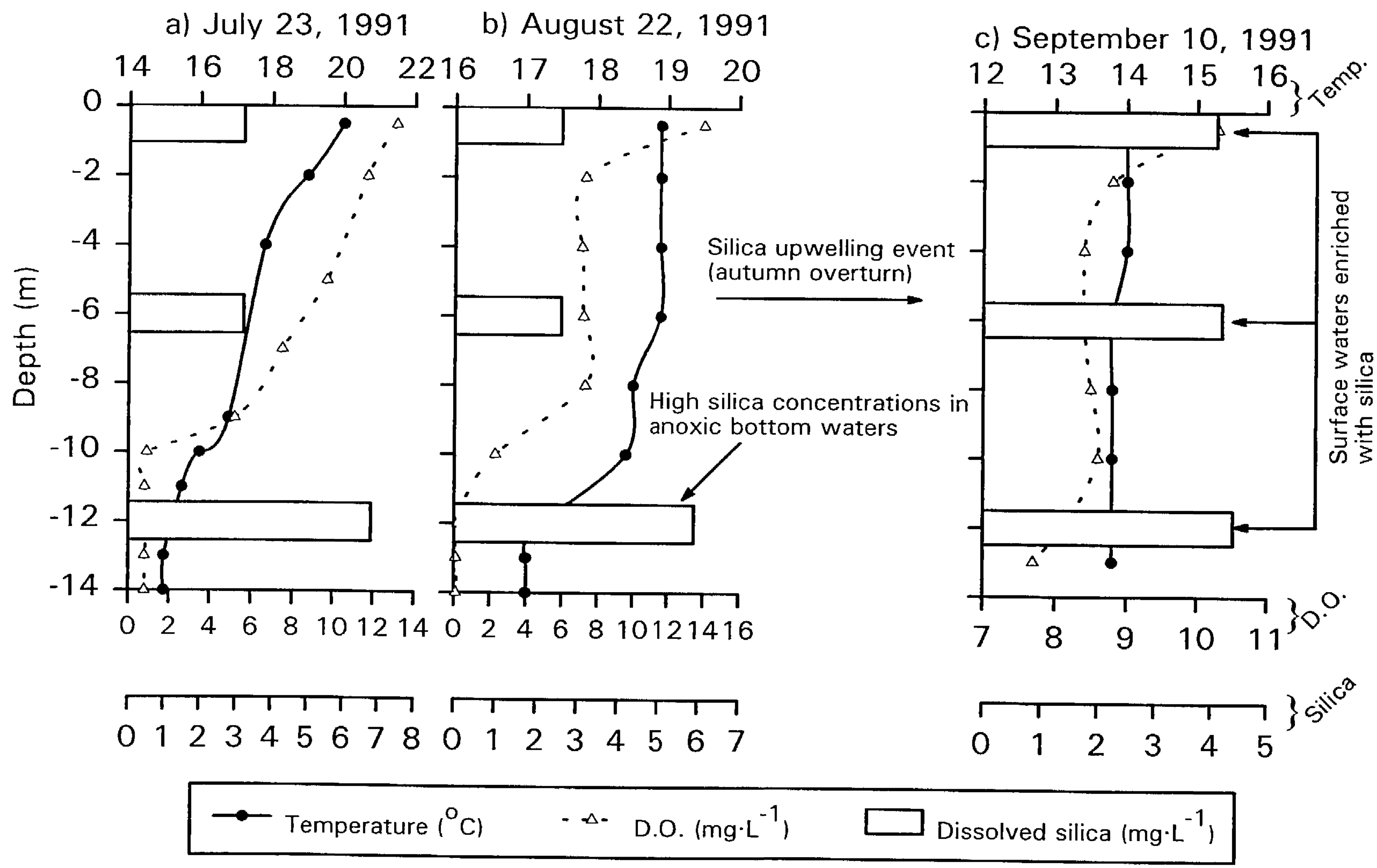

FIGURE 12. EXAMPLE AUTUMN SILICA UPWELLING EVENT (SOUTH DEEP STATION, EMS SITE 0400390, 1992). AUTUMN UPWELLING EVENTS MAY TRIGGER AUTUMN PENNATE DIATOM BLOOMS.

5.6 LIMNOLOGICAL ANALYSES

5.6.1 Nutrient Limitation

Nutrient limitation in Charlie Lake can be described in three distinct phases: under-ice P-limitation (Phase 1, March to mid-May); open-water P limitation (Phase 2, mid-May to mid-July); and open-water no- or co-limitation (Phase 3, mid-July to late-October) (Figure 13). Under-ice Chrysophyte (golden-brown algae) and dinophyte (dinoflagellates) blooms were associated with the Phase 1 dynamic (18 of 21 observations indicated P-limitation) (Figures 10 and 13). Open-water Chlorophyte (green algae) blooms occurred during the Phase 2 dynamic, when 34 of 43 observations indicated P-limitation. Cyanophyte (blue-green algae) blooms occurred in the late summer-early autumn months (Phase 3) when N may have been co-limiting (39 of 48 observations indicated co-limitation or no limitation) (Figures 10 and 13). Cyanophytes may have dominated the phytoplankton community during periods of possible N-limitation because they are the only algal group that has the ability to fix atmospheric N (Lee 1989).

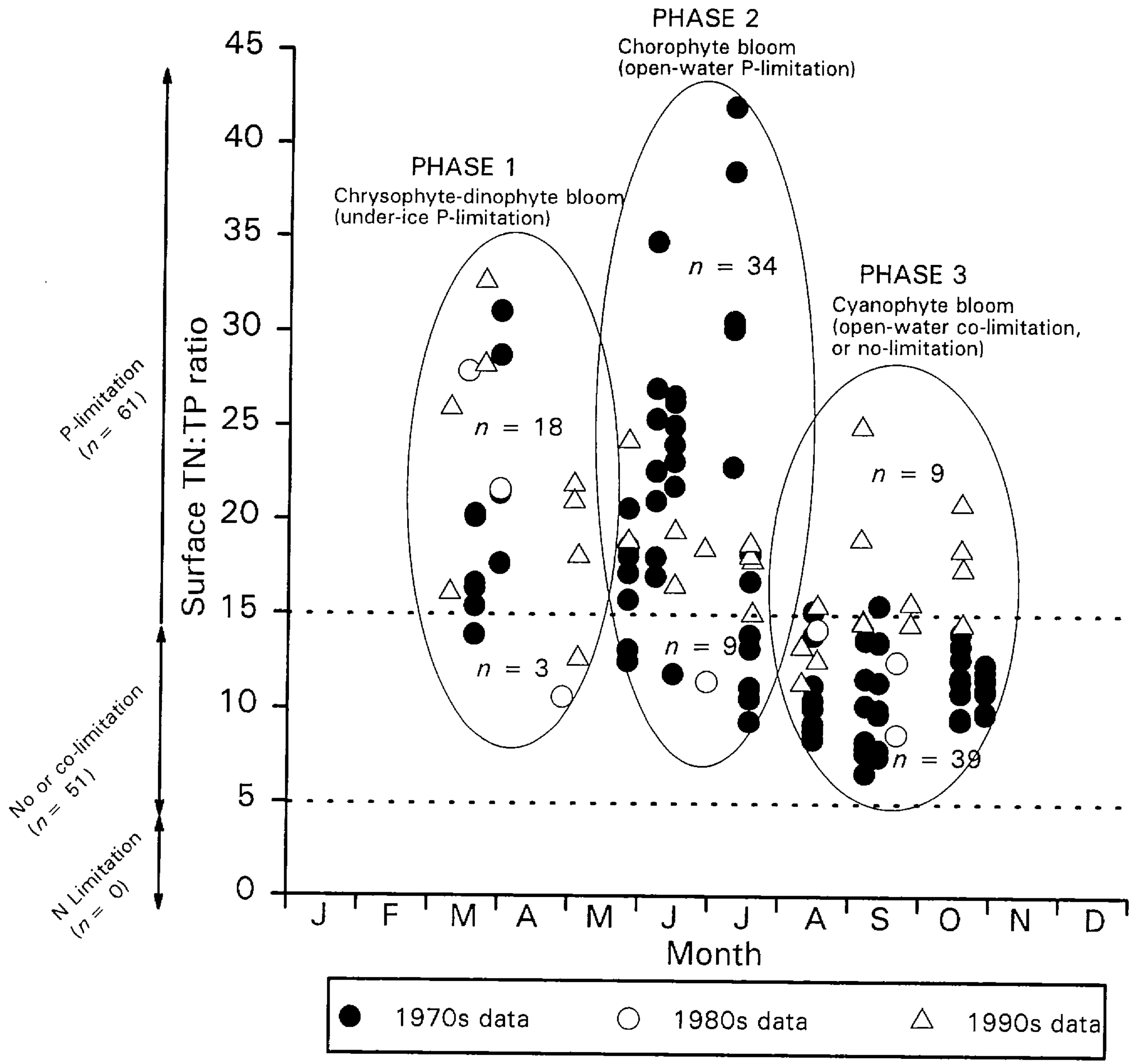

FIGURE 13. TRI-PHASIC SURFACE WATER TN:TP RATIOS (1974 TO 1995).

5.6.2 Effects of Watershed Development

TDP concentrations ranged from 9 to 166 $\mu g \cdot L^{-1}$ in the control stream (Table 17k). They were significantly ($P<0.05$) higher in the tributary streams, such that they ranged from 15 to 200 $\mu g \cdot L^{-1}$ in East Stoddart Creek, 27 to 650 $\mu g \cdot L^{-1}$ in West Stoddart Creek, 20 to 355 $\mu g \cdot L^{-1}$ in 281st Road Creek, 20 to 640 $\mu g \cdot L^{-1}$ in Beatton Park Creek, 38 to 934 $\mu g \cdot L^{-1}$ in Ditch #5, 23 to 354 $\mu g \cdot L^{-1}$ in Ditch #3, 4 to 995 $\mu g \cdot L^{-1}$ in Boat Launch Creek, and 33 to 700 $\mu g \cdot L^{-1}$ in Coffee Creek (Table 17c-j) (Table 18). On average, TDP concentrations in the three largest tributaries (i.e., East and West Stoddart creeks and Coffee Creek) were 1.2×, 2.2×, 3.6×, and 4.6× those observed in the control stream in March, April, May and June, respectively. Thus, the data suggest that watershed development has substantially increased the TDP content of the tributaries and, thereby, the rate at which they export TDP to the lake (Plates 13-14). Under the assumption that algal production in the lake is P-limited, we conclude that the elevation of tributary TDP concentrations has accelerated eutrophication processes in the lake. This conclusion is consistent with that of Reavie *et al.* (1995b), who inferred from paleolimnological data that P concentrations in the lake increased with the onset (mid- to late 1800s) of rapid human settlement (Section 5.1.4). By comparison, TP concentrations in the control stream were similar ($P>0.05$) to those observed in the tributary streams (Table 19).

Some within-watershed variation was evident (Tables 18 and 19). Based on the number of significant ($P<0.05$) Wilcoxin ranks, the highest TDP concentrations were observed in Ditch # 5 (8+ ranks), Coffee Creek (3+ ranks), and Boat Launch Creek (3+ ranks). The lowest TDP concentrations were observed in West Stoddart Creek (2+ ranks), 281st Road Creek (2+ ranks), Ditch #3 (2+ ranks), Beatton Park Creek (2+ ranks) and East Stoddart Creek (1+ rank). By comparison, the highest TP concentrations were observed in 281st Road Creek (5+ ranks), West Stoddart Creek (4+ ranks) and Coffee Creek (3+ ranks). The lowest TP concentrations were observed in Ditch #5 (1+ rank), East Stoddart Creek (0+ rank), Beatton Park Creek (0+ rank), Ditch #3 (0+ rank) and Boat Launch Creek (0+ rank).

TABLE 17. SUMMARY OF CHARLIE LAKE TRIBUTARY DATA: A) STODDART CREEK AT 64 ROAD (0410026); B) STODDART CREEK 2 KM UPSTREAM OF CHARLIE LAKE (0400396); C) EAST STODDART CREEK (E207903); D) WEST STODDART CREEK (E207904); E) 281ST ROAD CREEK (E207902); F) BEATTON PARK CREEK (E207896); G) RESIDENTIAL DITCH 5 (E207898); H) RESIDENTIAL DITCH 3 (E207899); I) BOAT LAUNCH CREEK (E207900); J) COFFEE CREEK AT 114 ROAD CROSSING (E207901); K) FORESTED CONTROL CREEK (E207906). LDC = LOWEST DETECTABLE CONCENTRATION.

TABLE 17A. STODDART CREEK AT 64 ROAD (0410026).

Parameter	Sampling periods (sample size)	Mean/median	Range	# < LDC
Carbon, tot. ($mg \cdot L^{-1}$)	1980s (1)	50 (single measure)	-	0
Carbon, tot. organic ($mg \cdot L^{-1}$)	1980s (1)	20 (single measure)	-	0
Chloride, diss. ($mg \cdot L^{-1}$)	1980s (1)	3.4 (single measure)	-	0
Coliforms, *Enterococcus* ($CFU \cdot cL^{-1}$)	1980s (2)	11/-	2-20	0
Coliforms, *E. coli* ($CFU \cdot cL^{-1}$)	1980s (2)	104/-	7-200	0
Coliforms, fecal				
MPN	1980s (3)	14/14	5-23	0
$CFU \cdot cL^{-1}$	1980s (3)	115/76	8-270	0
Coliforms, *Streptococcus* ($CFU \cdot cL^{-1}$)	1980s (3)	13/6	0-32	0
Color (TAC)	1980s (2)	120/-	80-160	0
Conductivity ($\mu S \cdot cm^{-1}$)	1980s (10)	235/196	150-388	0
Nitrogen, ammonia ($mg \cdot L^{-1}$)	1980s (41)	0.073/0.079	0.010-0.202	0
Nitrogen, nitrite+nitrate ($mg \cdot L^{-1}$)	1980s (41)	0.07/0.05	<0.02-0.50	9
Nitrogen, tot. ($mg \cdot L^{-1}$)	1980s (9)	1.5/1.5	0.6-2.6	0
Nitrogen, tot. Kjeldahl ($mg \cdot L^{-1}$)	1980s (10)	1.5/1.4	0.6-2.1	0
Nitrogen, tot. organic ($mg \cdot L^{-1}$)	1980s (9)	1.3/1.3	0.6-2.1	0
Oxygen, biological demand ($mg \cdot L^{-1}$)	1980s (1)	1 (single measure)	-	0
pH (pH units)	1980s (10)	7.6/7.6	7.2-7.9	0
Phosphorus, ortho ($mg \cdot L^{-1}$)	1980s (41)	0.033/0.021	0.003-0.179	0
Phosphorus, tot. ($mg \cdot L^{-1}$)	1980s (47)	0.189/0.142	0.044-0.623	0
Phosphorus, tot. diss. ($mg \cdot L^{-1}$)	1980s (46)	0.069/0.053	0.027-0.242	0
Potassium, diss. ($mg \cdot L^{-1}$)	1980s (1)	3.4 (single measure)	-	0
Residues, filt. ($mg \cdot L^{-1}$)	1980s (5)	227/252	160-276	0
Residues, non-filt. ($mg \cdot L^{-1}$)	1980s (7)	68/11	3-376	0
Silica, diss. ($mg \cdot L^{-1}$)	1980s (1)	4.4 (single measure)	-	-
Turbidity (NTU)	1980s (1)	1 (single measure)	-	-

TABLE 17B. STODDART CREEK 2 KM UPSTREAM OF CHARLIE LAKE (0400396).

Parameter	Sampling periods (sample size)	Mean/median	Range	# < LDC
Calcium, dissolved (mg·L^{-1})	1970s (6)	21.3/31.6	11.2-39.7	0
Carbon, tot. organic (mg·L^{-1})	1970s (6)	28/38	16-38	0
Color (TAC)	1970s (2)	89/146	17-160	0
Conductivity (μS·cm^{-1})	1970s (6)	199/290	108-388	0
Hardness, diss. (mg·L^{-1})	1970s (6)	44/156	44-156	0
Magnesium, diss. (mg·L^{-1})	1970s (6)	7.1/10.7	3.9-13.7	0
Nitrogen, ammonia (mg·L^{-1})	1970s (6)	0.584/1.550	0.089-1.930	0
Nitrogen, nitrate+nitrite (mg·L^{-1})	1970s (1)	0.15 (single measure)	-	-
Nitrogen, tot. Kjeldahl (mg·L^{-1})	1970s (6)	1.9/3.5	0.6-4.0	0
Nitrogen, tot. organic (mg·L^{-1})	1970s (6)	1.3/2.0	0.5-2.1	0
Oxygen, diss. (mg·L^{-1})	1970s (2)	5.5/5.7	5.2-5.8	0
pH (pH units)	1970s (6)	7.3/7.6	7.0-7.8	0
Phosphorus, ortho (mg·L^{-1})	1970s (2)	0.227/0.371	0.046-0.407	0
Phosphorus, tot. (mg·L^{-1})	1970s (6)	0.361/0.603	0.094-0.786	0
Phosphorus, tot. diss. (mg·L^{-1})	1970s (1)	0.442 (single measure)	-	-
Residues, filt. (mg·L^{-1})	1970s (6)	172/244	110-279	0
Residues, non-filt. (mg·L^{-1})	1970s (6)	35/64	9-84	0
Temperature, °C	1970s (3)	12.3/16.3	6.5/17.0	0
Turbidity (NTU)	1970s (5)	30/54	5-64	0

TABLE 17C. EAST STODDART CREEK (E207903).

Parameter	Sampling periods (sample size)	Mean/median	Range	# < LDC
Coliforms, *Enterococcus* (CFU·cL^{-1})	1980s (1)	20 (single sample)	-	-
Coliforms, *E. coli* (CFU·cL^{-1})	1980s (1)	<2 (single sample)	-	-
Coliforms, fecal (CFU·cL^{-1})	1980s (8); 1990s (31)	114/3	<1-2760	12
Nitrogen, ammonia (mg·L^{-1})	1980s (4)	0.211/0.192	0.138-0.322	-
Phosphorus, tot. (mg·L^{-1})	1980s (4); 1990s (38)	0.205/0.185	0.017-0.500	0
Phosphorus, tot. diss. (mg·L^{-1})	1980s (9); 1990s (39)	0.079/0.059	0.015-0.200	0

TABLE 17D. WEST STODDART CREEK (E207904).

Parameter	Sampling periods (sample size)	Mean/median	Range	# < LDC
Coliforms, *Enterococcus* (CFU·cL^{-1})	1980s (3)	289/228	41-599	-
Coliforms, *E. coli* (CFU·cL^{-1})	1980s (3)	48/55	10-79	-
Coliforms, fecal (CFU·cL^{-1})	1980s (8); 1990s (32)	276/36	<1-3700	5
Nitrogen, ammonia (mg·L^{-1})	1980s (4)	0.429/0.494	0.088-0.640	0
Phosphorus, tot. (mg·L^{-1})	1980s (8); 1990s (40)	0.298/0.180	0.084-0.900	0
Phosphorus, tot. diss. (mg·L^{-1})	1980s (7); 1990s (40)	0.119/0.070	0.027-0.650	0

TABLE 17E. 281ST ROAD CREEK (E207902).

Parameter	Sampling periods (sample size)	Mean/median	Range	# < LDC
Coliforms, *Enterococcus* (CFU·cL^{-1})	1980s (3)	936/539	120-2150	0
Coliforms, *E. coli* (CFU·cL^{-1})	1980s (3)	19/9	<2-45	1
Coliforms, fecal (CFU·cL^{-1})	1980s (8); 1990s (24)	3952/12	<1-65000	3
Coliforms, fecal (MPN)	1980s (1);	> 1600 (single sample)	-	-
Nitrogen, ammonia (mg·L^{-1})	1980s (4)	0.073/0.075	0.023-0.118	0
Phosphorus, tot. (mg·L^{-1})	1980s (9); 1990s (33)	0.324/0.246	0.026-0.700	0
Phosphorus, tot. diss. (mg·L^{-1})	1980s (9); 1990s (33)	0.123/0.105	0.020-0.355	0

TABLE 17F. BEATTON PARK CREEK (E207896).

Parameter	Sampling periods (sample size)	Mean/median	Range	# < LDC
Coliforms, *Enterococcus* (CFU·cL^{-1})	1980s (3)	311/298	72-563	0
Coliforms, *E. coli* (CFU·cL^{-1})	1980s (4)	3/2	<2-6	3
Coliforms, fecal				
MPN	1990s (1)	240 (single measure)	-	-
CFU·cL^{-1}	1980s (7); 1990s (27)	85/3	2-930	14
Nitrogen, ammonia (mg·L^{-1})	1980s (4)	1.548/1.710	0.043-2.730	0
Phosphorus, tot. (mg·L^{-1})	1980s (8); 1990s (34)	0.253/0.170	0.038-0.940	0
Phosphorus, tot. diss. (mg·L^{-1})	1980s (10); 1990s (36)	0.144/0.073	0.020-0.640	0

TABLE 17G. RESIDENTIAL DITCH #5 (E207898).

Parameter	Sampling periods (sample size)	Mean/median	Range	# < LDC
Coliforms, *Enterococcus* (CFU·cL^{-1})	1980s (1)	136 (single measure)	-	0
Coliforms, *E. coli* (CFU·cL^{-1})	1980s (1)	2 (single measure)	-	0
Coliforms, fecal (CFU·cL^{-1})	1980s (8); 1990s (28)	185/9	<1-1900	9
Nitrogen, ammonia (mg·L^{-1})	1980s (4)	0.371/0.200	0.016-1.070	0
Phosphorus, tot. (mg·L^{-1})	1980s (8); 1990s (36)	0.295/0.235	0.071-1.210	0
Phosphorus, tot. diss. (mg·L^{-1})	1980s (10); 1990s (36)	0.194/0.153	0.038-0.934	0

TABLE 17H. RESIDENTIAL DITCH #3 (E207899).

Parameter	Sampling periods (sample size)	Mean/median	Range	# < LDC
Coliforms, *Enterococcus* ($CFU \cdot cL^{-1}$)	1980s (3)	355/396	104-565	0
Coliforms, *E. coli* ($CFU \cdot cL^{-1}$)	1980s (3)	3/2	<2-4	2
Coliforms, fecal ($CFU \cdot cL^{-1}$)	1980s (6); 1990s (19)	202/30	<2-1500	1
Nitrogen, ammonia ($mg \cdot L^{-1}$)	1980s (4)	0.014/0.014	0.006-0.022	0
Phosphorus, tot. ($mg \cdot L^{-1}$)	1980s (8); 1990s (26)	0.334/0.317	0.089-0.977	0
Phosphorus, tot. diss. ($mg \cdot L^{-1}$)	1980s (8); 1990s (26)	0.129/0.092	0.023-0.354	0

TABLE 17I. BOAT LAUNCH CREEK (E207900).

Parameter	Sampling periods (sample size)	Mean/median	Range	# < LDC
Coliforms, *Enterococcus* ($CFU \cdot cL^{-1}$)	1980s (3)	188/217	80-268	0
Coliforms, *E. coli* ($CFU \cdot cL^{-1}$)	1980s (3)	150/134	47-270	0
Coliforms, fecal				
MPN	1980s (1)	164/6.5	-	-
$CFU \cdot cL^{-1}$	1980s (6); 1990s (22)		<1-1300	7
Nitrogen, ammonia ($mg \cdot L^{-1}$)	1980s (4)	0.015/0.014	0.008-0.026	0
Phosphorus, tot. ($mg \cdot L^{-1}$)	1980s (8); 1990s (29)	0.321/0.249	0.040-1.180	0
Phosphorus, tot. diss. ($mg \cdot L^{-1}$)	1980s (9); 1990s (29)	0.189/0.098	0.004-0.995	0

TABLE 17J. COFFEE CREEK AT 114 ROAD CROSSING (E207901).

Parameter	Sampling periods (sample size)	Mean/median	Range	# < LDC
Coliforms, *Enterococcus* (CFU·cL^{-1})	1980s (1)	10 (single sample)	-	-
Coliforms, *E. coli* (CFU·cL^{-1})	1980s (1)	2 (single sample)	-	-
Coliforms, fecal (CFU·cL^{-1})	1980s (8); 1990s (30)	45/2	<1-710	15
Nitrogen, ammonia (mg·L^{-1})	1980s (4)	0.042/0.029	0.017-0.093	0
Phosphorus, ortho (mg·L^{-1})	1980s (1)	0.048 (single sample)	-	-
Phosphorus, tot. (mg·L^{-1})	1980s (8); 1990s (37)	0.348/0.196	0.078-3.490	0
Phosphorus, tot. diss. (mg·L^{-1})	1980s (10); 1990s (48)	0.138/0.073	0.033-0.700	0

TABLE 17K. FORESTED CONTROL CREEK (E207906).

Parameter	Sampling periods (sample size)	Mean/median	Range	# < LDC
Coliforms, *Enterococcus* (CFU·cL^{-1})	1980s (2)	77/77	-	0
Coliforms, *E. coli* (CFU·cL^{-1})	1980s (2)	89/89	-	0
Coliforms, fecal (CFU·cL^{-1})	1980s (6); 1990s (15)	87/14	<2-1150	4
Nitrogen, ammonia (mg·L^{-1})	1980s (2)	0.026/0.026	-	0
Phosphorus, tot. (mg·L^{-1})	1980s (2); 1990s (16)	0.187/0.130	0.055-1.190	0
Phosphorus, tot. diss. (mg·L^{-1})	1980s (6); 1990s (16)	0.028/0.014	0.009-0.166	0

TABLE 18. WILCOXIN-PAIRED-SAMPLE MATRIX FOR TDP COMPARING TRIBUTARY STREAMS/DITCHES AND CONTROL STREAM. CELLS WITH ASTERISKS (**) INDICATE STATISTICALLY SIGNIFICANT COMPARISONS. CELLS WITH UNDERLINED *P*-VALUES AND ASTERISKS ARE NOT, SIGNIFICANT AT α 0.05. HOWEVER, IT IS BELIEVED THAT THEY WOULD BE SIGNIFICANT IF A LARGER NUMBER OF SAMPLES WERE TAKEN. THE DENOMINATOR OF THE BRACKETED FRACTION IN EACH CELL IS THE NUMBER OF COMPARISONS MADE (N). THE NUMERATOR IS THE NUMBER OF WILCOXIN RANKS WON BY THE TRIBUTARY THAT THE ARROW IS POINTING TO. THE ARROW POINTS TO THE TRIBUTARY HAVING THE GREATEST NUMBER OF WINNING RANKS (I.E., THE HIGHEST TDP CONCENTRATION).

	West Stoddart (E207904)	East Stoddart (E207903)	281st Road (E207902)	Beatton Park (E207896)	Ditch 5 @ mouth (E207898)	Ditch 3 @ mouth (E207899)	Boat Launch (E207900)	Coffee @ 114 Road crossing (E207901)	Forested "control" (E207906)
West Stoddart	-	$P < 0.05$** ← (35/45)	$P = 0.80$ ← (19/37)	$P = 0.56$ = (36 pairs)	$P < 0.05$** ↑ (33/36)	$P = 0.42$ ← (14/26)	$P = 0.27$ ↑ (17/30)	$P < 0.05$** ↑ (32/41)	$P < 0.05$** ← (16/17)
East Stoddart	-	-	$P < 0.05$** ↑ (28/40)	$P < 0.05$** ↑ (32/39)	$P < 0.05$** ↑ (38/39)	$P < 0.05$** ↑ (19/29)	$P < 0.05$** ↑ (25/33)	$P < 0.05$** ↑ (37/44)	$P < 0.05$** ← (17/18)
281st Road Creek	-	-	-	$P = 0.23$ ↑ (24/38)	$P < 0.05$** ↑ (38/38)	$P = 0.14$ ← (20/31)	$P = 0.16$ ↑ (20/35)	$P = 0.51$ ↑ (23/40)	$P < 0.05$** ← (12/13)
Beatton Park	-	-	-	-	$P < 0.05$** ↑ (36/42)	$P = 0.10$ ← (19/33)	$P = 0.79$ ← (21/37)	$P = 0.14$ ↑ (25/41)	$P < 0.05$** ← (15/16)
Ditch 5 near mouth	-	-	-	-	-	$P < 0.05$** ← (32/34)	$P < 0.05$** ← (28/37)	$P < 0.05$** ← (37/42)	$P < 0.05$** ← (15/16)
Ditch 3 near mouth	-	-	-	-	-	-	$P < 0.05$** ↑ (22/34)	$P = 0.38$ ↑ (17/31)	<u>$P = 0.24$**</u> (6/7)
Boat launch creek	-	-	-	-	-	-	-	$P = 0.25$ ↑ (18/35)	<u>$P = 0.11$**</u> ← (7/9)
Coffee Creek at 114 Road	-	-	-	-	-	-	-	-	$P < 0.05$** ← (17/18)
Forested "control"	-	-	-	-	-	-	-	-	-

Plate 13. Severe riparian damage in upper watershed. Note erosion to edge of tributary.

Plate 14. Severe riparian damage in upper watershed.

TABLE 19. WILCOXIN-PAIRED-SAMPLE MATRIX FOR TP COMPARING TRIBUTARY STREAMS/DITCHES AND CONTROL STREAM. CELLS WITH ASTERISKS (**) INDICATE STATISTICALLY SIGNIFICANT COMPARISONS. CELLS WITH UNDERLINED *P*-VALUES AND ASTERISKS ARE NOT, SIGNIFICANTLY DIFFERENT AT α 0.05. HOWEVER, IT IS BELIEVED THAT THEY WOULD BE SIGNIFICANT IF A LARGER NUMBER OF SAMPLES WERE TAKEN. THE DENOMINATOR OF THE BRACKETED FRACTION IN EACH CELL IS THE NUMBER OF COMPARISONS MADE (*N*). THE NUMERATOR IS THE NUMBER OF WILCOXIN RANKS WON BY THE TRIBUTARY THAT THE ARROW IS POINTING TO. THE ARROW POINTS TO THE TRIBUTARY HAVING THE GREATEST NUMBER OF WINNING RANKS (I.E., THE HIGHEST TP CONCENTRATION).

	West Stoddart (E207904)	East Stoddart (E207903)	281st Road (E207902)	Beatton Park (E207896)	Ditch 5 @ mouth (E207898)	Ditch 3 @ mouth (E207899)	Boat Launch (E207900)	Coffee @ 114 Road crossing (E207901)	Forested "control" (E207906)
West Stoddart	-	$P < 0.05$** ← (32/46)	$P = 0.42$ ← (21/37)	$P < 0.05$** ← (29/35)	$P = 0.12$** ← (5/6)	$P = 0.64$ ← (14/26)	$P = 0.11$** ← (22/29)	$P = 0.08$ ← 24/39)	$P = 0.66$ ← (10/16)
East Stoddart	-	-	$P < 0.05$** ↑ (23/40)	$P = 0.50$ ← (38)	$P = 0.48$ = (8 pairs)	$P < 0.05$** ↑ (19/29)	$P = 0.18$ ↑ (17/32)	$P < 0.05$** ↑ (25/42)	$P = 0.21$ ← (10/17)
281st Road Creek	-	-	-	$P < 0.05$** ← (25/37)	$P = 0.07$** ← (7/8)	$P < 0.05$** ← (24/31)	$P < 0.05$** ← (23/34)	$P = 0.66$ ← (19/38)	$P = 0.39$ ← (7/12)
Beatton Park	-	-	-	-	$P = 0.80$** ↑ (7/10)	$P = 0.14$ ↑ (21/33)	$P = 0.25$ = (36 pairs)	$P < 0.05$** ↑ (32/39)	$P = 0.12$** ↑ (11/14)
Ditch 5 near mouth	-	-	-	-	-	$P = 0.86$ ← (6/11)	$P = 0.96$ ↑ (6/11)	$P = 0.14$** ↑ (6/9)	Unable to compare (two pairs)
Ditch 3 near mouth	-	-	-	-	-	-	$P = 0.94$ ← (19/34)	$P = 0.29$ ↑ (19/30)	$P = 0.74$ ← (4/7)
Boat launch creek	-	-	-	-	-	-	-	$P = 0.12$ ↑ (22/33)	$P = 0.78$ = (8 pairs)
Coffee Creek at 114 Road	-	-	-	-	-	-	-	-	$P = 0.50$ ← (9/16)
Forested "control"	-	-	-	-	-	-	-	-	-

5.6.3 Phosphorus Budgets

Correlative Approach

Lake P content (1,731–9,507 kg) and P concentrations (0.04–0.10 $mg \cdot L^{-1}$) averaged 4,476 ± 468 kg (mean ± 1S.E., $n = 19$) and 0.07 ± 0.00 $mg \cdot L^{-1}$ ($n = 19$), respectively, during the months April to July (Figures 14 and 15). Lake P content (4,658–11,113 kg) and P concentrations (0.09–0.21 $mg \cdot L^{-1}$) were significantly ($P < 0.05$, t-test) higher during the late summer and early autumn months (August and September), when they averaged 8,348 ± 543 kg and 0.15 ± 0.01 $mg \cdot L^{-1}$ (Figures 14 and 15).

The finding that lake P content and P concentrations peaked in the late summer and early autumn (during the descending hydrograph and basal flow periods), suggests that P loads to the lake water came primarily from internal sources. It would seem that internal P loading rates accelerate during periods of stratification, when hypolimnetic oxygen concentrations approach zero. The P that accumulates in the hypolimnion upwells when stratification breaks down, increasing P concentrations throughout the water column (Figures 2-4). It seems highly likely that this upwelling process triggers the late summer and early autumn algal blooms observed in Charlie Lake (Figure 16a). P upwelling also appears to occur during spring overturn when winter deep-water P (lake is inversely stratified during ice-covered periods) is carried to the surface with circulating water (Figures 2-4).

An alternative explanation to the observation that lake P content and P concentrations peaked post-freshet is that P concentrations in the tributaries may have been extremely high in August and September; however, it is highly unlikely that the late summer and early autumn loads to the lake water were the result of tributary loadings, as flows in the tributaries were exceedingly low (approaching nil) after freshet.

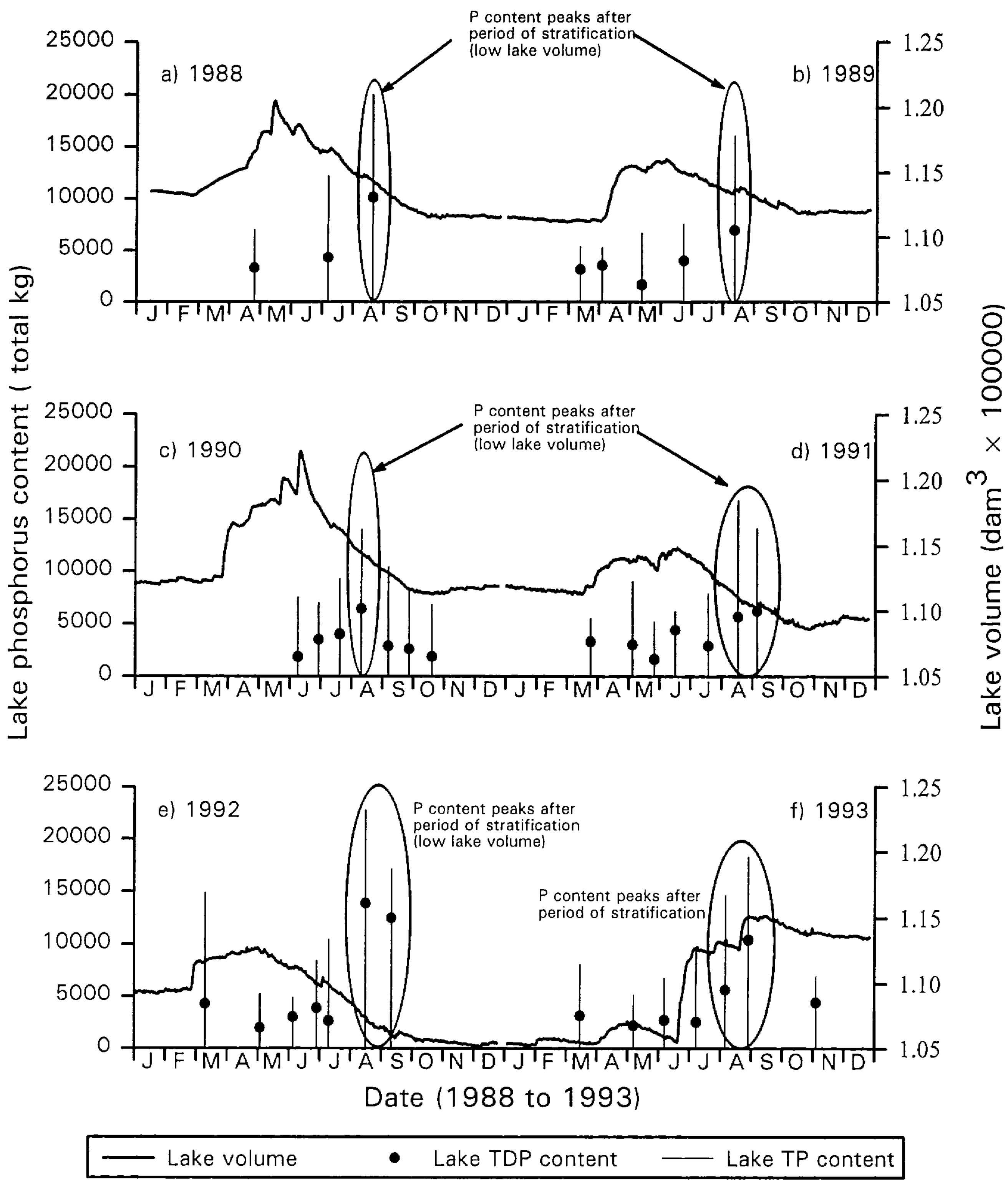

FIGURE 14. LAKE P CONTENT (TOTAL KGS) IN RELATION TO LAKE VOLUME (1988 TO 1993).

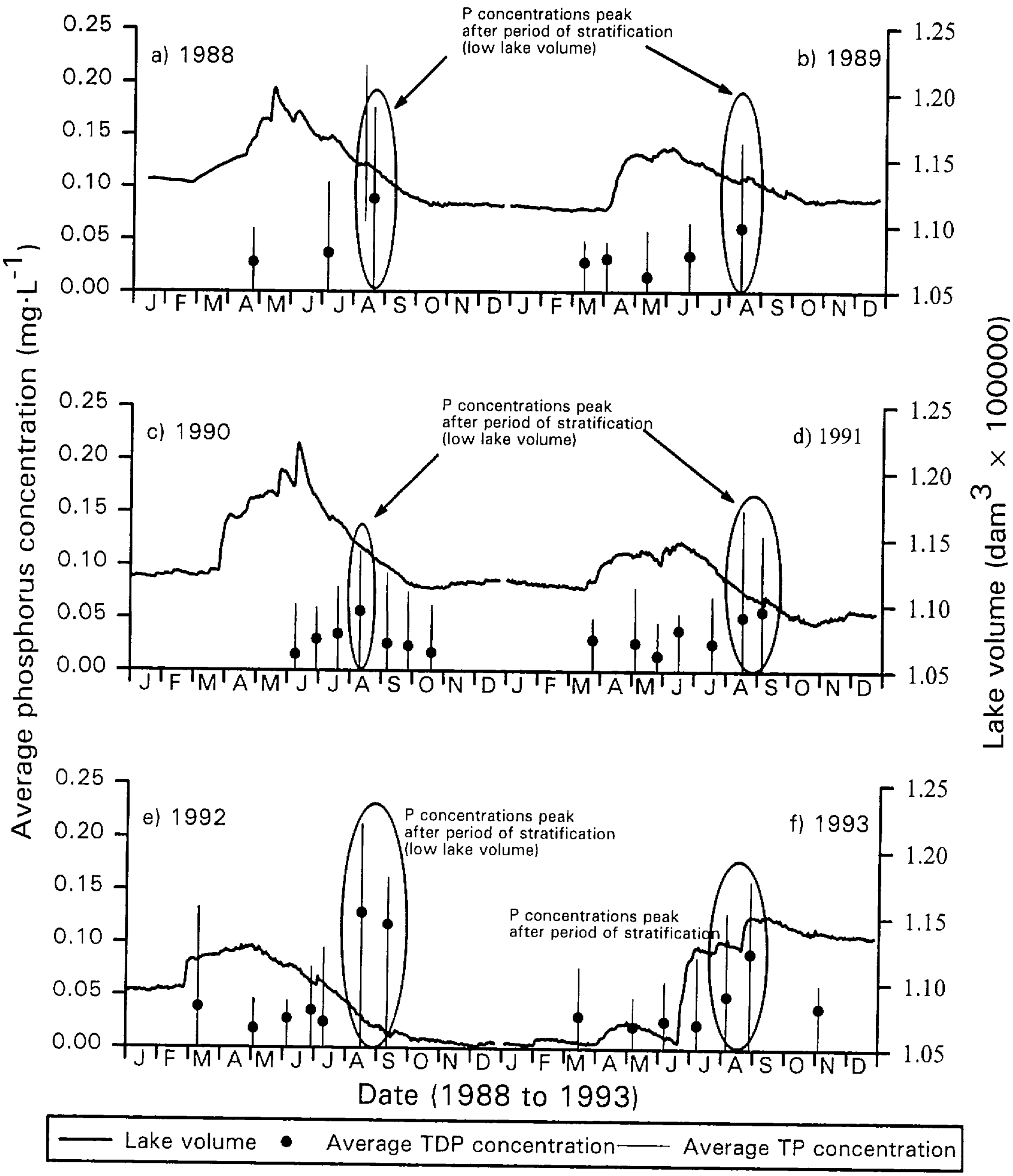

FIGURE 15. AVERAGE LAKE TP CONCENTRATION (ASSUMING TOTAL MIXING) IN RELATION TO LAKE VOLUME (1988 TO 1993).

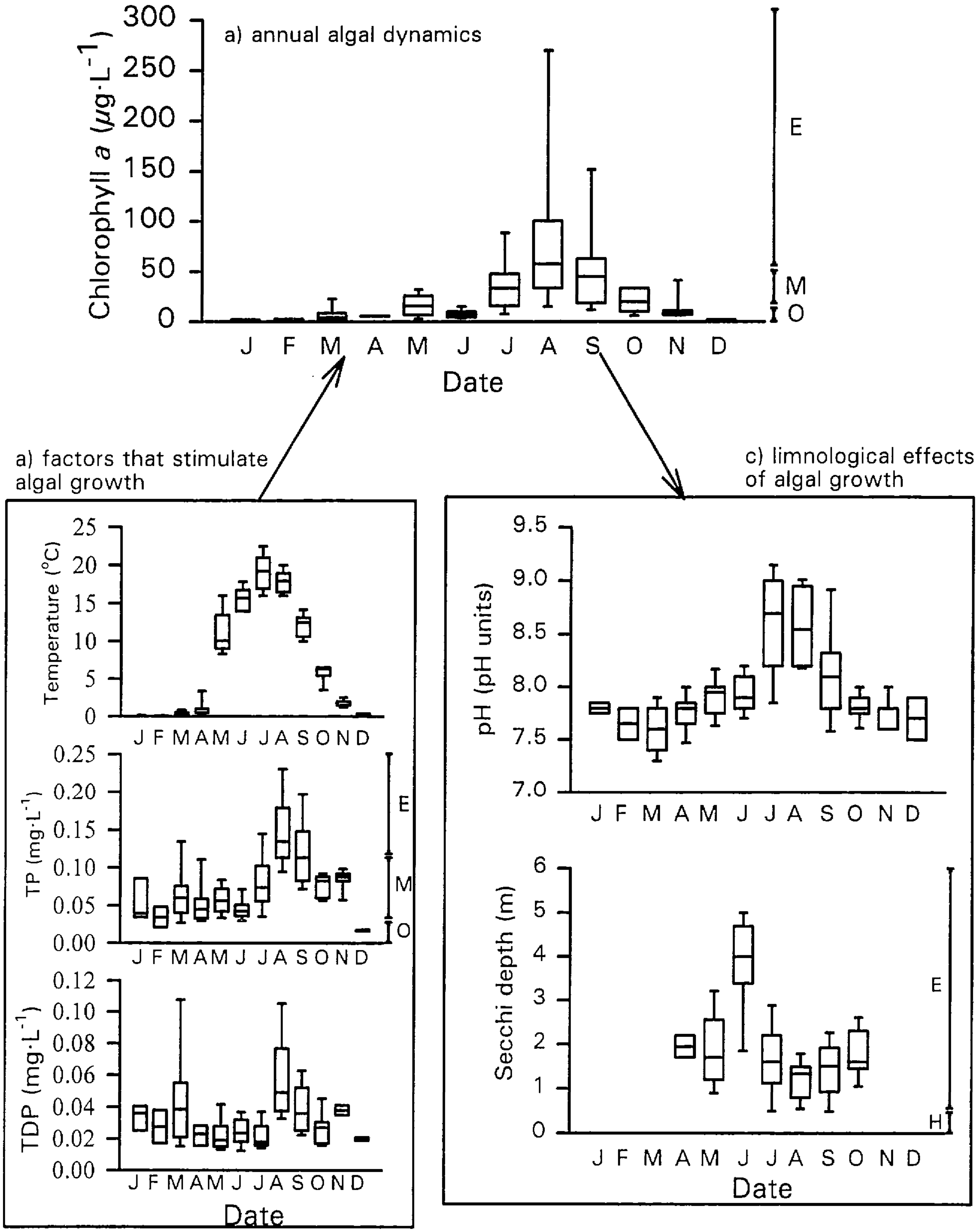

FIGURE 16. CORRELATES OF ALGAL ABUNDANCE (SURFACE WATER DATA). TOP BAR IS THE 90TH PERCENTILE, TOP OF VERTICAL RECTANGLE IS 75TH PERCENTILE, MIDDLE BAR IN RECTANGLE IS 50TH PERCENTILE, BOTTOM OF RECTANGLE IS 25TH PERCENTILE, BOTTOM BAR IS 19TH PERCENTILE. H = HYPEREUTROPHIC, E = EUTROPHIC, M = MESOTROPHIC, O = OLIGOTROPHIC (VOLLENWEIDER 1979).

Mass-Balance Approach

Under Scenario 2 conditions (50^{th} percentile tributary plus groundwater P concentrations), tributaries and groundwater (combined) contributed about 24% (13,350 kg P) of the P to the April 1988 to September 1991 P budget (Tables 20 and 21). Internal and atmospheric loads contributed, respectively, 72% (39,681 kg) and 4% (2,013 kg) of the P to the Scenario-2 P budget. As illustrated by Scenario 3 , the relative importance of tributary plus groundwater P loads increases dramatically as tributary plus groundwater P concentrations rise. Thus, tributaries and groundwater would contribute about 41% of the P to the April to September P budget if typical tributary plus groundwater P concentrations rose to 0.36 $mg \cdot L^{-1}$ (75^{th} percentile of observed values). At first glance it would seem that tributary plus groundwater P loads would not have significant control over lake P dynamics (because internal loading rates are so much greater); however, the results of Scenario 1 and 2 examples show, over the long term, that internal P loads are more-or-less offset by internal P consumption (sedimentation) (Table 21). Therefore, any P gains in the lake would be the result of external loadings.

The finding that internal P loading rates accelerate in July and August after deep water oxygen deficits develop is consistent with the results of the "Correlative Approach" described above. The mass-balance computations indicate that between 4,616 and 8,890 kg P ($\overline{X}$ = 7,307 kg P, n = 12) were consistently loaded to the lake during every July to August sampling period between April 1988 and September 1991 (Table 21). The late-summer internal P loadings occurred just prior to beginning to the late-summer Cyanophyte blooms, suggesting that late-summer Cyanophyte production is dependent on within-lake P sources. As shown by the boxes in the "Net internal" P consumption column of Table 21, internally loaded P appears to be scavenged from the water column after the cessation of late summer algal blooms (probably within algal corpses and/or associated organic material).

TABLE 20. WATER BUDGET OF EACH SAMPLING PERIOD BETWEEN 1988 AND 1991.

Time Interval	Δ lake volume (dam^3)	Water to lake (dam^3)			Water from Lake (dam^3)				
		Precipitation	All other sources[a]	Total	Lower Stoddart Creek	Evaporation	City of Fort St. John	Anderson Exploration Ltd.	Total
Apr. 28/88 - Jul. 11/88	+45	2,806	12,360	15,166	8,631	5,829	490	171	15,121
Jul. 11/88 - Aug. 25/88	-2,376	1,406	911	2,317	761	3,530	298	104	4,693
Aug. 25/88 - Mar. 15/89	-2,981	1,056	negligible	1,056	310	1,650	1,339	466	3,765
Mar. 15/89 - Apr. 6/89	+61	283	336	619	16	345	146	51	558
Apr. 6/89 - May 16/89	+3,784	209	7,590	7,799	1,705	1,953	265	92	4,015
May 16/89 - Jun. 27/89	-136	1,664	2,433	4,097	484	3,374	278	97	4,233
Jun. 27/89 - Aug. 17/89	-1,377	2,079	1,710	3,789	466	4,244	338	118	5,166
Aug. 17/89 - Jun. 12/90	+7,189	5,979	25,496	31,475	15,035	6,580	1,981	690	24,286
Jun. 12/90 - Jul. 3/90	-2,845	830	3,548	4,378	5,217	1,819	139	48	7,223
Jul. 3/90 - Jul. 24/90	-1,756	650	2,017	2,667	2,322	1,914	139	48	4,423
Jul. 24/90 - Aug. 15/90	-1,831	139	750	889	861	1,662	146	51	2,720
Aug. 15/90 - Sept. 11/90	-1,543	654	negligible	654	176	1,414	179	62	1,831
Sept. 11/90 - Oct. 2/90	-1,211	139	negligible	139	52	565	139	48	804
Oct. 2/90 - Oct. 25/90	-363	675	negligible	675	13	120	152	53	338
Oct. 25/90 - Mar. 27/91	+578	661	3,022	2,361	78	338	1,014	353	1,783
Mar. 27/91 - May 8/91	+1,952	29	4,209	4,238	280	1,631	278	97	2,286
May 8/91 - May 30/91	-378	426	1,104	1,530	163	1,548	146	51	1,908
May 30/91 - Jun. 20/91	+1,286	1,486	1,950	3,436	175	1,788	139	48	2,150
Jun. 20/91 - Jul. 23/91	-1,650	690	1,170	1,860	260	2,955	219	76	3,510
Jul. 23/91 - Aug. 22/91	-2,043	217	332	549	93	2,231	199	69	2,592
Aug. 22/91 - Sept. 10/91	-787	481	1,328	1,809	17	2,409	126	44	2,596

[a]tributaries plus groundwater.

TABLE 21. ESTIMATED P BUDGET (APRIL 1988 TO SEPTEMBER 1991). BOXES IN "NET INTERNAL" LOADING COLUMN MARK LATE-SUMMER P INPUTS THAT TRIGGER LATE-SUMMER EARLY-AUTUMN CYANOPHYTE BLOOM. CONNECTED BOXES IN "NET INTERNAL" CONSUMPTION COLUMN MARK PERIODS IMMEDIATELY AFTER LATE-SUMMER EARLY-AUTUMN CYANOPHYTE BLOOM WHEN P IS SCAVENGED BACK TO THE BOTTOM.

Time interval	Δ Lake P (kg)	P loading (kg)			P consumption (kg)			Dominant P dynamic
		Atmosphere	Tributaries + groundwater	"Net" internal	Major withdrawals	Lower Stoddart Creek	"Net" internal	
I. SCENARIO 1 (TRIBUTARY AND GROUNDWATER CONCENTRATIONS = 0.12 mg·L^{-1} (25TH PERCENTILE OF ALL MEASURED TRIBUTARY P CONCENTRATIONS))								
Apr. 28/88 - Jul. 11/88	+5,149	185	1,483	3,950	50	419	0	Internal release
Jul. 11/88 - Aug. 25/88	+7,859	105	109	7,766	31	90	0	Internal release
Aug. 25/88 - Mar. 15/89	-14,647	140	negligible	0	141	36	14,610	Internal consumption
Mar. 15/89 - Apr. 6/89	-109	32	40	0	15	1	165	Internal consumption
Apr. 6/89 - May 16/89	+1,406	62	911	473	27	113	0	Tributaries & groundwater
May 16/89 - Jun. 27/89	+902	106	292	557	29	24	0	Internal release
Jun. 27/89 - Aug. 17/89	+8,468	133	205	8,218	35	53	0	Internal release
Aug. 17/89 - Jun. 12/90	-8,504	432	3,060	0	204	1,049	10,743	Internal consumption
Jun. 12/90 - Jul. 3/90	-516	54	426	0	14	279	703	Internal consumption
Jul. 3/90 - Jul. 24/90	+2,216	49	242	2,222	14	283	0	Internal release
Jul. 24/90 - Aug. 15/90	+4,801	35	90	4,796	15	105	0	Internal release
Aug. 15/90 - Sept. 11/90	-3,655	58	negligible	0	18	21	3,674	Internal consumption
Sept. 11/90 - Oct. 2/90	-2,056	33	negligible	0	14	6	2,069	Internal consumption
Oct. 2/90 - Oct. 25/90	-1,491	36	negligible	0	16	2	1,509	Internal consumption
Oct. 25/90 - Mar. 27/91	-1,337	218	363	0	105	8	1,805	Internal consumption
Mar. 27/91 - May 8/91	+3,514	60	505	2,998	28	21	0	Internal release
May 8/91 - May 30/91	-3,829	44	132	0	15	8	3,982	Internal consumption
May 30/91 - Jun. 20/91	+972	74	234	687	14	9	0	Internal release
Jun. 20/91 - Jul. 23/91	+1,687	67	140	1,525	22	23	0	Internal release
Jul. 23/91 - Aug. 22/91	+8,947	49	40	8,890	21	11	0	Internal release
Aug. 22/91 - Sept. 10/91	-2,676	41	159	0	13	2	2,761	Internal consumption
TOTAL	-	2,013	8,431	42,082	841	2,563	42,021	Loading > consumption

Time interval	Δ Lake P (kg)	P loading (kg)			P consumption (kg)			Dominant P dynamic
		Atmosphere	Tributaries + groundwater	"Net" internal	Major withdrawals	Lower Stoddart Creek	"Net" internal	
II. SCENARIO 2 (TRIBUTARY AND GROUNDWATER CONCENTRATIONS = 0.19 mg·L^{-1} (50TH PERCENTILE OF ALL MEASURED TRIBUTARY P CONCENTRATIONS))								
Apr. 28/88 - Jul. 11/88	+5,149	185	2,348	3,085	50	419	0	Internal release
Jul. 11/88 - Aug. 25/88	+7,859	105	173	7,702	31	90	0	Internal release
Aug. 25/88 - Mar. 15/89	-14,647	140	negligible	0	141	36	14,610	Internal consumption
Mar. 15/89 - Apr. 6/89	-109	32	64	0	15	1	189	Internal consumption
Apr. 6/89 - May 16/89	+1,406	62	1,442	42	27	113	0	Tributaries & groundwater
May 16/89 - Jun. 27/89	+902	106	462	387	29	24	0	Tributaries & groundwater
Jun. 27/89 - Aug. 17/89	+8,468	133	325	8,098	35	53	0	Internal release
Aug. 17/89 - Jun. 12/90	-8,504	432	4,844	0	204	1,049	12,527	Internal consumption
Jun. 12/90 - Jul. 3/90	-516	54	674	0	14	279	951	Internal consumption
Jul. 3/90 - Jul. 24/90	+2,216	49	383	2,081	14	283	0	Internal release
Jul. 24/90 - Aug. 15/90	+4,801	35	143	4,743	15	105	0	Internal release
Aug. 15/90 - Sept. 11/90	-3,655	58	negligible	0	18	21	3,674	Internal consumption
Sept. 11/90 - Oct. 2/90	-2,056	33	negligible	0	14	6	2,069	Internal consumption
Oct. 2/90 - Oct. 25/90	-1,491	36	negligible	0	16	2	1,509	Internal consumption
Oct. 25/90 - Mar. 27/91	-1,337	218	574	0	105	8	2,016	Internal consumption
Mar. 27/91 - May 8/91	+3,514	60	800	2,703	28	21	0	Internal release
May 8/91 - May 30/91	-3,829	44	210	0	15	8	4,060	Internal consumption
May 30/91 - Jun. 20/91	+972	74	371	530	14	9	0	Internal release
Jun. 20/91 - Jul. 23/91	+1,687	67	222	1,443	22	23	0	Internal release
Jul. 23/91 - Aug. 22/91	+8,947	49	63	8,867	21	11	0	Internal release
Aug. 22/91 - Sept. 10/91	-2,676	41	252	0	13	2	2,954	Internal consumption
TOTAL	-	2,013	13,350	39,681	841	2,563	44,559	Loading > consumption

Time interval	Δ Lake P (kg)	P loading (kg)			P consumption (kg)			Dominant P dynamic
		Atmosphere	Tributaries + groundwater	"Net" internal	Major withdrawals	Lower Stoddart Creek	"Net" internal	
III. SCENARIO 3 (TRIBUTARY AND GROUNDWATER CONCENTRATIONS = 0.36 mg·L^{-1} (75TH PERCENTILE OF ALL MEASURED TRIBUTARY P CONCENTRATIONS))								
Apr. 28/88 - Jul. 11/88	+5,149	185	4,450	983	50	419	0	Tributaries & groundwater
Jul. 11/88 - Aug. 25/88	+7,859	105	328	7,547	31	90	0	Internal release
Aug. 25/88 - Mar. 15/89	-14,647	140	negligible	0	141	36	14,610	Internal consumption
Mar. 15/89 - Apr. 6/89	-109	32	121	0	15	1	246	Internal consumption
Apr. 6/89 - May 16/89	+1,406	62	2,732	0	27	113	1,248	Tributaries & groundwater
May 16/89 - Jun. 27/89	+902	106	876	0	29	24	27	Tributaries & groundwater
Jun. 27/89 - Aug. 17/89	+8,468	133	616	7,631	35	53	0	Internal release
Aug. 17/89 - Jun. 12/90	-8,504	432	9,179	0	204	1,049	16,862	Internal consumption
Jun. 12/90 - Jul. 3/90	-516	54	1,277	0	14	279	1,554	Internal consumption
Jul. 3/90 - Jul. 24/90	+2,216	49	726	1,740	14	283	0	Internal release
Jul. 24/90 - Aug. 15/90	+4,801	35	270	4,616	15	105	0	Internal release
Aug. 15/90 - Sept. 11/90	-3,655	58	negligible	0	18	21	3,674	Internal consumption
Sept. 11/90 - Oct. 2/90	-2,056	33	negligible	0	14	6	2,069	Internal consumption
Oct. 2/90 - Oct. 25/90	-1,491	36	negligible	0	16	2	1509	Internal consumption
Oct. 25/90 - Mar. 27/91	-1,337	218	1,088	0	105	8	2,530	Internal consumption
Mar. 27/91 - May 8/91	+3,514	60	1,515	1,988	28	21	0	Internal release
May 8/91 - May 30/91	-3,829	44	397	0	15	8	4,247	Internal consumption
May 30/91 - Jun. 20/91	+972	74	702	219	14	9	0	Tributaries & groundwater
Jun. 20/91 - Jul. 23/91	+1,687	67	421	1,244	22	23	0	Internal release
Jul. 23/91 - Aug. 22/91	+8,947	49	120	8,810	21	11	0	Internal release
Aug. 22/91 - Sept. 10/91	-2,676	41	478	0	13	2	3,180	Internal consumption
TOTAL	-	2,013	25,296	34,778	841	2,563	51,756	Loading > consumption

Table 21 (Cont.)

5.6.4 Correlates of Algal Abundance (1974 to 1995 data)

Surface chlorophyll *a* concentrations ranged from 1 to 72 $\mu g \cdot L^{-1}$ (median = 8 $\mu g \cdot L^{-1}$, n = 86) during the months October to June, and from 5 to 412 $\mu g \cdot L^{-1}$ (median = 43 $\mu g \cdot L^{-1}$, n = 80) during the months July to September, with peak concentrations being observed in August (Figure 16).

The simple spring overturn TP-chlorophyll *a* concept (Dillon and Rigler 1974; Prepas and Trew 1983) accurately predicted the trophic status of Charlie Lake (Figure 17). However, the data suggest that spring overturn TP concentrations do not have mechanistic control over algal production. Rather, the algal community appears to bloom in response to mid-summer P inputs and, possibly, warm water temperatures (Figure 16). The sources of mid-summer P have not been quantified; however, it likely comes from within-lake sources (Section 5.6.3). Bottom sediments (e.g., Zicker *et al.* 1956; Mortimer 1971; Kirchner and Dillon 1975; Larsen and Mercier 1976; Sonzogni *et al.* 1976; Larsen *et al.* 1981; Nürnberg 1984; Riley and Prepas 1984) and biota (e.g., Carpenter 1980; Landers 1982; Gabrielson *et al.* 1984; Rørslett *et al.* 1986) may be significant within-lake P reservoirs.

Although the observed relationship between P, temperature and algal biomass is strictly correlative, as opposed to mechanistic, it is highly probable that mid-summer P inputs to the lake stimulates algal production. Conversely, algal production seems to increase surface pH (consumption of inorganic carbon) and decrease water clarity (Figure 16). While P availability appears to influence algal abundance in Charlie Lake, seasonal changes in the TDP:TP ratio suggest that algal production has an influence on P speciation. As indicated on Figure 18, most of the surface water P was in a dissolved form during the months of January and February, when algal production was low. As algal production increased during the spring and summer months, the dissolved forms were transformed into particulate forms (biological matter). The dissolved forms were then released back into the water in the early winter as the primary producers perished and decomposed (Figure 18).

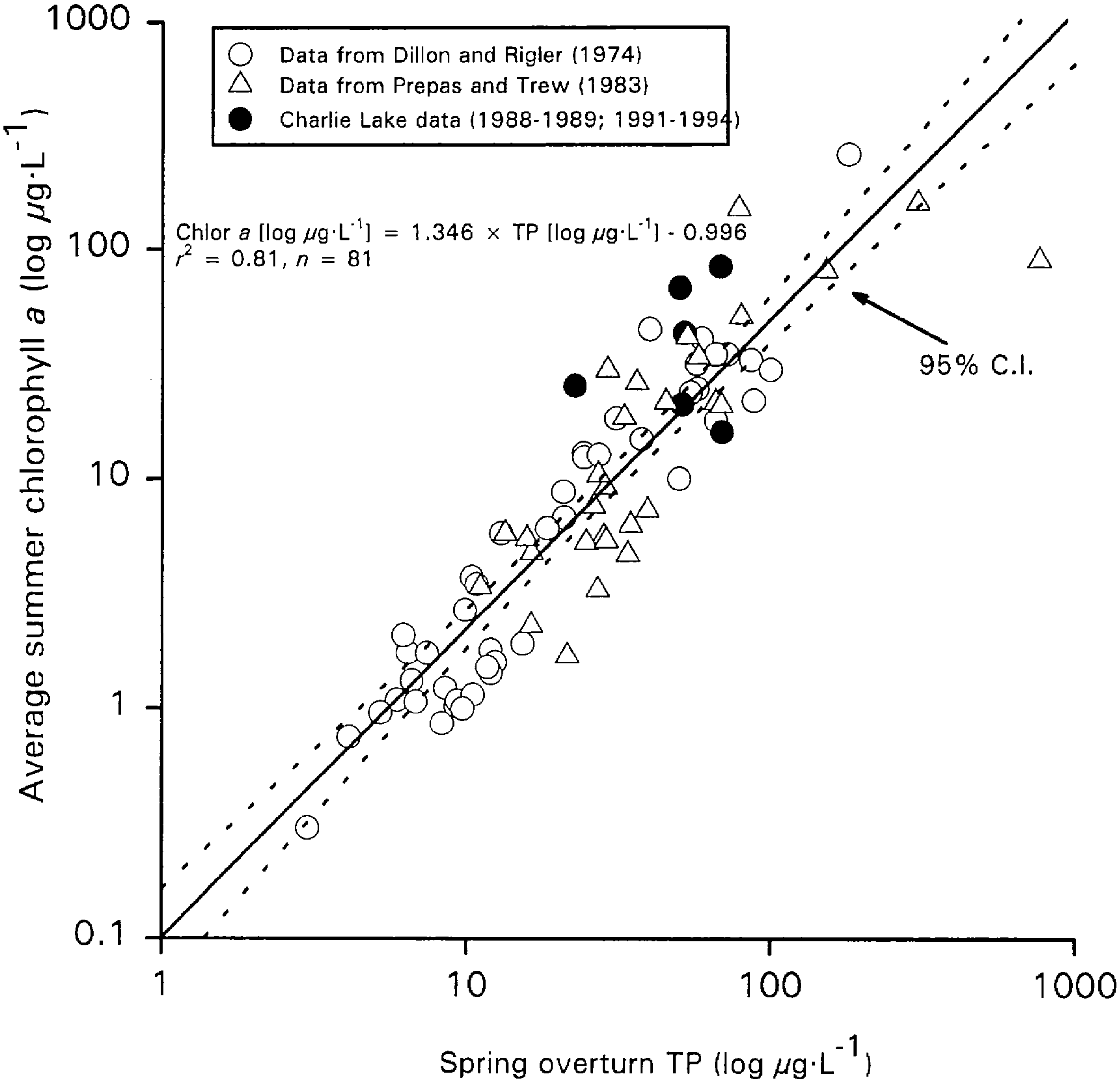

FIGURE 17. RELATIONSHIP BETWEEN SPRING-OVERTURN TP CONCENTRATION AND AVERAGE SUMMER ALGAL BIOMASS.

FIGURE 18. CHARLIE LAKE SURFACE WATER TDP:TP RATIOS (1974 TO 1995). NUMBERS ABOVE BARS ARE SAMPLE SIZES.

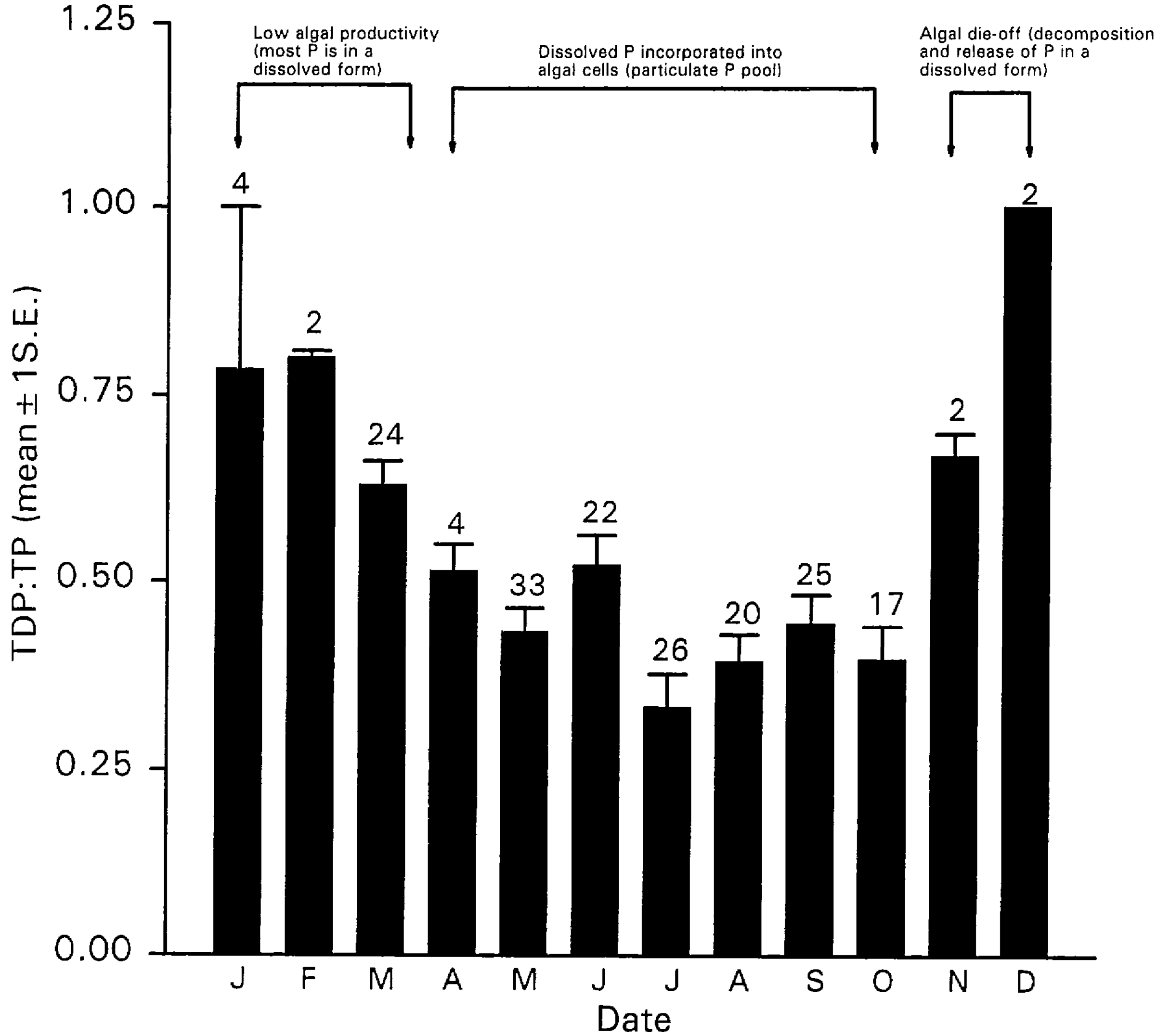

FIGURE 18. CHARLIE LAKE SURFACE WATER TDP:TP RATIOS (1974 TO 1995).

5.6.5 Water Quality Criteria (Provincial Standards)

The tributary data are summarized in Table 17. British Columbia's water quality standards were satisfied for all measured chemical parameters. However, excessively high fecal coliform concentrations were observed in all tributary streams and the control stream. The geometric mean coliform concentration ranged from 9 $CFU \cdot cL^{-1}$ (East Stoddart Creek) to 42 $CFU \cdot cL^{-1}$ (Ditch #3 and 281st Road Creek). The 90th percentile coliform concentration ranged from 109 (control stream) to 8790 $CFU \cdot cL^{-1}$ (281st Road Creek). According to the fecal coliform standards presented in Nagpal *et al.* (1995), which are based upon geometric mean and 90th percentile concentrations, the tributary streams and control stream do not pose a health threat to recreational users or to land irrigators. The stream waters should not, however, be consumed by humans or livestock, or be used to process foods unless they are treated (partial) and disinfected beforehand. People living in the watershed should be advised that they could become ill if they drink tributary water directly, wash food with tributary water or fail to wash their hands after touching tributary water.

Runoff from livestock arenas may be contributing to the apparent fecal coliform problem. To alleviate this problem, riparian zones should be fenced to keep livestock away from the tributaries. Ranchers may also be advised to drain their livestock arenas into collecting ponds where runoff waters can be treated prior to allowing them to enter stream courses. While leaking septic mounds may have contributed to coliform contamination in the past, most residences are now connected to a sewage trunk system that removes domestic waste from the watershed.

The lake data are summarized in Table 22. All total organic C concentrations (n = 203) exceeded 4 $mg \cdot L^{-1}$. Therefore, "disinfection by-products" (including trihalomethanes) could be formed if chlorine is used in water treatment processes (Nagpal *et al.* 1995). Given that colour ranged from 15 to 60 (TCU), the lake water may not be visually appealing to those using the lake as a water supply (i.e., it has a brownish hue). All TP and chlorophyll *a* concentrations exceeded the drinking water, aquatic life and recreation standards. The dense blue-green algae populations may liberate neuro- and/or hepato-toxins into the water (see Carmichael and Gorham 1977; DeMott *et al.* 1991; Kenefick *et al.* 1992; Kotak *et al.* 1993, 1995; Gilbert 1994; Reinikainen *et al.* 1994). Since fecal coliforms are present in the lake water, the lake water should not be consumed. Of the 559 pH readings taken, 74 exceeded the drinking water standard of 8.5. Twenty-one pH readings exceeded the recreational and aquatic life standards of 9. Since algal photosynthesis consumes inorganic carbon (the major determinant of pH in aqueous solutions), it is not surprising that most of the pH exceedences occurred in surface waters during periods of high algal abundance (July, August and September). Standards for iron (total) and manganese (total) were exceeded on one occasion (n = 4 for both parameters).

As predicted by Nordin and Pommen (1985), Charlie Lake develops deep-water D.O. deficits during periods of prolonged thermal stratification. Between overturns (spring and autumn), D.O. concentrations can approach zero in the 7- to 15-m depth stratum, and get as low as 3 $mg \cdot L^{-1}$ in the 3- to 7-m depth stratum. According to the standards reported in Nagpal *et al.* (1995), the observed deep-water D.O. deficits are severe enough to harm developing fish embryos; however, eggs are likely deposited above the zone of D.O. depletion (i.e., close to shore where wave action can provide a constant supply of D.O. to developing embryos).

TABLE 22. SUMMARY OF CHARLIE LAKE DATA (1974-1995).

Parameter	Sampling periods (sample size)	Mean/median	Range	# < LDC
Alkalinity, tot. to pH 4.5 (mg·L⁻¹)	1970s (201); 1980s (2); 1990s (30)	65 / 62	54-156	0
Aluminum, tot. (μg·L⁻¹)	1980s (4)	15.0 / 15.0	10.0-20.0	2
Antimony, tot. (μg·L⁻¹)	1980s (2)	-	-	2
Arsenic, tot. (μg·L⁻¹)	1980s (4)	-	<1-2	3
Barium, tot. (μg·L⁻¹)	1980s (2)	-	-	2
Cadmium, tot. (μg·L⁻¹)	1980s (4)	-	-	4
Calcium, diss. (mg·L⁻¹)	1970s (201)	16 / 16	14-20	0
Carbon, tot. inorganic (mg·L⁻¹)	1970s (90); 1980s (10)	15 / 14	10-30	0
Carbon, tot. organic (mg·L⁻¹)	1970s (200); 1980s (3)	15 / 15	10-45	0
Chloride, diss. (mg·L⁻¹)	1980s (4)	2.4 / 2.3	2.1-2.9	0
Chlorophyll *a* (μg·L⁻¹)	1970s (352); 1980s (23); 1990s (53)	33.6 / 14.3	1.0-413.0	0
Chromium, tot. (μg·L⁻¹)	1980s (4)	-	-	4
Cobalt, total (μg·L⁻¹)	1980s (4)	-	-	4
Coliforms, fecal				
MPN	1980s (5)	-	<2-4.5	4
CFU·cL⁻¹	1980s (3)	-	1-2	2
Color, true (TCU)	1970s (60)	27 / 30	15-60	0
Conductivity (μS·cm⁻¹)	1970s (201); 1980s (118); 1990s (241)	176 / 180	132-301	0
Copper, tot. (μg·L⁻¹)	1980s (4)	-	<10-30	3
Hardness, diss. (mg·L⁻¹)	1970s (185)	61 / 60	56-74	0
Iron, tot. (μg·L⁻¹)	1980s (4)	470 / -	30-1720	0
Magnesium, diss. (mg·L⁻¹)	1970s (201)	5 / 5	5-6	0
Manganese, tot. (μg·L⁻¹)	1980s (4)	218 / -	<10-820	2
Molybdenum, tot. (μg·L⁻¹)	1980s (4)	-	-	4
Nickel, tot. (μg·L⁻¹)	1980s (4)	-	-	4
Nitrogen, ammonia (mg·L⁻¹)	1970s (201); 1980s (109); 1990s (235)	0.094 / 0.048	<0.005-2.470	32
Nitrogen, nitrate (mg·L⁻¹)	1970s (123); 1980s (18); 1990s (12)	0.78 / <0.02	<0.02-0.38	6
Nitrogen, nitrite (mg·L⁻¹)	1970s (123); 1980s (33); 1990s (12)	0.005 / < 0.005	<0.005-0.011	19
Nitrogen, nitrite+nitrate (mg·L⁻¹)	1970s (123); 1980s (72); 1990s (231)	0.06 / < 0.02	<0.02-0.69	40
Nitrogen, tot. Kjeldahl (mg·L⁻¹)	1970s (184); 1980s (25); 1990s (87)	1.15 / 0.99	0.27-5.00	0
Nitrogen, tot. organic (mg·L⁻¹)	1970s (185); 1980s (14)	1.07 / 0.91	0.21-4.97	0
Oxygen, diss. (mg·L⁻¹)	1970s (132); 1980s (188); 1990s (880)	7.6 / 8.2	0-17	0
pH (pH units)	1970s (201); 1980s (117); 1990s (241)	8.0 / 9.3	7.1-9.3	0
Phosphorus, ortho (mg·L⁻¹)	1970s (67); 1980s (144); 1990s (415)	0.024 / 0.016	<0.003-0.152	56
Phosphorus, tot. (mg·L⁻¹)	1970s (202); 1980s (123); 1990s (289)	0.092 / 0.072	0.007-0.900	0
Phosphorus, tot. diss. (mg·L⁻¹)	1970s (14); 1980s (126); 1990s (416)	0.039 / 0.029	<0.003-0.304	1
Potassium, diss. (mg·L⁻¹)	1980s (4)	5.9 / -	5.0-6.4	0
Residues, filt. (mg·L⁻¹)	1970s (201); 1980s (11); 1990s (12)	115 / 112	100-150	0
Residues, non-filt. (mg·L⁻¹)	1970s (201); 1980s (9); 1990s (58)	7 / 4	1-97	0
Silica, diss. (μg·L⁻¹)	1980s (47); 1990s (95)	1972 / 1350	<500-16900	39
Sodium, diss. (mg·L⁻¹)	1980s (4)	6.6 / -	5.7-7.1	0
Sulfate, diss. (mg·L⁻¹)	1980s (4)	9.2 / -	8.1-10.2	0
Temperature (°C)	1970s (179); 1980s (181); 1990s (901)	10.6 / 12.4	0-22.5	0
Turbidity (NTU)	1970s (182)	4.4 / 3.8	0.5-26	0
Uranium, tot. (μg·L⁻¹)	1980s (2)	0.6 / -	0.2-0.9	0
Vanadium, tot. (μg·L⁻¹)	1980s (4)	-	-	4
Zinc, tot. (μg·L⁻¹)	1980s (4)	-	<10-40	2

5.6.6 Nutrient Export by Major Water Withdrawals

The Anderson Exploration Ltd. withdrawal currently exports, on average, 5 kg P per month (1-13 kg), or about 57 kg annually (Figure 19). Our estimates indicate that the withdrawal would export 169 kg P annually, or about 36 kg monthly, if water were taken from 8-m depth or deeper, where P is concentrated during periods of stratification (Figure 19a). By comparison, the City of Fort St. John withdrew about 162 kg P annually, or an average of 14 kg (3-37 kg) per month (Figure 19b). The city would have exported 3-times (about 473 kg) as much P if it would have drawn from hypolimnetic waters (Figure 19b).

By drawing dilute epilimnetic waters, the withdrawals are increasing lake water P concentrations. Our estimates indicate that the Anderson Exploration Ltd. withdrawal increases average lake water P concentrations (assuming complete mixing) by, on average, 10 $ng \cdot L^{-1}$ (-19 to +32 $ng \cdot L^{-1}$) per month (Figure 20a). The City of Fort St. John would have increased lake water P concentrations by 3-times this amount, with the average monthly increase being 30 $ng \cdot L^{-1}$ (-46 to +104 $ng \cdot L^{-1}$) (Figure 20b). If the withdrawal waters would have been taken from 8-m depth or deeper, the withdrawals would have actually benefited the lake by decreasing lake water P concentrations and, thereby, decreasing the intensity of algal blooms (Figure 20a,b).

5.6.7 Nutrient Export by Sewage Trunk System

The data indicate that the sewage trunk system removes about 261,000 m^3 of raw sewage from the Charlie Lake watershed annually (Plates 15-16). In-pipe N concentrations were about 3-times greater than the maximum concentrations observed in the lake (~ 5 $mg \cdot L^{-1}$) and tributaries (~ 4 $mg \cdot L^{-1}$ based on Stoddart Creek data; Table 17b), indicating that the previously-used ground disposal systems were concentrating N in the lake. Based on an average in-pipe TN concentration of 13.1 $mg \cdot L^{-1}$, it was estimated that the sewage trunk system removes 3000 kg N from the watershed annually.

In-pipe TP concentrations were often considerably greater than TP concentrations observed in the lake ($\bar{X} = 92$ $\mu g \cdot L^{-1}$), indicating that the ground disposal systems were also concentrating P in the lake. Based on an average in-pipe TP concentration of 132 $\mu g \cdot L^{-1}$, it was estimated that the sewage trunk system removes about 40 kg P from the watershed annually; with this export rate being the equivalent of about 0.4% of the external P loading rate or 0.1% of the total (i.e., including internal loading rates) P loading rate.

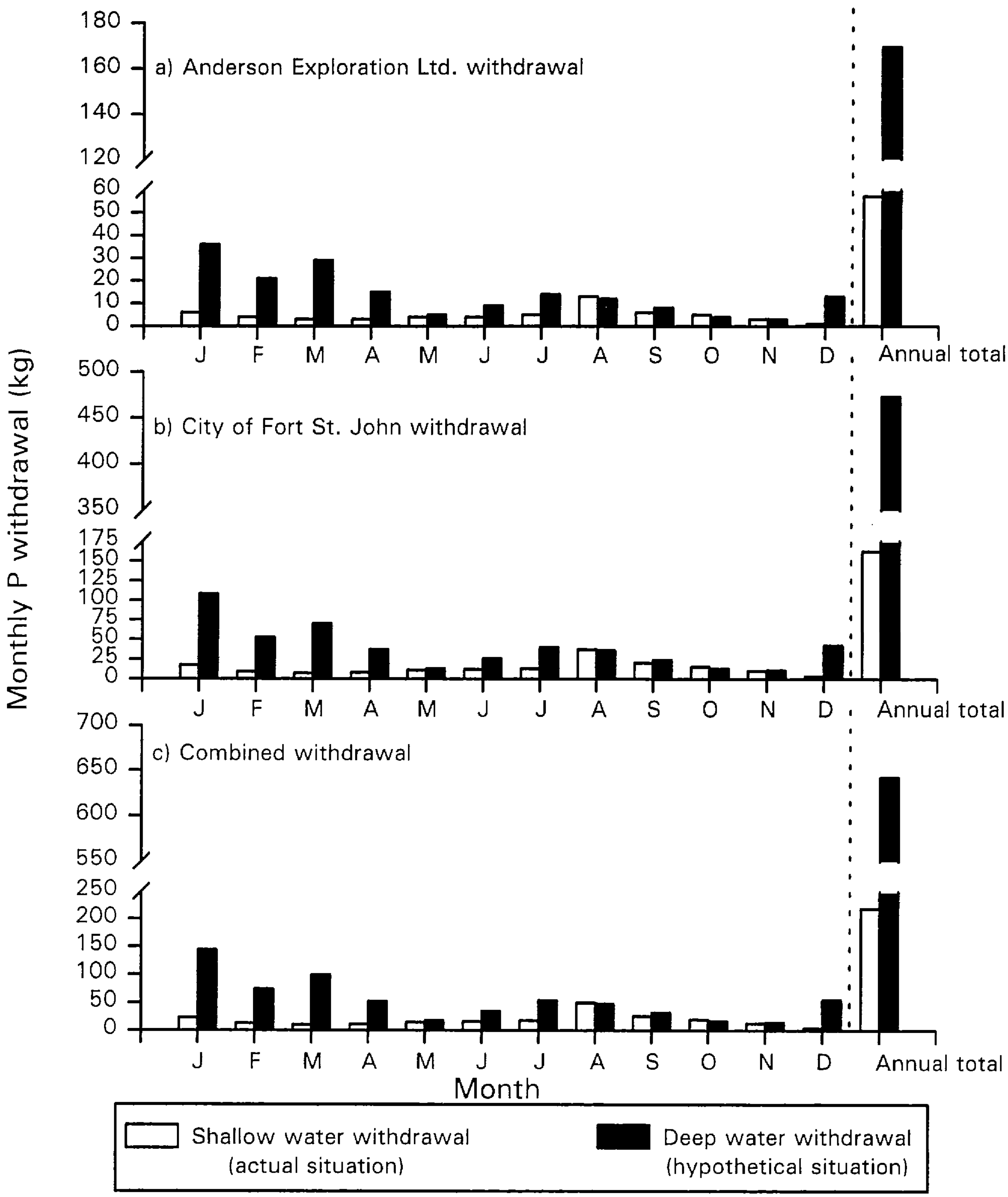

FIGURE 19. MONTHLY P EXPORTS BY MAJOR WATER WITHDRAWALS (ANDERSON EXPLORATION LTD. AND CITY OF FORT ST. JOHN) UNDER THE SCENARIO THAT THEY TAKE EPILIMNETIC OR METALIMNETIC WATERS (ACTUAL SITUATION) AND THE SCENARIO THAT THEY TAKE HYPOLIMNETIC WATER (HYPOTHETICAL SITUATION).

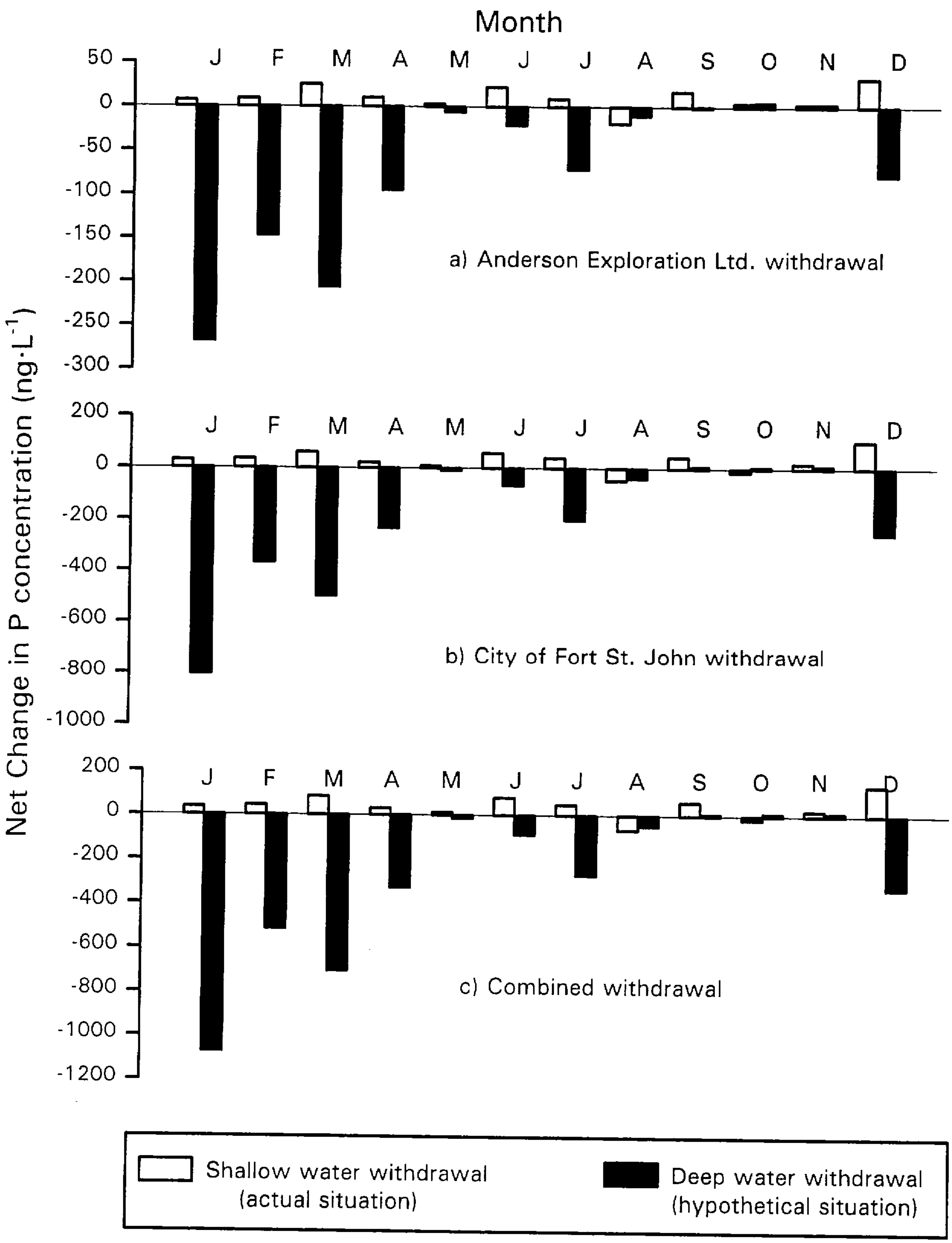

FIGURE 20. NET CHANGE IN LAKE TP CONCENTRATION INDUCED BY MAJOR WATER WITHDRAWALS (ANDERSON EXPLORATION LTD. AND CITY OF FORT ST. JOHN) UNDER THE SCENARIO THAT THEY TAKE EPILIMNETIC OR METALIMNETIC WATERS (ACTUAL SITUATION) AND THE SCENARIO THAT THEY TAKE HYPOLIMNETIC WATER (HYPOTHETICAL SITUATION).

Plate 15. Old mounded septic field in west-shoreline residential development, 1996

Plate 16. Residential ditch runoff before sewage trunk system was constructed. Note periphyton mat.

6.0 MAJOR CONCLUSIONS

The data support the following conclusions:

Public Interests and Concerns (PRRD 1996)

(1) summer fishing, motorized pleasure boating and residential living were the top-three public uses of Charlie Lake;
(2) the public ranked algal abundance (eutrophication) as their greatest environmental concern, with fishing quality and lakeshore development ranking second and third, respectively.

Agricultural Pesticide Residues (BCMELP 1992)

(1) no organophosphate, organochlorine or solvent soluble pesticide residues were observed in walleye flesh, lake-bottom sediments or in Stoddart Creek water (conclusion based on a very small sample size; thus, it may not be reliable).

Fish Parasites (Bangham and Adams 1954)

(1) *Diplostomulum*, *Eustrongylides*, *Glaridacris catostomi*, *Myxosporidia* and *Pomphorhynchus bulbocolli* were found in or on white suckers (n = 28);
(2) *Budoderina eucaliae*, *Gyrodactyloidea*, *Schistocephalus* and *Tetracotyle* were observed in brook sticklebacks.

Paleolimnological Investigations (Reavie *etal.* 1995b)

(1) it is highly probable that Charlie Lake is naturally eutrophic;
(2) there is evidence that human settlement in the watershed has increased P-loading rates to the lake, thereby increasing algal abundance.

Morphometry

Maximum Length = 15 km
Shoreline Length = 38 km
Surface Area = 19 km^2
Volume = 136,839 dam^3

Mean Depth = 7 m
Maximum Depth = 15 m
Relative Depth = 0.3
Watershed Area = 281 km^2 (not including lake area)

Basic Hydrology

(1) for the years 1989 to 1992, the annual inflow to Charlie Lake ranged from 18,588 dam^3 to 41,828 dam^3;
(2) Charlie Lake has an average flushing rate of 0.21 yr^{-1}, and an average retention rate of about 5 yr;
(3) Flows in Upper Stoddart Creek and Coffee Creek account for 80% of Charlie Lake's tributary inflow.

Phytoplankton Community Dynamics

(1) Chrysophyte, Cryptophyte and Dinoflagellate populations bloomed in early April;
(2) diatom populations bloomed soon after spring (Centrales) and autumn (Pennales) overturns;
(3) Chlorophyte populations bloomed in May, June (Chlorococcales and Tetrasporales) and July (Volvocales);
(4) Cyanophyte populations bloomed in August and September.

Nutrient Limitation

(1) three phases of nutrient limitation were observed: under-ice P-limitation (Phase 1, March to mid-May); open-water P-limitation (Phase 2, mid-May to mid-July); and open water no- or co-limitation (Phase 3, mid-July to late October);
(2) under-ice Chrysophyte and Dinophyte blooms were associated with the Phase 1 dynamic, Chlorophyte blooms with the Phase 2 dynamic, and Cyanophyte blooms with the Phase 3 dynamic.

Effects of Watershed Development

(1) TDP concentrations in the de-forested tributaries were greater than those observed in a forested control stream;
(2) TP concentrations in the de-forested tributaries were similar to those observed in a forested control stream.

Phosphorus Budgets

(1) P-dynamics in Charlie Lake are controlled primarily by within-lake P production and within-lake P consumption;
(2) late-summer and early-autumn P loads come primarily from within-lake sources;
(3) late-summer and early-autumn Cyanophyte blooms occur in response to pulses of internal P loading.

Correlates of Algal Abundance

(1) Surface chlorophyll *a* concentrations ranged from 1 to 72 $\mu g \cdot L^{-1}$ (median = 8 $\mu g \cdot L^{-1}$, n = 86) during the months October to June, and from 5 to 412 $\mu g \cdot L^{-1}$ (median = 43 $\mu g \cdot L^{-1}$, n = 80) during the months July to September, with peak concentrations being observed in August

(2) summer and autumn algal blooms appear to be triggered by internal P-loading events.

Water Quality Criteria (Provincial Standards)

(1) tributary waters satisfied all chemical water quality standards; fecal coliform concentrations in the tributaries generally exceeded the set standards;

(2) lake organic C concentrations exceeded 4 $mg \cdot L^{-1}$; therefore "disinfection by-products" could be formed if chlorine is used in water treatment processes;

(3) all lake TP and chlorophyll *a* concentrations exceeded provincial drinking water, aquatic life and recreation standards;

(4) seventy-four of 559 lake pH readings exceeded the drinking water standard of 8.5; 21 exceeded the recreational and aquatic life standards of 9;

(5) Charlie Lake develops deep-water D.O. deficits during periods of prolonged thermal stratification.

Nutrient Export by Major Water Withdrawals

(1) the Anderson Exploration Ltd. withdrawal currently exports, on average, 5 kg P per month, or about 57 kg annually;

(2) the City of Fort St. John withdrew about 162 kg P annually, or an average of 14 kg per month;

(3) by drawing dilute surface waters from the lake, both licensed withdrawals increase P concentrations in the lake (albeit by extremely small amounts).

Export by Sewage Trunk System

(1) the trunk system removes about 261,000 m^3 of raw sewage from the Charlie Lake watershed annually;

(2) the trunk system exports about 3000 kg P and 40 kg N from the watershed annually.

7.0 RECOMMENDATIONS

The Charlie Lake watershed is valued for its natural beauty and residential suitability, its angling and boating opportunities, and its fertile soils that support a thriving agricultural base. While the watershed supports several human endeavors, settlement in the region has not been without its environmental costs. Nutrient and soil losses from deforested uplands and riparian belts have accelerated natural eutrophication and sediment transport processes. Bacteria, nutrients and other substances have leached from residential developments into ditches that drain into Charlie Lake. In many instances, cattle have been given free access to major streams.

Some first steps towards improving the quality of Charlie Lake watercourses have already been taken. The sewage trunk system activated in the early 1990s removes about 261,000 m^3 raw sewage from the watershed annually. Prior to the implementation of the trunk system, domestic wastes were released to septic fields or mounds, many of which leached into ditches that drained directly into Charlie Lake. In 1996, the Peace River Regional District asked the public for their views on environmental issues relevant to the Charlie Lake watershed. The results of the survey indicated that people were primarily concerned about the severity of algal blooms in Charlie Lake and the potential for a decline in angling opportunities.

The recommendations presented here are intended to direct future restoration initiatives. They are based on our current understanding of the Charlie Lake watershed and are at least partly achievable with government's current limited resources. Our recommendations are:

1. Strategic Planning

Effort should be made to develop a formal long-term environmental management plan for the watershed that: (1) states and prioritizes objectives, (2) identifies actions that are required to meet the objectives, (3) prioritizes those actions based on their feasibility and cost-effectiveness, (4) identifies potential project funding sources, (5) provides timelines and (6) outlines how the success of restoration activities will be evaluated. Such a plan would direct restoration activities, assess and prioritize all potential restoration options and allow stakeholder input, thereby increasing the chances of success. Residents, industry, the agricultural community and the public should be permitted to state their interests and concerns as the plan is being developed.

The strategic plan should take into account the fact that virtually all restorative activities have associated financial costs (e.g., costs associated with riparian planting). Committees directing restoration activities (e.g., the Charlie Lake Technical Advisory Committee) may want to apply for grants from regional programs, such as: the Habitat Conservation Trust Fund (HCTF), Habitat Restoration and Salmonid Enhancement Program (HRSEP) and Fisheries Renewal British Columbia (FsRBC). Committees directing restorative activities could very well be comprised of volunteers from various stakeholder groups. However, it would be unrealistic to think the necessary work could all be accomplished by volunteers.

Funds awarded by any of the aforementioned programs could be used to pay technicians who undertake projects recommended by the committees. For example, funds could be used to pay a team of students to plant unvegetated riparian areas.

2. Program Implementation

Although much of the annual available phosphorus load to Charlie Lake is internally generated, a substantial portion is delivered from the surrounding watershed each year. It therefore makes reasonable sense to first control external nutrient source that may otherwise minimize the benefits to be realized by any in-lake management activities. This applies particularly to those external sources for which reductions would be more cost effective than for various within-lake control measures.

Committees directing future restoration activities may want to consider the following options. The cost-effectiveness of these or any other options should be determined so as to establish priorities and the order of implementation. The review of cost-effectiveness should be ongoing.

WATERSHED-BASED OPTIONS

Project concept	Rationale	Priority
Conduct a survey to identify restoration opportunities in the watershed (by air photo, ground and helicopter). Identity riparian disturbances, points of erosion, tributary obstructions, lakeshore disturbances, problems with road crossings and ditches, etc. Opportunities should be outlined in a report and presented to the public for their input.	The results of air photo and ground assessments will help the restoration program formulate solid objectives and quantify specific areas of concern; a report will convey watershed concerns to the public and solicit their input. The findings presented in the report could be summarized at a public workshop.	HIGH
Stabilize lakeshore and streamside erosional zones (on private and public lands) with vegetation (e.g., willows, alders and conifers).	Planted vegetation will decrease sediment loads to the tributaries and lake; Vegetation will increase nutrient buffering capacity of riparian zones and reduce nutrient loading to the lake; Vegetation will enhance local wildlife (perching bird) habitat. Could include effectiveness monitoring.	HIGH
Work with Ducks Unlimited, landowners and industry to develop wildfowl wetlands in the Charlie Lake watershed	In addition to habitat improvement, wetlands have the potential to absorb nutrients that would otherwise enter Charlie Lake. Must include effectiveness monitoring.	HIGH
Construct livestock corridors to keep cattle out of riparian zones (build stabilized watering areas away from shores and streams)	Will give riparian vegetation a chance to regenerate; a more productive riparian belt will increase nutrient assimilation and decrease erosion; a more diverse riparian belt will enhance aesthetic appeal and bird habitat.	HIGH
Treat runoff from livestock arenas in constructed dugouts or ponds (could treat with $CaCO_3$, for example).	Will decrease nutrient and bacterial loadings to tributaries and lake.	MOD
Oversee a limited study of historical land development in the Charlie Lake watershed and compare this to the 1991 paleolimnological study.	This may go to identify what land developments and uses might have contributed to trophic shifts in Charlie Lake.	LOW

LAKE-BASED (INTERNAL SOURCE CONTROL) OPTIONS

Project concept	Rationale	Priority
Initiate a Charlie Lake Volunteer Lake Monitoring Program	Although a reasonable historical data based exists for the lake, ongoing and frequent monitoring for a limited number of parameters will improve the understanding of phosphorus dynamics and will provide an indicator of the success of restoration projects.	MEDIUM
Extend the Anderson Exploration Ltd. water intake port to deeper waters (preferably to 12-m depth or deeper)	By drawing deeper water, the withdrawal will export nutrient-rich waters from the lake, as opposed to dilute waters (the current situation).	MEDIUM
Investigate the merit of periodically injecting lime ($CaCO_3$) into the lake hypolimnion.	Should reduce nutrient upwelling rates at spring and fall overturns; May immobilize P in surficial sediments.	LOW
Investigate the merit of mechanically aerating the lake hypolimnion.	Should reduce nutrient upwelling rates at spring and fall overturns; May immobilize P in surficial sediments.	LOW
Investigate the merit of reactivating the City of Fort St. John water withdrawal system by extending the intake port to a depth of 12-m or more, and discharging water to Lower Stoddart Creek or the Peace River during periods of internal loading.	A deep-water withdrawal will increase nutrient export rates and increase the velocity of Lower Stoddart Creek (water should only be drawn when the lake is thermally stratified).	LOW

Finally, we recommend that the Charlie Lake Technical Advisory Committee oversee both the development of the environmental management plan and the delivery of restoration activities within the plan.

Nordin and Pommen (1985) recommended that three survey-type projects be undertaken. They are reviewed here:

Fecal Coliform Survey

"A systematic survey of fecal coliform bacteria should be undertaken to determine densities in recreation areas, inflow streams and at domestic water intakes".

The Peace-Liard Community Health Services Society has conducted fecal coliform monitoring at the waterfronts of Charlie Lake, Beatton and Rotary Parks for many years. These summer programs are expected to continue for the foreseeable future (personal communication, Ann Thomas, Chief Public Health Inspector).

Limited inflow monitoring of bacteria was undertaken by MELP and PLCHSS prior to the installation of the Charlie Lake community sewer, an action that redirected most residential sewage away from the watershed. A limited followup program could be considered for the ditched drainages of southwest Charlie Lake, but is not considered a priority.

The assumption is made here that very few, if any, residents rely on Charlie Lake for domestic (at least potable) water supply.

Watershed Study

"A watershed study should be undertaken to establish the sources of nutrients, particularly phosphorus, which maintain the present eutrophic condition of Charlie Lake. This should include quantification of phosphorus from:
(1) ground disposal sewage,
(2) agriculture,
(3) inflow streams,
(4) ground water,
(5) dustfall and precipitation, and
(6) internal lake processes."

This level of assessment will not be possible with current resources at Ministry of Environment, Lands and Parks. Accordingly, we recommend moving directly to restoration works that have the potential to benefit water quality in Charlie Lake, while providing direct benefit to other resource interests (ie. wildfowl wetland development. As opportunities arise within the framework of the watershed management plan, the quantification of various phosphorus loading components should continue.

Insecticide/Herbicide Survey

"A survey should be undertaken to determine the type, quantity and location of insecticides and herbicides used in the Charlie Lake watershed. A sampling program could then be undertaken to monitor agricultural chemicals (insecticides and herbicides) in lake sediments, particularly in periods when these chemicals are being used, or in the heaviest runoff period (freshet)".

A limited survey was conducted in 1992, and is reported in this document. Additional pesticide monitoring is recommended and should concentrate on fine bottom sediments of the larger or agriculturally significant tributaries to Charlie Lake (East and West Stoddart Creeks, Coffee Creeks, etc) or on the best media for the specific pesticide in question. The ecological significance of any located pesticides should be determined.

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