NUTRIENT CHARACTERISTICS OF URBAN STORMWATER DETENTION PONDS ON THE CANADIAN PRAIRIES

KENNETH A. SCOTT

A thesis submitted in fulfilment of the
requirements of Napier University
for the degree of Doctor of Philosophy

This research programme was carried out
in collaboration with the Saskatchewan Research Council
Saskatoon, Saskatchewan, Canada.

February 1997

Dedication

I dedicate this thesis to my parents, Alexander and Patricia.

TABLE OF CONTENTS

page

TABLE OF CONTENTS

TABLE OF CONTENTS

page

TABLE OF CONTENTS

LIST OF TABLES

page

LIST OF TABLES

page

LIST OF FIGURES

page

LIST OF FIGURES

LIST OF FIGURES

page

GLOSSARY OF ABBREVIATIONS

ABS	-	acrylonitrile butadiene styrene
AES	-	Atmospheric Environment Service
APHA	-	American Public Health Association
AWT	-	Advanced Wastewater Treatment
BMP(s)	-	Best Management Practice(s)
BOD	-	Biological Oxygen Demand
COD	-	Chemical Oxygen Demand
DIC	-	dissolved inorganic carbon
DIN	-	dissolved inorganic nitrogen
DIP	-	dissolved inorganic phosphorus
DO	-	dissolved oxygen
EMC	-	event mean concentration
hp	-	horse power
i.d.	-	internal diameter
LOI	-	loss on ignition
NTA	-	Nitriloacetic acid
NURP	-	Nationwide Urban Runoff Program (USEPA)
OECD	-	Organization for Economic Cooperative Development
PC	-	particulate carbon
PC-SWMM	-	personal computer Stormwater Management Model
P_{int}	-	interstitial dissolved inorganic phosphorus
PP	-	particulate phosphorus
PSC	-	phosphorus sorption capacity
SRC	-	Saskatchewan Research Council
SWDP(s)	-	stormwater detention pond(s)
TKN	-	total Kjeldahl nitrogen
TN	-	total nitrogen
TP	-	total phosphorus
TSS	-	total suspended solids
U.K.	-	United Kingdom
USA	-	United States of America
USEPA	-	United States Environmental Protection Agency
UV	-	Ultra Violet
WODR	-	winter oxygen depletion rate

ACKNOWLEDGEMENTS

My thanks are extended to Dr. Hans Peterson, Principal Research Scientist of SRC's Water Quality Section, without whose support this project would not have been possible. Equally, many thanks are due to Professor Nick Christofi of Napier University for making the project happen and for sound advice, encouragement, and humour when required. For financial support, I thank the Pollution Research Unit of Napier University, and the Water Quality Section, Saskatchewan Research Council.

Among items too numerous to mention, Yvonne Wilkinson is thanked for excellent administrative and secretarial support, and especially for generously giving her time to get the final manuscript in shape. For personal support through the life of a graduate student I thank my colleague Joanne Sketchell.

The technical staff of the Water Quality Section were together responsible for creating a positive, enjoyable, and efficient workplace. Therefore, I thank Mary Moody, Norma Ruecker, and Ilene Cantin for all their support throughout. Thanks are given to Jennifer Neudorf for help with zooplankton work. I thank Barry Paquin, Dr. Guy Melville, and Brett Smith for their helpful discussions, and for lending of field equipment.

For help with field work in the City of Regina, I thank Municipal Engineering staff, including, Simon Hashem, and summer students Bill Heywood, Gord Barschall, Rob Thompson, Samantha Rizac, and Arlen Babcock. Special thanks are also given to Mark Smishek and other staff members of the Regina Parks Department.

Lastly, the efficiency and good humour of SRC's Stores staff, Wally Reilkoff, Darcy Friesen, Greg McLean, Brent Hannah, and Terry Mazurkevich never went unnoticed, and are hereby gratefully acknowledged.

DECLARATION

This thesis records details of experimental research carried out by myself at the Saskatchewan Research Council (Water Quality Section), Saskatoon, Saskatchewan, Canada.

The thesis is of my own composition and has not previously been accepted, in part or whole, for a higher degree.

Signed

STATEMENT

I declare that Kenneth A. Scott has spent 12 terms of research under my direction, has satisfied the regulations of Napier University, and is qualified to submit the accompanying thesis in application for the degree of Doctor of Philosophy.

Signed

The work detailed in this thesis was performed in the Water Quality Section, Saskatchewan Research Council, Saskatoon, Saskatchewan, Canada from December 1991 to October 1995.

ABSTRACT

The use of artificial ponds for the temporary storage of urban stormwater runoff is a commonly used environmental engineering practice in North America. By releasing runoff at a rate slower than the initial generation rate, on-line flood control is achieved. Urban runoff typically has a high eutrophication potential, so that single unit detention ponds may sustain excess algal/macrophyte growth within only a few years of construction. A research project was undertaken between 1992 and 1995 on stormwater detention ponds in the Province of Saskatchewan, Canada. Three ponds aged 15 to 17 years old are described in this thesis. The focus of the study was to describe the nutrient characteristics and associated phytoplankton cycles within these systems, and to further identify potential management options for water quality improvement.

Four to five months of permanent winter ice cover occurs in the central zone of the Province. Of the three systems reported here, complete winter anoxia is typical in two (1.7 to 1.8 m deep - mixed), while partial or complete anoxia occurs in the third pond according to the timing of snowmelt (2.7 m deep-thermally stratified in summer). Therefore, biological community structure is limited by the overwintering potential. Accumulated nitrogen, phosphorus, and silica sustained green algae and diatom blooms following ice-melt, and pH values of >9.5 often occurred by late April. During the open water season, hypereutrophic conditions were sustained and the systems typically oscillated between blue-green and green algal dominance according to flushing, external N- loading and mixing. Dissolved inorganic nitrogen:phosphorus (DIN:DIP) ratios of stormwater were typically below Redfield stoichiometry. Therefore, in-pond DIN:DIP ratios of <3 were typical, with one system frequently <1. These low ratios were the result of both internal P-loading and N loss mechanisms during dry weather storage periods, and DIP >0.25 mg/L sometimes occurred within the euphotic zone. Dense N-fixing *Anabaena* sp. blooms periodically developed under suitable climatic conditions. Nitrogen fixed into the system in turn supported non-fixing species as a subsiding bloom was mineralized. Nutrient and phytoplankton cycles fluctuated within short time scales, according to physical disturbances and algal self-shading at peak biomass (sometimes >100 mm^3/L). Average algal biomass levels in the stratifying pond were lower

than the other ponds on account of sedimentary losses to an anaerobic bottom zone.

Grazing by herbivorous zooplankton was generally not significant in promoting phytoplankton species successions. The zooplankton of all three systems were dominated by cyclopoid copepods and rotifers. Predation by fish (minnows), food quality (blue-green algae dominance), and losses during large flushing events are among factors which may suppress desirable large-bodied cladoceran zooplankton in these systems.

The mean surficial (1 cm) sediment iron content was lowest ($\approx$27 mg Fe/g dry wt *vs.* 38 mg Fe/g dry wt) and the organic content was highest ($\approx$18% LOI *vs.* 11% LOI) in the stratifying pond compared to the two non-stratifying ponds. Also, the mean sediment depth in the stratifying pond (not including the littoral slope sediments) was highest ($\approx$21 cm *vs.* 9 to 13 cm), when compared to the two non-stratifying ponds. Iron may be lost from the stratifying pond by flushing of anaerobic hypolimnetic waters during stormflows. Groundwater inflows to the stratified pond resulted in a higher alkalinity system and consequently the highest mean sediment calcium content of the three ponds ($\approx$82 mg Ca/g dry wt, *vs.* 31 to 34 mg Ca/g dry wt). The surficial sediment total-P [TP] of the three ponds ranged from 0.97 to 1.26 mg/g dry wt, and in all cases 11 to 15% of this was associated with inorganic extractable P, with more in the calcium than the iron/aluminium bound fraction.

At peak water temperatures ($\leq$26°C) internal P loading rates >30 mg/m^2/day were calculated from field data in the non-stratifying ponds. An average P release rate of 15 mg/m^2/day was measured during anaerobic incubation of sediment cores from one of these ponds at 20°C. However, aerobic incubation of these sediments showed that Fe concentrations were sufficient to provide high P uptake potential when oxidized. Anaerobic incubations of intact cores from the stratifying pond gave average release rates of 5 to 16 mg/m^2/day from 5 to 20°C. Field data also showed that net internal P loads were reflected by changes in the surficial sediment P pool.

A nutrient input budget for the stratifying pond showed that groundwater baseflow supplied a massive amount of DIN (as NO_3) relative to stormflows. If the seasonal stormwater DIP load was expressed as an averaged areal mass/day, the average seasonal internal P loading (~8 to 10 mg/m^2/day) was four times higher. Theoretical P removal efficiencies of 63 to 80% were calculated for the ponds, but resuspension and flushing of internally loaded P accumulated during dry weather may reduce these values. A TP mass export of 0.25 to 0.3 kg/ha impervious/0.58 year from 250 mm precipitation was calculated from runoff studies in Saskatoon.

In experimental work, inorganic nitrogen additions to the most N-limited pond were carried out from May to July 1994. Complete dominance of the spring to mid-summer phytoplankton by green algae and diatoms was maintained. However, warming water increased P recycling, and during a period of lower than average wind speeds a non-fixing blue-green algae bloom developed in place of the usual N-fixing algae bloom. No significant alteration to the zooplankton species composition was evident despite structural changes to the vernal phytoplankton composition.

Phosphorus inactivation with aluminium sulphate was successful in improving water quality for a six week period during which the control pond developed a dense N-fixing algae bloom. Sediment surface oxidation was promoted by the reduction of productivity, and P adsorption to sediment iron complexes was an important secondary benefit. Several very large storms were ultimately responsible for exchanging approximately 100% of the storage volume, after which bloom conditions were restored. The procedure may be an effective short-term measure, but benefits will not extend beyond major exchange events.

Management options for aesthetic improvement are very limited in these hypereutrophic ponds. External DIP loads will continue to be at least 5 to 10 times greater than threshold values for nuisance algal growth, and seasonal internal loading of P is high. The inability of increased N availability to prevent blue-green algae bloom formation, together with high exchange volumes and a general lack of herbivorous zooplankton, suggest that top-down management interventions (limited by overwintering) to control zooplanktivores are unlikely to prevent algae bloom formation in ponds with lower volume:catchment area ratios.

More work is required with regard to nutrient budgets if pond operational efficiencies are to be accurately assessed. In addition, measurement of primary productivity would provide invaluable information for any attempt to model algal growth in these ponds. Sediment removal is ultimately required as a long-term maintenance measure, but more information on the incorporation of P inactivation agents directly into the sediment structure is needed as a means to retard internal P loading.

Chapter 1
INTRODUCTION

SECTION A
GENERAL LITERATURE REVIEW

1.1 Eutrophication: A Global Problem

Freshwater ecosystems comprise a relatively small proportion of the total water on the planet. According to Overbeck (1989) freshwater accounts for 2.4% of the total global reserve. Of this freshwater reserve, polar ice and glaciers account for 87.4%, groundwater 12.3%, with only 0.3% as rivers and lakes (atmospheric reserve is minuscule).

Cultural eutrophication of both freshwater and marine environments remains a fundamental problem throughout the world. The prime causes of eutrophication are considered to be the build up of plant nutrients (in particular, nitrogen and phosphorus), organic matter, and silt, which combine to produce increasing algal and plant biomass, reduced water clarity, reduced depth, decreased diversity of the biotic components, and generally impaired resource value (Harper, 1992).

Recognition and investigation of the eutrophication problem was well underway by the 1960s (Vollenweider, 1968). In particular, the impact of inadequately treated sewage and the discharge of P containing detergents emerged as two of the most obvious causes of water quality deterioration (Edmondson, 1991; Forsberg and Ryding, 1980). In addition to identifiable point source loading, significant eutrophication potential of diffuse or non-point source loading of nutrients from both rural and urban environments has been recognized (*e.g.,* Duda, 1985; USEPA, 1983). Diverse non-point sources are much more difficult to deal with than identifiable point sources in terms of eutrophication control.

In the United States, the extent to which surface waters have deteriorated at the trophic level has recently been assessed (Duda *et al.,* 1987; USEPA, 1990*a* and 1989). Of the U.S. lakes and reservoirs assessed in 1988, 30% were classified as eutrophic or hypereutrophic with impaired usage. The status of a large number of privately owned lakes and reservoirs is not known. The concept of impaired usage is difficult to standardize since criteria used in the classification or assessment process is variable, and different usage represents different priorities to different user groups, *e.g.,* quality potable water source *vs.* productive sport fishery (USEPA, 1990*b*).

Canada has the largest lake area in the world. Most of these lakes are located within the northern Shield regions and are typically deep, cold, oligotrophic waters. However, some waters located in the southern regions are reported to be seriously impaired from domestic and agricultural runoff (Janus and Vollenweider, 1981). Numerous reports from Europe and elsewhere indicate that the problem of cultural eutrophication is globally widespread. In addition to other resource impairments, blooms of toxin producing blue-green algae in potable water supplies may present serious health concerns, as well as taste and odour problems *(e.g.,* Codd *et al.,* 1989).

1.2 Urbanization

1.2.1 Demophora

By the end of the 20th century it is predicted that half of the world population will live in urban areas (Barica, 1992; Lindh, 1987). During the period 1975 to 2000, urbanization will have been three times that of the period 1950 to 1975 (Lindh, 1987). According to Whyte (1985), in 1900 no city in the world had more than 5 million inhabitants. By 1950 there were six, by 1980 there were 26, and the United Nations predicts that by the year 2000, there will be 60 cities of over 5 million inhabitants (Lindh, 1987). The combined effect of increased production and consumption *per capita*, and the growth of population and urbanization is termed demophora. The demands that demophoric growth imposes on the planet's water resources will increasingly represent one of the most fundamental challenges for sustainable development.

1.2.2 Alteration to the hydrological cycle by urbanization

The process of urbanization leads to significant alterations in the hydrological cycle (Ripl, 1992). The primary changes associated with land development result from an increased percentage of impervious cover, so that both the runoff volume and runoff rate are increased following precipitation. As development proceeds, loadings become increasingly concentrated in surges of stormwater (Redfield, 1991). The generalized pattern of change as it relates to a catchment area water budget is shown in Figure 1.1.

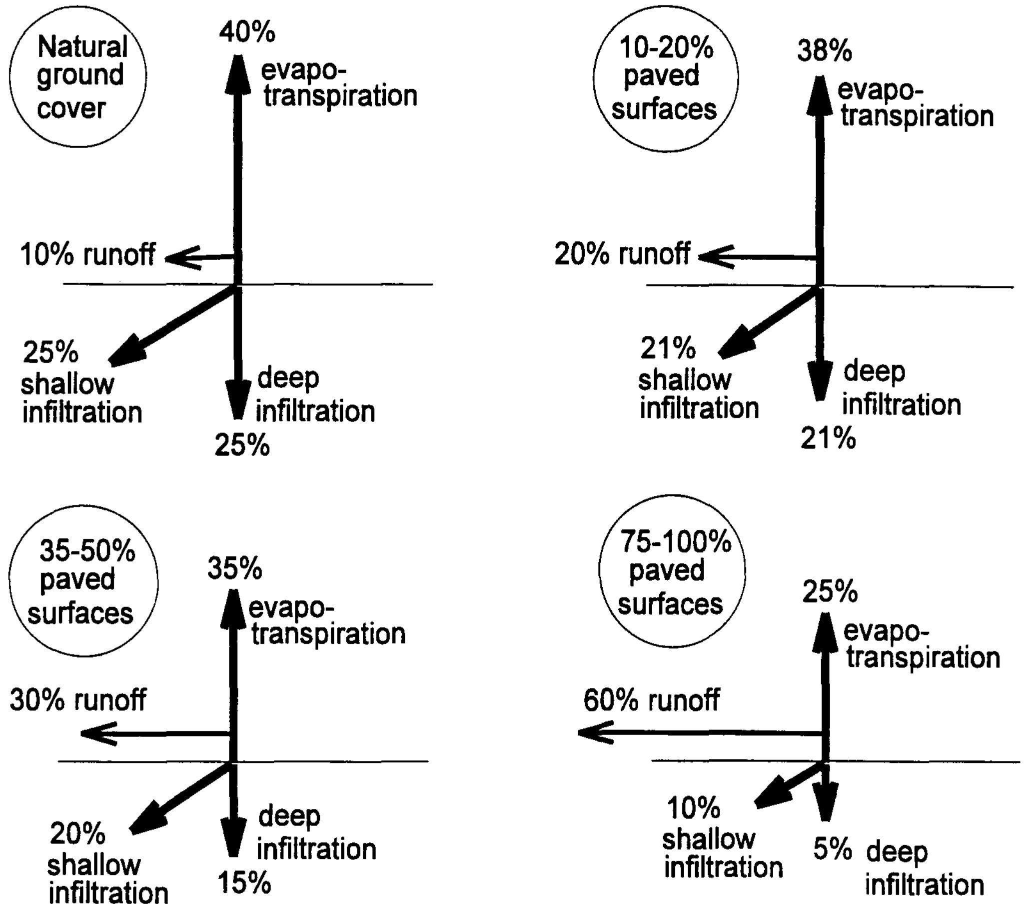

Figure 1.1 Effect of impervious ground cover on stormwater runoff flows. (adapted from Tourbier and Westmacott, n.d.)

As a consequence of these runoff changes, exports of nutrients, suspended solids, metals, and hydrocarbons, *etc.* are increased in the urban environment and carry with them the potential for negative impacts on receiving waters and their biota. Such impacts include eutrophication, dissolved oxygen depletion, silting, toxic accumulations (metals and organics), erosion, bacterial contamination, and thermal alterations (Browman *et al.,* 1979; Field and Pitt, 1990; Makepeace *et al.,* 1995; Sartor *et al.,* 1974).

1.3 Eutrophication Potential of Urban Runoff

Urban watersheds typically export 5 to 20 times as much P as a comparable undeveloped watershed (Athayde *et al.,* 1983; Reckhow *et al.,* 1980). The export of N may also be increased from developed *vs.* predeveloped watersheds; however, as development proceeds, P export tends to outpace N so that a progressive divergence in the N:P ratio towards lower values is typical of urban watersheds (Redfield and Jones, 1982). Lower N:P ratios in urban runoff have ecological implications for receiving waters and storage impoundments in relation to nutrient limitation of primary productivity. Numerous studies have demonstrated the stimulatory potential and availability of nutrients for algal growth in urban runoff (*e.g.*, Sonzogni *et al.*, 1982). Short-term reductions in phytoplankton biomass and productivity may initially follow stormwater inputs due to flocculation of algal cells with charged particles, cell washout, and short-term dominance of the water column turbidity by sediment (Jones and Redfield, 1984).

Reported concentrations of N and P in urban stormwater vary greatly according to geographical location, composite land uses, catchment soil types, season, inter-event dry period (build up), rainfall intensity and duration patterns (wash-off), paving materials (roughness), and ongoing practices within the catchment boundary (Athayde *et al.*, 1983). Total-P concentrations in urban stormwater generally range from 0.01 to 2.5 mg/L. Although peak concentrations of dissolved inorganic phosphorus [DIP] as high as 4.93 mg/L have been reported, ranges of 0.01 to 1.0 mg/L are more common (Browman *et al.*, 1979). The DIP contribution is generally in the range 35 to 70% of the TP (Wanielista and Yousef, 1992). Since the "first flush" usually produces both peak discharge volume and peak concentrations, the above concentration ranges may represent mass loads that greatly exceed threshold values for P of 0.025 mg/L (Vollenweider, 1976) above which nuisance algal growth is likely to occur in lakes. The USEPA Nationwide Urban Runoff Program [NURP] final report (Athayde *et al.*, 1983) summarized and transformed stormwater quantity and quality data collected from 81 representative sites (residential, mixed, commercial, industrial, and open/non-urban) in 22 U.S. cities. The data comprised more than 2300 discrete storm events. Data summaries and analyses were performed based on "Event Mean Concentration" (EMC = total constituent mass divided by the total runoff volume). Table 1.1 shows the median EMC values calculated for four different land use categories. Table 1.2 shows EMC medians transformed to means and an estimate for a mean EMC for median category urban sites.

Table 1.1 Median event mean concentration [EMC] for all Nationwide Urban Runoff Program [NURP] sites by land-use category. (Athayde *et al.*, 1983)

	Residential		Mixed		Commercial		Open/ Non-urban	
Constituent	**Median**	**CV**	**Median**	**CV**	**Median**	**CV**	**Median**	**CV**
TSS mg/L	101	0.96	67	1.14	69	0.35	70	2.92
BOD mg/L	10	0.41	7.8	0.52	9.3	0.31	-----	-----
COD mg/L	73	0.55	65	0.58	57	0.39	40	0.78
Total-P μg/L	383	0.69	263	0.75	201	0.67	121	1.66
DIP μg/L	143	0.46	56	0.75	80	0.71	26	2.11
TKN μg/L	1900	0.73	1288	0.50	1179	0.43	965	1.00
NO_{2+3}-N μg/L	736	0.83	558	0.67	572	0.48	543	0.91
Total-Cu μg/L	33	0.99	27	1.32	29	0.81	-----	-----
Total-Pb μg/L	144	0.75	114	1.35	104	0.68	30	1.52
Total-Zn μg/L	135	0.84	154	0.78	226	1.07	195	0.66

Table 1.2 Annual urban runoff loads based on 40 inches of precipitation per year. (Athayde *et al.*, 1983)

Constituent	**Site Mean Concentration (mg/L)**	**Residential (kg/ha/year)**	**Commercial (kg/ha/year)**
Assumed Runoff Coefficient		0.3	0.8
TSS	180	550	1460
BOD	12	36	98
COD	82	250	666
Total-P	0.42	1.3	3.4
DIP	0.15	0.5	1.2
TKN	1.90	5.8	15.4
NO_{2+3}-N	0.86	2.6	7.0

The extreme variability of values reported in the literature, even from different events at the same site suggest that for certain management purposes, site-specific studies are necessary for reliable characterization of runoff quantity and quality.

1.4 Urban Runoff Quantity and Quality Models

The use of simulation models for predicting stormwater runoff quantity and quality is now common practice. Information may be used for a variety of purposes, such as design of stormwater conveyance and detention/treatment systems. Such models can often be used to describe systems in such a way that otherwise impracticable costs of continuous data collection can be avoided. A large number of models exist, with varying degrees of complexity and simulation potential. Nix (1990) has given a comprehensive review of urban stormwater management models. For this study, the USEPA model PC-SWMM, version 4.2 (Huber and Dickinson, 1988) was used to estimate seasonal nutrient loads after calibration of the model in 1994 and 1995 for a City of Saskatoon detention pond (Raymond *et al.,* 1995).

1.5 Non-point Sources in the Urban Environment

Nitrogen and phosphorus may derive from a number of non-point sources in the urban environment. Sources include dry and wet precipitation, lawns and other soils within the catchment, fertilizers, leaf litter, organic decomposition processes, pets, and wildlife. Areas undergoing construction can be expected to export more sediment than the final "stable" developed watershed.

For atmospheric inputs, Rast and Lee (1983) estimated a value of 0.25 kg/ha/yr P for a typical watershed based on literature values. Wetzel (1975) found an atmospheric P deposition rate of 0.1 to 1.0 kg/ha/yr, with most values falling at the lower end. Wetzel (1975) noted that the P content of precipitation and particulate fallout was variable, with seasonal highs in agricultural regions in the spring.

Particle size distributions of street sweeping from Orlando, Florida, have been summarized by Wanielista and Yousef (1992). Size fractions between 0.25 and 0.5 mm diameter comprised about 50% of the total weight. Particles less than 0.125 mm accounted for about 2% of the total weight, but this fraction is reported to account for greater than 80% of the suspended solids in runoff water, although the velocity of peak flows may be sufficient to transport some larger particles. Studies have shown that most PP tends to be associated with the fines ($\leq$0.125 mm) fraction (Sartor *et al.,* 1974). Iron may also be associated with these fine particulates at concentrations higher than other metals such as Pb, Cu, and Zn (Harrison and Wilson, 1985). Stormwater Fe concentrations ranging from 0.08 to 440 mg/L have been found, with mean ranges of 1.0 to 12 mg/L. Aluminium and calcium concentrations ranging from 0.1 to 16 mg/L and 4.8 to

26.5 mg/L, respectively have been measured in urban runoff. Sources for these metals may be natural or anthropogenic (Makepeace *et al.,* 1995). These metals are of interest since they may determine the characteristics of P in lake sediments (Boström *et al.,* 1982).

Street sweeping practices have been reported to be generally ineffective at reducing the fine particulates and soluble components of stormwater. Prych and Ebbert (1987) stated that the main reason for the statistically insignificant effect of street sweeping in their study was that sweeping removed only about half of the total surface solids, and much less of the <62 μm fraction, which accounted for most of the suspended solids in runoff. Athayde *et al.,* (1983) concluded from all the NURP experimental street sweeping programs that "the hypothesis that street sweeping decreases EMCs is generally not shown by the data, though it could occur in isolated, site-specific cases". Vacuum removal may be more efficient than brush sweeping with regard to reduction of the fines fraction.

1.6 The Use of Stormwater Detention Ponds

1.6.1 Operational characteristics

Amendment of the conventional closed conduit urban drainage system with additional on-site storage impoundments has become a common practice in North America (Driscoll, 1986). Impoundments of this kind are usually referred to as Stormwater Detention Ponds [SWDPs]. Their basic function is flood control and the attenuation of peak flows through restoration of an extended hydrograph, permitting stormwater quality modification in the process. Temporary detention of flood water is achieved by routing of branch sewers to one or several inflow pipes to a pond. A single outlet is characteristic, and temporary detention is achieved when the capacity of the outflow pipe or channel is less than the summed capacities of the inflow pipe(s). On-line pond systems with a permanent storage volume (wet ponds) are the most common practice, and may provide high quality treatment if adequately designed (Urbonas, 1986; Walker, 1987). The presence of a permanent pool in wet detention ponds facilitates modification of stormwater quality during both the operational (*i.e.,* while inflow and/or outflow is still taking place) and permanent storage (closed system function during dry weather) phases, *via* the following mechanisms:

Solids: settling - direct sedimentation; enhanced sedimentation by precipitation of agglomerated particulates *via* charge interaction (*e.g.,* clays) during and shortly after operational phase.

Dissolved components: settling *via* adsorption on charged particulates in the water column; precipitation *via* interaction with soluble⇔insoluble metal phases (*e.g.,* Fe^{3+}, $CaCO_3$); direct adsorption on and within the sediment surface structure; biological uptake/transformation.

Other engineering practices for stormwater control include detention in underground vaults, diversion of stormwater to basins with no permanent storage volume ("dry" ponds operating both on-line and off-line with or without infiltration), wetland diversions, or various in-series combinations of the above (Driscoll, 1986). Construction of SWDPs may allow drainage from new developments to be routed through existing trunk sewers and avoid the need for trunkline enlargement/expansion. Therefore, their function can be considered to be of physical (quantity), chemical (quality), and financial benefit. A typical schematic for a single-unit wet detention pond is shown in Figure 1.2.

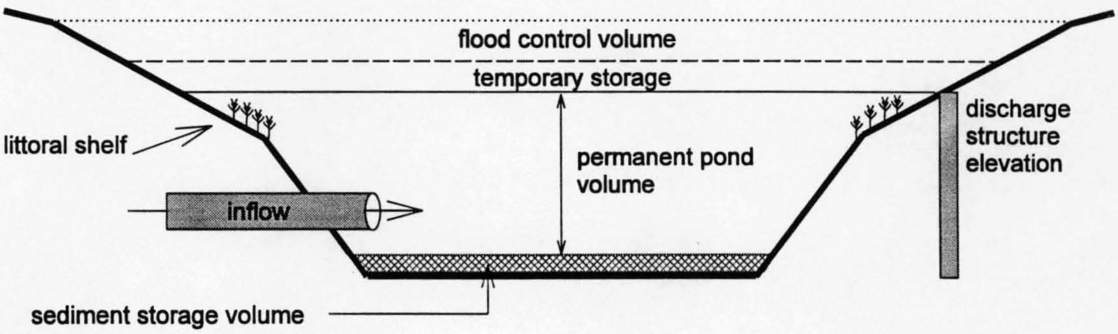

Figure 1.2 Schematic representation of a typical wet stormwater detention pond.

1.6.2 Sizing and basic treatment concepts in stormwater detention ponds

From a stormwater quantity perspective, SWDPs are designed to accommodate the theoretical runoff volume generated by the maximum precipitation volume of a specified return period (usually 25, 50, or 100 year flood falling in a 24 hour period), in excess of the permanent storage volume (*i.e.,* above the invert elevation). This is based upon the catchment area, runoff coefficient, and the time of runoff concentration within the system with provision for restoration of adequate operating capacity of the pond in a specified time period, *e.g.,* 72 to 120 hours. The frequency distribution (frequency/ intensity/ duration curves) of a regional rainfall record is used to calculate the exceedence probability of any specified event by one of several statistical methods (Wanielista and Yousef, 1992). Similar calculations can be used to obtain average inter-event dry periods. These frequencies can be used to approximate average treatment efficiencies for a given pond design. For example, if optimum treatment by sedimentation requires 72 to 120 hours of detention, the frequency of inter-event dry periods, which are less than this interval can be used to approximate the percentage of storms that will result in less than optimal detention time.

The pond depth to pond surface area ratio is an important parameter for several reasons:

(a) surface area/pond volume will influence the extent to which in-pond sediment retention/release of nutrients/metals, *etc.* affects the water column, and the overall treatment efficiency of the pond.

(b) the proximity of the photosynthetic zone to the sediment may determine the extent to which phytoplankton can access recycled nutrients.

(c) the greater the distance of inlets from outlet structures, the less likely that "short circuiting" may occur during stormflows. Inflow velocity dissipation is maximized, favouring high particulate removal *via* sedimentation and minimizing the counter-effects of localized sediment resuspension.

The engineering design will depend on topography, size of service area, dedicated land, and the rate capacity of the receiving water to receive discharge volume where appropriate. For aesthetic and maintenance requirements, ponds in the Cities of Saskatoon and Regina have been designed around a minimum surface area of about 2 ha. Pond permanent depth designs may be calculated based on specified detention requirements for average precipitation events (treatment optimization) and/or depend on landscape relief, hydraulic gradients, water table height, and excavation costs. Permanent pond depths in the range 1 to 3 m are typical, and are recognized as depths that help promote an aerobic water column profile. From a treatment perspective, there have been significant technical advances in detention pond designs in the last decade (*e.g.,* Reid

Crowther, 1995; Schueler *et al.*, 1992). This is particularly the case in Canada where SWDPs were previously constructed strictly from a stormwater quantity perspective. In terms of design and optimization of the treatment function of SWDPs, the U.S. has been ahead of Canada, with the regulatory emphasis coming from the U.S. *Water Quality Act* (1987), which includes provisions for stormwater permitting (Section 402(p)) and non-point source control (Section 319).

1.7 Empirical and Theoretical Models for Constituent Loading, Retention, and Eutrophication

1.7.1 Phosphorus modelling concepts

Phosphorus is usually regarded as the primary limiting nutrient in the productivity of freshwater ecosystems (Schindler, 1977). Empirical models that predict P retention/steady-state concentration in lakes and reservoirs have been developed based on P loading, lake morphometry, and hydraulic detention time (Jones and Bachman, 1976; Vollenweider, 1968, 1975, and 1976). The basic Vollenweider model (with additional term for internal loading) describes the basic mass balance of total-P in a lake as:

Equation 1.1

$$\frac{dP_t}{dt} = \frac{L_{ex}}{\bar{Z}} - \rho P_t - \sigma P_t + \frac{L_i}{\bar{Z}}$$

where:

L_{ex} = the external TP loading ($mg/m^2/yr$)

Z = mean depth of the lake

P_t = total-P concentration in the lake

σ, ρ = rate coefficients (1/yr) for P sedimentation and flushing, respectively

L_i = the internal loading ($mg/m^2/yr$) - optional

In equilibrated or steady-state systems (*i.e.,* $dP_t/dt = 0$), these mass balance models have been successful in predicting the generalized trophic status of water bodies with a range of P loadings and flushing rates (more particularly large and great lakes). Prediction of planktonic algal biomass is based on the regression relationships between P and Chl *a*, a predictive relationship which holds for P limited systems (N:P >12 by weight) in which other cellular growth requirements are assumed satisfied (Ahlgren *et al.,* 1988; Dillon and Rigler, 1974*a*). Development of the basic P loading equation assumed that systems are completely mixed, P influx/efflux is constant, and that sedimentation rate is directly proportional to the lake water concentration. Successors to the basic TP model have tended to focus on estimations of the sedimentation coefficient (σ), or comparable P loss rates to the sediments. Approaches to this problem by various researchers for particular data sets were summarized by Canfield and Bachmann (1981), and included a value of 10/mean lake depth, where 10 represents m/yr (Vollenweider, 1975), a fixed sedimentation coefficient of 0.65 yr^{-1} (Jones and Bachmann, 1976), development of a lake P retention coefficient (R) based on measured annual input/output values for P (Dillon and Rigler, 1974*b*), a settling velocity parameter (Chapra, 1975), and a P retention coefficient of 1/√hydraulic detention time (Larsen and Mercier, 1976; Vollenweider, 1976). Other refinements to the basic Vollenweider models have included the separation of cellular (as actual trophic status indicator) *vs.* dissolved P components for rapidly flushed lakes in which basic Michaelis-Menten terms for P removal by algae are introduced (Uttormark and Hutchins, 1985). As shown in Equation 1.1, inclusion of terms to account for internal loading of P may also be required (Nurnberg, 1984).

For systems such as SWDPs operating as batch loaded reactors, Equation 1.1 cannot be used to predict an average annual or seasonal water column TP and algal biomass value in a meaningful way. From a quantity and quality standpoint, both stormflow batch loading and flushing are highly variable. Therefore, transient application of Equation 1.1 would be required at discreet time intervals for SWDPs. Compartmentalization of P in stratified systems may complicate prediction of epilimnetic P and realized algal biomass. Also, since N and/or light rather than P may limit primary production in some eutrophic systems, TP alone cannot be used to predict algal biomass due to the variable DIP component. In N-limited productive systems, chlorophyll *a* prediction is more uncertain and may necessitate the inclusion of terms for N in regression relationships (*e.g.,* Prairie *et al.,* 1989; Smith, 1982).

For conservative and non-conservative substances in SWDPs, Wanielista and Yousef (1992) gave the following for mass balance calculation:

Equation 1.2

$$\frac{d(VC)}{dt} = QC_i - QC_o + V(r_g) - V(r_u)$$

where:

V= pond volume, m^3

C = concentration of component, mg/m^3

QC_i = incoming load of component, mass t^{-1}

QC_o = outlet load of component, mass t^{-1}

r_g = generation rate, mass m^{-3} t^{-1}

r_u = utilization rate, mass m^{-3} t^{-1}

Equation 1.2 is basically the same as the P mass balance in Equation 1.1, where r_g accounts for internal loading/regeneration with r_u representing the sum of the various sink mechanisms. The term r_u, or utilization rate, as a factor in changing the water column concentration is not valid if the component is a dissolved substance moving into planktonic biomass and being measured as a total (*i.e.,* dissolved plus particulate) as opposed to sedimentation or other loss mechanisms (*e.g.,* denitrification). Upon restoration of a closed system during dry weather, $V_{op} \rightarrow V_{storage}$, and QC_i and $QC_o = 0$. Therefore, expansion of r_u into various pathways introduces basic mass balance concepts for both P and N.

1.7.2 Physical stormwater detention pond design and operational performance models

The permanent storage volume/mean runoff event volume ratio is an important parameter for detention pond treatment efficiencies. As in natural systems, the ratio determines the average hydraulic detention time of stormwater and its constituents, with a positive relationship between ratio increase and detention time. Using data from the NURP study (Athayde *et al.,* 1983), Driscoll (1983) has shown that removal efficiencies for suspended solids and total-P in SWDPs are related to the following hydraulic variables:

Q_m/A = mean surface overflow rate during storm periods (cm/hr)
= pond outflow/surface area

V_p/V_m = permanent pool volume/mean storm volume (dimensionless)

The first ratio determines the potential for removal of particulates of given settling velocity during the operational phase. Particles having settling velocities greater than Q_m would be removed (assumes complete mixing), while the remainder would either be discharged or remain suspended in the pond when outflow ceases. The second ratio determines the potential of the pond to retain and, subsequently, remove remaining particulates and dissolved components during storage periods between rain events (Walker, 1987). The assumption by Driscoll (1983) that TP removal can be directly accounted for by suspended solids removal is described by Walker (1987) as deficient in that it ignores the variable dissolved P component of the TP in the inflow. However, Walker concedes that, for basic operational purposes, it is extremely difficult to model all processes implicitly. Typical frequency distributions of particle settling velocities in urban runoff are shown in Table 1.3.

Table 1.3 Settling velocity distribution for particulates in typical urban runoff. (Driscoll, 1983)

Percentile	10	30	50	70	90
Velocity (cm/hr)	0.9	9	46	210	2000

A number of hydraulic variables relevant to pond design and long-term average treatment performance were given by Walker (1987) in relation to the application of a P retention model to urban SWDPs. The model was developed from 60 U.S. Army Corps of Engineers reservoirs and tested against other independent lake and reservoir data (OECD Reservoirs and Shallow Lakes - Clasen and Bernhardt, 1980). The term "relative volume" (V_{rel} = pond volume/ impervious watershed area) is given as a factor which normalizes pond size against contributing watershed. The pond performance indicators Q_m/A and V_p/V_m can be obtained from V_{rel} and the regional precipitation characteristics, while hydraulic detention time (T, years) is the pond storage volume/mean seasonal outflow.

In relation to the above parameters, the empirical P retention model developed by Walker (1987) assumes a second order P decay rate during operational and quiescent periods, related to the surface overflow rate, the external TP load concentration, and the ratio of inflow dissolved P to TP as follows:

Equation 1.3 Second Order Decay Rate (m^3/mg-yr)

$$K_2 = 0.56\ Q_s F_o^{-1}/(Q_s + 13.3)$$

Equation 1.4 Dimensionless Reaction Rate

$$N_r = K_2 P_i T$$

Equation 1.5 Retention Coefficient (mixed system)

$$R_p = 1 + [1 - (1 + 4N_r)^{0.5}]/(2N_r)$$

where:

F_o = inflow ortho-P/total-P ratio

T = mean hydraulic residence time

Q_s = mean surface overflow rate (m/yr)
= mean outflow rate/mean surface area

P_i = inflow total-P concentration (mg/m^3)
= total-P loading/mean outflow rate

Model errors (mean squared errors in the logarithms of predicted outflow P concentrations) were of similar magnitude for the three data sets: the model development data set (Corps of Engineers Reservoirs - 0.017); the model testing data set (OECD shallow lakes and reservoirs - 0.034); and the urban detention pond data set (0.012). Walker (1987) concluded that with appropriate regional data the model is useful for predicting average P removal efficiencies in SWDPs without detailed simulation of individual storm events.

1.8 The Role of Sediments and Internal Phosphorus Loading

The capacity of sediments as sinks and sources for P have been extensively described (*e.g.,* Boström *et al.,* 1988, 1982; Forsberg, 1989). Net release of stored P to the overlying water column may be highly significant, particularly in warm, shallow, and well mixed water bodies since the ratio of sediment surface area/water column volume is high and nutrient recycling takes place within or in close proximity to the photosynthetic zone (Marsden 1989). This mechanism is a significant feature of the SWDPs described in this thesis, that function as self-contained systems during dry weather.

1.8.1 Forms of phosphorus

Phosphorus is present in aquatic systems almost exclusively as PO_4^{3-}, occurring in the water and sediments in particulate (>0.45 µm) and dissolved forms. Environmentally significant forms of P include adsorbed and exchangeable P (*e.g.,* as present on clay and silt particles), organic-P, precipitates of metals such as Fe-oxy-hydroxy complexes, crystalline minerals, and amorphous P. Dissolved P is considered to be orthophosphate, inorganic poly-phosphates, and organic P in solution (Boström *et al.,* 1988).

Different P retention characteristics of different lakes have been accounted for in part by differences in the fractional composition of sedimentary P. Broad fraction associations have been obtained by several sequential extraction techniques (Hieltjes and Lijklema, 1980; Williams *et al.,* 1976). Extraction by the method of Hieltjes and Lijklema (1980) yields loosely bound P (NH_4Cl extractable at neutral pH), a high level that is indicative of P binding site saturation; NaOH extractable P (thought to correspond to Fe and Al bound P), a high level that is indicative of a high retention capacity; and HCl extractable P, corresponding to Ca bound P (Boström *et al.,* 1982).

A number of investigations into the bioavailability of various sediment fractions have been carried out. Generally, the combined NH_4Cl and NaOH P fractions have been reported to correspond approximately to the amount accessed by algae in bioassays where sediment was the sole source of P (Hegemann *et al.,* 1983). Williams *et al.,* (1980) also found NTA extractable P (roughly equivalent to NH_4Cl and NaOH P) to correspond with algal bioassay results. However, Klapwijk and Bruning (1984) state that the extent to which sediment P is available depends on the pH of the bioassays, which may be due to partial extraction of the NaOH fraction at high pH.

1.8.2 Phosphorus retention/release mechanisms

Both the organic and inorganic forms of P in aquatic systems are involved in various transformation equilibrium reactions, so that soluble P may be released from or taken up by the solid phase. The balance of these exchange transformations across the sediments will determine whether they act as net sinks or sources of P. Under normal conditions, sediments generally act as net sinks for P, with adsorption and sedimentation of P exceeding the amount released (Boström *et al.*, 1988). However, under certain conditions, export of soluble P from sediments may be considerably greater than P import and may be sufficient to cause, or at least perpetuate, eutrophic conditions (Marsden 1989).

Boström *et al.* (1988) distinguish six transfer mechanisms for the deposition of P in lake sediments: (1) sedimentation of detrital P derived from the watershed (mostly rapidly settling); (2) adsorption or precipitation with clays, amorphous hydroxides, and Fe, Mn, Al, and Ca metals; (3) sedimentation of P with allochthonous organic material; (4) sedimentation of P with autochthonous organic material; (5) direct uptake by biota and subsequent sedimentation; and, (6) direct adsorption of dissolved P onto sediment particles (difficult to distinguish from number (2) above). The quantitative contributions of these mechanisms will vary according to such factors as watershed characteristics (physical, chemical, and biological), lake morphometry and hydrology, and the size and density of particulate P carriers.

A number of complex interactions govern the transformation of P within the water and sediments. Net release of P from sediments may be derived from the inorganic and organic pool, depending on redox potential (Eh) of the water and sediment, pH, temperature, and related microbial activity. P and N release may occur by molecular diffusion along a negative concentration gradient. The greater the concentration difference, the higher the release rate. Internal loading rates may be estimated from mass balance budgets, direct measurement of release rates, or from flux calculations based on dissolved concentration gradients at the sediment-water interface. Advanced conceptual models incorporating terms for sedimentation, mineralization, adsorption/ desorption, and diffusion to predict nutrient release have been formulated (*e.g.,* Boers and van Hese, 1988; van Eck and Smits, 1986). Release rates may be influenced in excess of diffusion by gas ebullition and the burrowing/ respiratory actions of benthic macro-organisms, in which case appropriate estimations of these influences must also be made. Diffusion coefficients for phosphate and ammonium in anaerobic sediment-water systems have been calculated to be approximately 0.31 cm^2 day^{-1} and 0.85 cm^2 day^{-1}, respectively (Krom and Berner, 1980). A simplified scheme relating the interactive factors, as described above, and P release mechanisms is given in Figure 1.3.

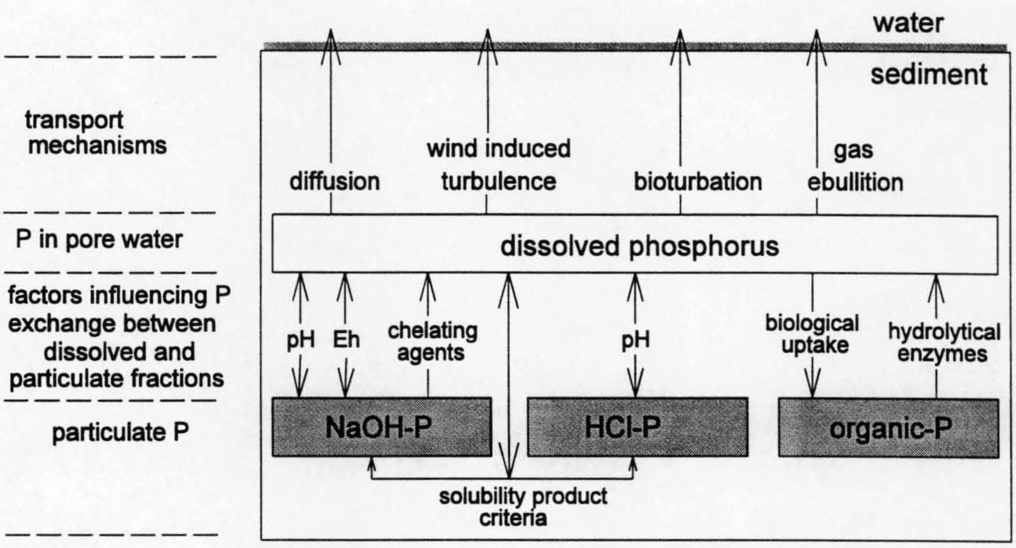

Figure 1.3 Dominant processes regulating the release of phosphorus from lake sediments. (adapted from Håkanson and Jansson, 1983)

Redox potential (Eh): The influence of redox potential on Fe speciation has been regarded as one of the dominant mechanisms in the storage and release of P from sediments *via* the precipitation/solubilization of Fe_3^+ oxy-hydroxy complexes to which phosphate may be adsorbed. These complexes are relatively insoluble under oxidizing conditions ($\geq$200 mV) and will tend to precipitate phosphate (Boström *et al.,* 1982). Frodge *et al.* (1991) reported 1 mg L^{-1} dissolved oxygen [DO] above the sediment surface maintained an oxidized sediment layer, while development of reducing conditions at DO <0.1 mg L^{-1} (Boström *et al.*, 1982) to 0.4 mg L^{-1} (Frevert, 1979) will cause the oxidized sediment surface layer to disappear, according to the temperature and organic matter dependent sediment oxygen demand. As Fe^{3+} is reduced to Fe^{2+}, phosphorus is solublized. The reduction of Fe^{3+} may result from direct chemical mediation or *via* its use as an alternative electron acceptor by iron-reducing bacteria (Jansson, 1986). Dissolved P in the interstitial pore water is then directly exchangeable with the water column and may transfer according to the mechanisms shown in Figure 1.3. However, P can also be released from sediments under aerobic conditions, particularly at elevated pH (Löfgren, 1987).

The use of NO_3^- as an alternative electron acceptor to buffer sediment-surface Eh and maintain the P binding capacity has been suggested and used by several workers. Ripl (1976, 1985) achieved some success with liquid $CaNO_3$ applications to sediments, and the work of Anderson (1982) on Danish lakes reported that little or no release of P took place if the concentration of NO_3^-N in anoxic hypolimnetic waters remained above 0.5 to 1.0 mg/L. Jensen and Andersen (1992) assessed the influence of NO_3^- additions on P release in laboratory experiments and, although considerable variation was evident, NO_3^- apparently increased the thickness of the oxidized layer at the sediment surface.

A different hypothesis on the influence of NO_3^- on P release was discussed by Böstrom *et al.* (1988) who considered that increased availability of NO_3^- has the potential to promote P recycling by facilitating mineralization of organics *via* denitrification. Also, a stimulated population of NO_3^- reducing bacteria may lead to increased Fe reduction when NO_3^- concentrations become low, if these bacteria also utilize Fe as an alternative electron acceptor (Jansson 1986).

pH and alkalinity: anaerobic release of phosphorus described above is relatively independent of pH under neutral to slightly alkaline conditions. However, an increase in pH of water in contact with the sediment will tend to decrease the P binding capacity of Fe and Al compounds, primarily due to ligand exchange reactions in which OH^- ions replace PO_4^{3-}. Minimum solubility of Fe-P complexes occurs around pH 6 (Boström *et al.*, 1982). Using intact sediment cores Andersen (1975) showed an increase in P release up to pH 9.5, followed by a decrease above this value. The mechanisms proposed were: (1) ligand exchange of PO_4^{3-} with OH^-; and (2) precipitation of PO_4^{3-} as hydroxyapatite, respectively.

Boers (1991) has suggested that the pH adjustment method used by many workers involving NaOH additions, induces misleadingly high release rates by adding alkalinity to the system. Boers (1991) found P release rates to be about five times less if pH was adjusted by CO_2 stripping. This method reduces sediment alkalinity due to efflux to the overlying water, thereby lowering sediment pH and reducing ligand exchange.

In calcareous systems, the effects may be slightly different. $CaCO_3$ precipitation is favoured at high pH and temperature, and P may be coprecipitated or adsorbed to the precipitate (within the interval pH 8.0 to 10.0). Formation of hydroxyapatite on $CaCO_3$ crystals is also favoured at higher pH levels, which may be a major P regulatory mechanism in calcareous, eutrophic systems (Boström *et al.*, 1988; Kelts and Hsü, 1978; Lopez and Morgui, 1992).

The above pH related mechanisms are particularly relevant under aerobic conditions. In shallow and predominantly mixed aerobic systems, where the euphotic zone may extend to the sediments, rapidly formed and destroyed micro-stratifications may occur. Photosynthetic pH elevation and high temperatures, along with fluctuating anaerobic micro-zones, may favour different P mobilization processes (Löfgren, 1987). The maintenance of a positive diffusion gradient due to planktonic assimilation of P in the water column may lead to net P efflux for extended periods of time (weeks or months). Such release rates may be of the same order as those occurring under anaerobic conditions, and rates of up to 50 mg P m^{-2} d^{-1} have been reported (Boström *et al.,* 1982; Nurnberg, 1988).

Temperature: release of soluble P and N increases positively with temperature mostly *via* indirect effects on biological metabolism - increasing mineralization of organic P and N (increasing O_2 consumption), increased bioturbation, or by stimulated release of stored P by bacteria. Temperature is the primary controlling factor in nutrient recycling in lakes (Forsberg, 1989).

1.8.3 Phosphorus retention capacities

In general, the potential P retention capacities of sediments will be highest in those systems rich in Fe/Al and humic material, and lowest in those dominated by sandy sediments. Concentrations are reported to vary between 0.1 mg total-P g^{-1} dm in sandy, coastal sediment, up to 10 mg g^{-1} dm in iron and carbonate rich sediment. Since an inverse relation between organic and dry matter content exists, the variation on a volumetric basis is much smaller - between 0.02 and 0.1 mg P cm^{-3} (Holtan *et al.,* 1988). Compared to anaerobic conditions, P retention capacities will usually be significantly higher under aerobic conditions. Danen-Louwerse *et al.* (1993) examined the aerobic phosphate adsorption properties of sediments in relation to their Fe and Al (hydr)oxide content (oxalate extractable) by fitting langmuir isotherms to their data. Their results produced an aerobic PSC (P Sorption Capacity) of 0.08 mol P (mol $(Al+Fe))^{-1}$ at pH 8, based on the assumption that both Fe and Al have equal affinities for P.

Adsorption/desorption equilibria for anaerobic and aerobic sediments has been the focus of much research in modelling of sediment dynamics (*e.g.*, Furumai and Ohgaki, 1989). Inducing transition from an anaerobic to an aerobic sediment surface state has significant potential as a management tool to reduce internal loading of P. Reduction of BOD by limitation of primary productivity and forced oxidation of sediment by aeration or nitrate injection are common techniques.

A review of the physical aspects of sedimentation rates of various particulate P carriers and their significance in retention characteristics has been given by Håkanson and Jansson (1983) and Sonzogni *et al.* (1982).

SECTION B
REMEDIATION SUMMARY AND STUDY FOCUS

1.9 Managing Eutrophication and its Symptoms

Although N may frequently limit algal growth with respect to other growth requirements, cyanobacteria (blue-green algae) capable of supplementing the N deficit *via* fixation of atmospheric N_2 may be encouraged, so that efforts to reverse or alleviate the symptoms of eutrophication target P as the conservative element to be reduced (Cooke *et al.*, 1993*b*). Efforts to control eutrophication range from reductions in external nutrient loading to reductions of P and internal recycling within water bodies (Marsden, 1989; Uhlmann, 1982).

External reductions in nutrient loads may comprise a number of case specific interventions across a wide scale of magnitude and complexity of application. Point source reductions can be specified through legal regulation and initiation of "Advanced Wastewater Treatment" [AWT] technologies. Integrated catchment "Best Management Practices" [BMPs] and designated zoning may address the role of land use in areas such as agriculture, forestry, and urban development in non-point source reductions. Additional measures may include source diversion (*e.g.*, Moss *et al.*, 1986), source dilution, and source interception with nutrient reduction prior to entering a lake or reservoir (*e.g.*, Hayes *et al.*, 1984).

Since significant reduction of bulk external loading from stable urban catchments to SWDPs is generally not feasible (street sweeping has been previously discussed), water quality improvements must rely on in-pond interventions and periodic maintenance.

The following sections delineate and review project specific research areas and the intervention techniques pursued to improve water quality. In-lake management techniques for phytoplankton and macrophyte control have been described by Cooke and Kennedy (1989) and Cooke *et al.* (1993*b*). Table 1.4 is a summary of techniques that may be applicable to phytoplankton and macropyhte control in existing Saskatchewan SWDPs. This does not include external loading and physical measures which would reduce pond P retention (*e.g.*, first flush diversion) or littoral slope safety features.

Table 1.4 Potentially applicable techniques for control in-lake control of nutrients and primary productivity.

Control Variable	Direction of Change	Control Measure	Control Measure Description
- Internal P load	⇓	- sediment removal - sediment sealing - sediment treatment - precipitation of water column P - hypolimnetic aeration - circulation through hydroponic gravel beds	 - physical (*e.g.*, clay) - biological (*e.g.*, *Chara*) - oxidizing agent and/or P binding agent (Fe, Al, Ca) - Fe, Al, Ca salts - (for stratifying ponds) - with or without DIN supplement for enhanced biological removal
- mixed layer depth	⇑	- destratification	- (for stratifying ponds)
- phytoplankton crop (direct)	⇓	- precipitation - mortality increase - increase grazing pressure	- Fe, Al, Ca, clay suspensions - chemical toxicity/inhibition (*e.g.*, barley straw liquor) - promote edible phytoplankton (N:P ratio increase, pH reduction)
- zooplankton crop	⇑	- control zooplanktivorous fish - reduce or eliminate anaerobic zone volume	- Pike introduction (*Esox lucius*) - aeration/destratification
- macrophyte growth	⇓	- littoral slope covers - herbicides - introduce grazers/ destructors - mechanical harvesting	- prevent rooting - *e.g.*, Crayfish, herbivorous fish

1.9.1 Phosphorus inactivation

Where internal loading of P is shown to be a significant factor producing excess P for plant growth, measures to improve water quality are required to address this source. In most cases, targeting a reduction in the internal load would only be considered worthwhile when the external load has been reduced (Cooke *et al.,* 1993*a*). The inability to significantly reduce the external load to SWDPs as a retrofit measure has been previously discussed. The rationale behind attempts to reduce internal loading in SWDPs is based on the fact that there is no external load during dry weather and that moderate rainfall may result in only minor loads and exchange volumes. Sediment removal is a costly exercise, and since SWDPs are relatively small impoundments, cost-effective treatment is possible using P precipitating agents, conceding that water quality improvements will be temporary.

Salts of aluminium, iron, and calcium are frequently used in lake management/restoration efforts to reduce both particulate and dissolved P in the water column (Cooke *et al.,* 1993*b*). The long-term benefit may be derived from a continued "sealing" or interceptory effect at the sediment surface from the settled floc and, as such, the principles of sediment metal-P interactions described in Section 1.8 apply. The method may be very effective in deep, stratifying lakes, but is less reliable in shallow systems. In some cases direct sediment injection of compounds is carried out. Ripl (1976, 1985) and Quaak *et al.* (1993) have developed sequential techniques in which sediment iron content is increased with FeCl simultaneous with pH/redox stabilization with $Ca(NO_3)_2$ injection. The principle of using calcium salts ($Ca(OH)_2$ and/or $CaCO_3$) alone rests on the precipitation of P with calcite at high pH (>9.5) and the formation of hydroxapatite (Murphy *et al.,* 1988). Babin *et al.* (1992) applied calcium salts to City of Edmonton SWDPs (Alberta, Canada) and concluded that slaked lime was more effective than calcite for P reduction.

Where sufficient alkalinity is present, the most common approach is the use of either Fe or Al salts to precipitate P. In these, cases pH reduction is induced into the range in which insoluble polymerized metal-hydroxide complexes are formed (maximal for both in the range pH 5.5 to 7.0). Fe salts may be used where an aerobic environment is expected to persist (*e.g.,* Hayes *et al.,* 1984). Under anoxic conditions, solubilization to the ferrous state may release P, while Fe may be effectively lost from the P control cycle due to precipitation with sulphide as FeS (De Vitre *et al.*, 1994), so that aeration may also be required to maintain aerobic conditions.

Since Al is not redox sensitive, it is more often the chemical of choice, either as aluminium sulphate, or in the more buffered sodium aluminate form. The hydroysis reaction for aluminium sulphate can be represented as follows:

$$Al_2(SO_4)_3 \cdot 14H_2O + 6HCO_3 \rightarrow 2Al(OH)_3 + 6CO_2 + 14H_2O$$

Lijklema (1980) has described the interactive mechanism and pH dependence of both Al and Fe hydroxide matrices with phosphate. The process of dose determination for aluminium salt applications requires that a minimum of pH 6.0 is maintained with at least 25 mg/L of alkalinity remaining (Cooke and Kennedy, 1989). Below pH 6.0, toxic soluble Al species are formed. Assuming the above safety limits are maintained, toxicity effects to biota such as fish and zooplankton are reported to be minimal, although physical changes may induce changes in the diversity of the biota, particularly benthic dwellers. A review of Al toxicity and chemistry has been given by Driscoll and Schecher (1990).

Following a series of laboratory and field trials with coagulants, application of aluminium sulphate was made to a hypereutrophic SWDP in 1995 to assess its efficacy and potential as a management technique for these systems.

1.9.2 Nitrogen:phosphorus manipulation as a means to influence phytoplankton composition

Rationale: one of the most commonly encountered symptoms of eutrophication is an increasing tendency for a seasonal shift in the algal species composition to blue-green algae (Cooke *et al.*, 1993*b*). Blue-green algal physiology and ecology continues to be an area of extensive research. There are multiple and complex interactive factors which may predispose blue-green algal dominance over other groups. Such factors include competitive mechanisms for light climate adaption (accessory pigments and buoyancy control), low half-saturation constants for light and nitrogen, nitrogen fixation, inorganic C uptake, pH optima, temperature optima, resistance to grazing, inhibitory toxin production, and the potential for facultative heterotrophy (*e.g.*, Carr and Whitton, 1982; Humphries and Lyne, 1988; Reynolds, 1987; Shapiro, 1990; Smith, 1986; Tilman *et al.,* 1986; Tilzer, 1987; Vincent, 1989; Zevenboom and Mur, 1980). Primary food web alterations towards blue-greens have multiple consequences, of which toxin production in water supplies for human and animal consumption is a very significant problem (Codd *et al.,* 1989).

Efforts to improve water quality should capitalize on the potential benefits of top-down and bottom-up control on phytoplankton abundance. Although some blue-green species may be grazed by zooplankton, for the most part, blue-green species are considered to be of poor nutritional value, and may present mechanical difficulties to consumption (*e.g.,* De Bernardi and Giussani, 1990; Fulton, 1988*a,b*; Haney, 1987). Therefore, more effective grazing may be encouraged if an improved food source quality, such as small green algal species, can be encouraged in otherwise blue-green dominated systems.

An additional consequence of encouraging non-blue-green species, is that if DIP can be moved into biomass with growth rates higher than those of buoyancy controlled blue-greens, P sedimentation rates may be increased, so that phosphorus may become less available according to the nutrient recycling rate. Also, more of the C, N, and P in blue-green biomass may be labile compared to the proportion present in green algal cells (Gunnison and Alexander, 1975), so that shifts to green species may have additional benefits.

Nutrient limitation and the effect of the nitrogen:phosphorus ratio on blue-green dominance: as previously discussed, systems tend to progressively accumulate P as eutrophication advances. In addition, N loss mechanisms may increase through coupled nitrification/denitrification and volatilization (Bouldin *et al.,* 1974; Kaspar 1985; Murphy and Brownlee 1981). In SWDPs, this scenario may be further compounded since lower N:P ratios are often associated with urban runoff loads, as compared with undeveloped watersheds (Redfield and Jones, 1982). Therefore, algal production in such systems may be N-limited and may favour blue-greens, which can either fix atmospheric N, or effectively outcompete other species for inorganic N at low concentrations (*e.g.,* Claesson and Ryding, 1977).

A "harmonic" stoichiometric composition of phytoplankton can be roughly approximated by an average atomic ratio of C106:N16:P1 (Redfield, 1958), although taxon variation and physiological state (population age, *etc.*) may cause significant deviations (*e.g.,* Healey, 1973; Vollenweider, 1985). Luxury uptake of

P in excess of immediate cellular requirements may also feature in C:N:P ratios when P is in non-limiting supply. Consideration of dissolved N:P may give a general indication of which nutrient(s) are likely to be limiting when compared to the stoichiometric ratio.

Uhlmann (1982) expressed deviation of inorganic N:P ratio from the stoichiometry (for N excess or N limitation) as a dimensionless concentration based on a fraction of 16 (atomic ratio). P limitation was given as being at a fractional value of 1.6 or greater (atomic N:P ratios >26), while N limitation is indicated by a fractional value of 0.7 or less (atomic N:P ratio <11). Forsberg *et al.* (1978) generalized trophic state transitions in relation to N:P ratios, and proposed that TN:TP ratios <10 (by weight) indicate N limitation and ratios >17 indicate P limitation.

The N:P hypothesis states that low N:P ratios favour dominance of the phytoplankton by N-fixing and non-fixing blue-greens (Shapiro, 1990). Based on summarized data from 17 lakes, Smith (1983) concluded that at TN:TP ratios >29, blue-greens should represent only a small proportion of the phytoplankton. If low N:P ratios do indeed encourage blue-greens, manipulation of N:P ratios by N addition and/or reduction of P in N limited systems may provide an opportunity to reduce blue-green incidence.

Relatively few studies involving whole lake inorganic N additions to manipulate algal blooms in hypereutrophic systems have been reported. Barica, *et al.* (1980) induced a shift from *Aphanizomenon flos-aquae* to small greens and cryptomonads with nitrate additions in enclosure and whole pond experiments. Similar shifts in species composition were reported by Brownlee and Murphy (1983) and Leonardson and Ripl (1980) using inorganic N additions to increase N:P ratio. However, Lathrop (1988) reported that ammonium nitrate additions to a hypereutrophic lake over two seasons failed to prevent the development of the summer blue-green blooms typical of the system (primarily *Anabaena* spp. and *Microcystis aeruginosa*). Lathrop (1988) concluded that low DIN did not trigger bloom development or cause the decline of the vernal non blue-green species and that low TN:TP ratios are resultant from these blue-green blooms and not causative (on account of excess DIP and luxury P uptake). However, unlike other studies referred to, in which shifts to non-blue-greens were induced, the dominant blue-green species typical in this lake was not an N_2 fixer. In cases where N_2 fixers are dominant, increasing N_2 availability may result in replacement by other non-fixing blue-greens.

In this study whole pond additions of inorganic N were added over a three month period in a hypereutrophic detention pond typically dominated by N-fixing *Anabaena* during the summer months. The aim was to investigate phytoplankton and zooplankton community response to altered N:P ratios.

1.9.3 Biomanipulation

The role that biomanipulation practices may play in conjunction with other lake management/restoration efforts, may provide very significant contributions (Shapiro and Wright, 1984). This may be particularly relevant in small shallow eutrophic systems in which food web loops can be readily manipulated and, subsequently, quantified. Although a number of biomanipulation efforts were carried out as part of this research project, these efforts are continuing and follow up data is being collected. Results of these interventions are not reported in this thesis. However, a brief discussion of such approaches and identification of manipulations with potential in SWDPs is relevant.

Biomanipulation concepts have been reviewed by Gophen (1990). While the term may include restructuring of various biotic components, in most instances it is the role of the fish community structure in nutrient recycling and predation on primary consumers that receives most attention. Efforts to reduce P release from sediments may require removal of benthivorous fish on account of their sediment riffling behaviour and excretion which combine to increase P recycling (Andersson *et al.,* 1978). In some cases, stocking of phytophagous fish (in particular silver carp) have been carried out. Miura (1990) reported a decreased *Microcystis* biomass and concomitant increase in the biomass of small green algae in a system stocked with both silver carp and fathead carp. Parr and Clarke (1992) have reviewed prospects for phytophagous and piscivorous fish manipulations to control algal levels in U.K. reservoirs and, in particular, blue-greens.

Size selective predation by zooplanktivorous fish such as minnows and the juveniles of many species has been well documented (*e.g.,* Furnass, 1979; Helfrich, 1976) This selective predation may eliminate or greatly reduce desirable large-bodied cladoceran zooplankton from the assemblage, with replacement by smaller species. Morin *et al.* (1991) demonstrated reduced zooplankton and increased primary production in experimental ponds in the presence of fathead minnows (*Pimephales promelas*). Increased zooplankton, increased transparency, and an increased macrophyte coverage response have been reported in several studies following fish kills involving planktivores (*e.g.*, Hanson and Butler, 1994). Therefore, the introduction of a suitable piscivore to reduce zooplanktivore numbers may result in periods of improved water clarity if herbivorous zooplankton biomass can be increased. However, it is also possible that blue-green species may be encouraged since preferential grazing on greens by zooplankton may remove former nutrient and light competition by the latter group.

Large populations of fathead minnows are characteristic of the SWDPs monitored in this study, and predation pressure on them is virtually nil. A very limited biological structure is a result from their being artificial systems and, in some cases, providing very poor overwintering conditions. The potential use of northern pike (*Esox lucius*) as a top level piscivore in lake restoration efforts is well recognised (*e.g.*, Grimm and Backx, 1990; Parr and Clarke, 1992). In 1995, limited northern pike introductions were made to one of the SWDPs reported here.

1.10 Scope of This Thesis

1.10.1 Project background

On the Canadian prairies the major Cities of Edmonton, Calgary, Winnipeg, Saskatoon, and Regina have SWDPs, with the majority of these ponds being in residential areas. Shallow depth, high external nutrient loads, sediment accumulation, high summer temperatures/radiation, and polymixis combine to produce hypereutrophic conditions within the ponds. Extended blooms of non-fixing blue-greens and green algae are characteristic, with periodic development of N-fixing species. Public complaints about poor water quality prompted the Cities to fund work to better understand the systems, and identify techniques which may improve water quality.

The work presented in this thesis is based on data collected from three SWDPs and their respective catchment areas (two in Regina, one in Saskatoon). The work was carried out between the winter of 1991/92 and the autumn of 1995 as part of a contractual agreement between the Saskatchewan Research Council [SRC] and the Cities of Regina and Saskatoon.

1.10.2 Geographical location and characteristics

The Province of Saskatchewan is characterized by a five month winter (permanent ice cover) with snowmelt from March to April. The summer climate is semi-arid, in which intense thunderstorms of short duration provide the bulk of the precipitation. During the summer months, the Province typically records a high average of bright sunshine hours per day. The mid- to southern portion of the Province is mostly flat prairie, much of which is under intensive arable crop production. The flat nature of the southern/central landscape predisposes frequent winds. Figure 1.4 shows the location of the two major Cities in the Province (about 300 km apart) in which the studies were carried out.

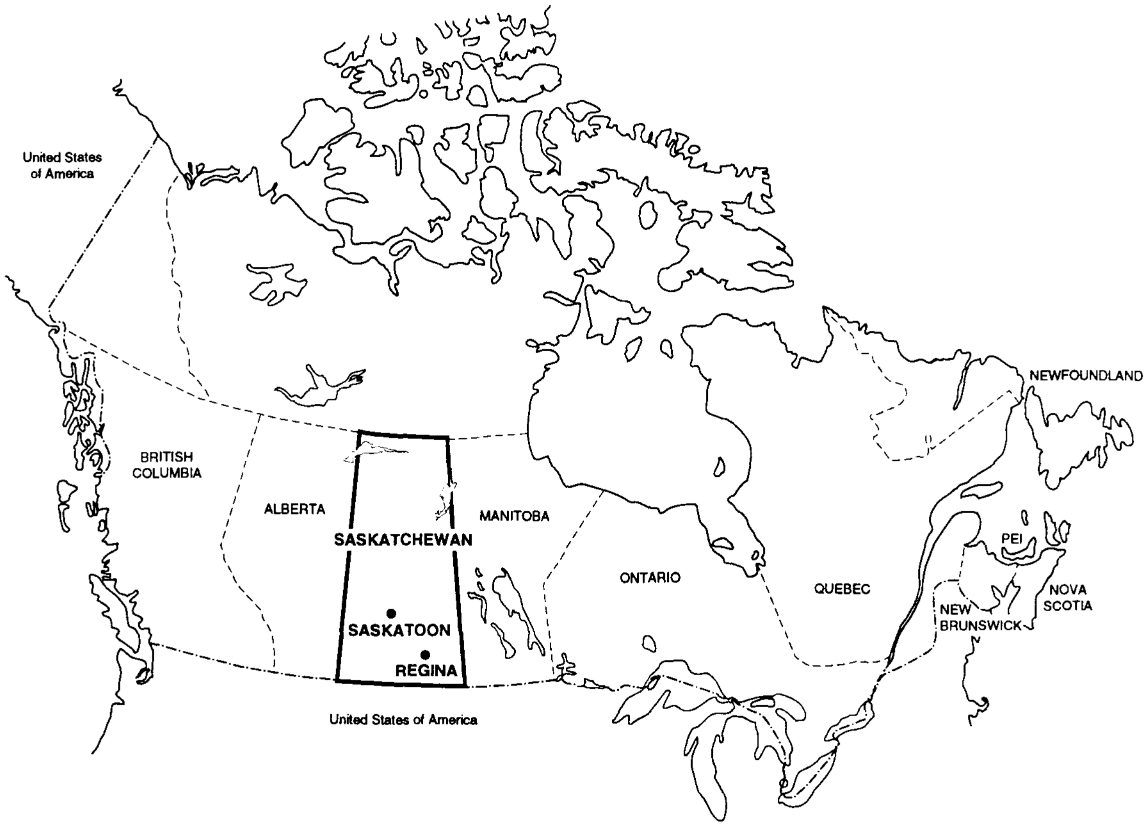

Figure 1.4 Geographical location of the Province of Saskatchewan, and the two major Cities of Saskatoon and Regina.

1.10.3 Project aims

The aims of the project were as follows:

(1) To compile a data set describing the physical, chemical, and biological (primarily phytoplankton and zooplankton) characteristics of SWDPs located in the province of Saskatchewan. Seven ponds were investigated, of which three are reported in this thesis.

(2) To assess sediment characteristics among the ponds, and determine their potential as a sink and source of P.

(3) To examine typical stormwater runoff quantity and quality for a prairie urban catchment, with a view to estimating the nutrient load and P removal efficiency of these impoundments.

(4) To assess the effect of increasing N availability in an N limited SWDP on the phytoplankton/zooplankton composition and density.

(5) To assess the potential for the use of P precipitation agents to reduce phosphorus and phytoplankton biomass for improved water quality.

Chapter 2

MATERIALS AND METHODS

2.1 Monitoring Routines and Water Sample Collection

For the period 1992 to 1995, open-water monitoring (April to September) of the Rochdale and Lakewood ponds (Regina) and Lakeview pond (Saskatoon) was mostly carried out at weekly or two-weekly intervals. During experimental periods, sampling frequencies were increased to twice or three times a week. This included the duration of Lakewood pond N additions (May to July 1994) and certain periods during the Lakeview pond catchment runoff study in 1994, according to rainfall. During 1995, the Lakeview pond was monitored less frequently compared to previous seasons (fortnight or monthly interval). During the winters of 1991/92 and 1993/94, sampling under the ice was carried out on a fortnight to monthly basis in the Lakeview pond.

For the shallow Rochdale and Lakewood ponds integrated column samples of 1.5 m depth were collected at a central location. During experimental N additions to Lakewood pond in 1994, integrated samples from both the north and south ends were collected. For the deeper and stratifying Lakeview pond, an integrated top water sample (0 to 1.5 m) and a single bottom sample (2.6 m) were collected from a central location in 1992/93. Sampling protocols in 1994 and 1995 were expanded to include top and bottom samples from both a north and south location in Lakeview pond.

The column sampler consisted of a rigid nylon tube with marked graduations and stoppered with a silicon bung. Bottom samples on Lakeview pond were collected at 2.6 m with a 2-L messenger activated sampler. Samples were stored in 2-L plastic containers, pre-rinsed with raw lake water. Samples were filtered as soon as possible in the laboratory and raw and filtered water transferred to pre-rinsed 500-mL plastic bottles. All filtration was done using Whatman GF/C filters. Samples were refrigerated at 4 °C prior to analysis. When City of Regina samples were transported overnight by bus, samples were always filtered prior to shipping.

2.2 Sedimentation Traps

The greater depth of Saskatoon's Lakeview pond provided the opportunity to measure sedimentation rates, notwithstanding the potential for interference due to sediment resuspension during stormflows. Duplicate traps were installed at a central location for the period June to September 1994, and at both north and south locations from May to September 1995. Traps were located at 2.4 m depth and consisted of duplicate 500 mL containers (orifice 2.5 cm diameter) clamped approximately 25 cm apart on either side of a section of ABS (acrylonitrile butadiene styrene) pipe. The pipe was weighted at the bottom end and in turn supported at the surface by a float. This setup was held in position at the centre of a 10 m line running between two anchored floats. The setup minimized lateral movement and prevented sediment disturbance. The traps were collected on every monitoring day and replaced with fresh containers filled with distilled water at 5 to 8 °C.

2.3 Stormwater Runoff Sampling, Precipitation, and Pond Depths

2.3.1 Lakeview stormwater detention pond (Saskatoon)

In 1994 and 1995, automated sampling of stormflows was carried out in the largest of three subcatchments draining to the Lakeview stormwater detention pond [SWDP] using an ISCO water sampler, in conjunction with a Millitronics sonic head sounding device and data logger system for hydrograph measurement. An electronic sensor initiated the water sampler when the water level rose 5 cm above baseflow level. Upon initiation, 28 discreet samples were collected at 2 to 6 minute intervals. The sampler was located in the last manhole on the storm sewer upstream of the detention pond, while the sonic head was located in the second last manhole. The catchment perimeter was instrumented with three tipping bucket rain gauge/data logger systems at spaces of 570 and 1061 m. One Shape 3200 pressure probe was used to measure the head on the sharp-crested weir at the detention pond outlet. Instrument location details and operational information have been described by Raymond *et al.* (1995).

2.3.2 Rochdale stormwater detention pond (Regina)

In June 1993 and 1994, two complete stormwater inflows entering the Rochdale detention pond *via* the south-west storm sewer were sampled using an automated sampler housed above ground. The sampler was designed and constructed by SRC's Instrumentation Section. Sampling was initiated by an optical sensor located at the bottom of the sewer. Collection was set at 5 minute intervals, with every three consecutive samples pooled into one of 24 collection bottles. A malfunction in sampler programming led to the sampler being removed permanently from this site in July 1994.

During 1994 and 1995, the head on the outlet weir of the Rochdale pond was measured using a Stephens chart recorder housed within a box/culvert structure in the pond. Chart paper was frequently removed and weekly checks on water height were made at the outlet weir to maintain calibration. Precipitation data was obtained from the Regina Airport Climate Station (AES, Canada). All outflow volumes were calculated using a standard equation for a sharp-crested weir (Smith, 1985):

Equation 2.1 $$Q = C \bullet b \bullet h^{3/2}$$

where:

Q = outflow volume (m^3/sec)

$C = 2/3\sqrt{2}\,g\,(0.605 + 0.001/h + 0.08\,h/P)$

h = head on weir (m)

P = height of weir (m)

b = weir length (m)

2.4 Physical and Chemical Water Analysis

2.4.1 Dissolved oxygen and temperature

Oxygen and temperature were determined in the field at 0.5 m depth intervals using a portable O_2/temperature metre (model 51B, Yellow Springs Instrument Corp. Yellow Springs, Ohio, USA).

2.4.2 pH

pH was measured on all samples using an Orion Research Expandable Ion analyser EA 940 pH meter. Field samples were also measured using a portable meter (Canlab H5503-11).

2.4.3 Alkalinity

Alkalinity was determined acidimetrically with indicator end points (Golterman *et al.,* 1978).

2.4.4 Nitrogen, phosphorus, silica, and carbon

Between 1992 and 1993, N, P, and Si analyses were read manually on a Gilford Stasar III spectrophotometer. Thereafter, analyses were performed by manual addition of reagents to a 96-well microplate which was read in a Dynatech MR7000 automated plate reader.

Nitrate-N: was determined colorimetrically after an alkaline hydrazine reduction to nitrite using an adaptation (manual) of the EPA method 353.1 (USEPA EPA-600/4-79-020, 1983).

Ammonium-N: was determined colorimetrically by adaptation (manual) of the phenate method outlined by Stainton, Capel, and Armstrong (1977).

Particulate-C and N: C/N retained on a 2.5 cm Whatman GF/C filter was determined on a C/N autoanalyser at the Department of Fisheries and Oceans Freshwater Institute, Winnipeg, Manitoba.

Total dissolved-N: was measured colorimetrically as nitrate concentration (as above) after alkaline persulphate digestion of filtrate in Teflon® lined bottles by autoclaving for 30 minutes (Koroleff, 1972). Nitrate standards and sample spikes were carried through the entire procedure.

Total-N: was calculated as the sum of particulate-N + total dissolved-N.

Total Kjeldahl-N: was measured in stormwater runoff samples by University of Saskatchewan according to APHA (1985).

Phosphate-P: was determined colorimetrically using a manual adaptation of the molybdenum blue method outlined by Stainton, Capel, and Armstrong (1977).

Total-P/total dissolved-P: was determined by acid persulphate digestion using Teflon® lined bottles and autoclaving for 30 minutes (APHA, 1985). Digestions were neutralized and phosphate determined colorimetrically as above. P standards and sample spikes were carried through the entire procedure.

Particulate-P: was obtained by subtraction of total dissolved-P from total-P. For sediment trap PP, material retained on Whatman GF/C filters was combusted at 550°C for two hours prior to acid persulphate digestion as above.

Silica: was determined colorimetrically by a heteropoly blue method of Stainton, Capel, and Armstrong (1977). All analyses were performed with plastic receptacles to prevent possible contamination from glass.

***Dissolved organic carbon*:** was determined on a Technicon Autoanalyser (System II) using a phenolphthalein colorimetric method developed by Crowther and Evans (1980).

***Dissolved inorganic carbon species*:** were calculated from alkalinity, pH and temperature data (Mackereth *et al.*, 1978).

2.4.5 Calcium, aluminium, iron, and manganese

For dried sediment samples total elemental composition was determined by SRC's Geochemistry Laboratory by atomic absorption after perchloric/ perfluoric acid digestion.

2.4.6 Suspended solids and loss on ignition

Samples were filtered through pre-weighed, pre-ashed Whatman GF/C filters, flushed with 10 mL distilled water, and dried at 70°C for 24 hours. Loss on ignition [LOI] was determined as weight difference of the oven-dried filters or sediment samples after 2 hours at 550°C. All analyses were done in duplicate.

2.5 Biomass and Pigment Measurements

2.5.1 Chlorophyll

Chlorophyll was determined by extraction of pigment on Whatman GF/C filters in $MgCO_3$ saturated 100% methanol. Extraction was overnight and samples were kept in darkness and refrigerated at 4°C. After extraction, filters were removed and samples spun at 4000 g (Centra-4b IEC) for five minutes and read in 1-cm cuvettes (Perkin-Elmer-124 Double Beam Spectrophotometer) at 650, 665, and 750 nm. Phaeophytins were not measured. Chlorophyll *a* and *b* concentrations were calculated using the extinction coefficients of MacKinney (1941).

2.5.2 Phytoplankton

Upon collection, phytoplankton were preserved in Lugol's iodine and stored in the dark at 4°C prior to enumeration. Counts were made using the inverted microscope technique on a Zeiss Axiovert 135 microscope, using the keys of Prescott (1962). Counts were accepted after a minimum of 100 individuals of each species had been counted (at least two right angled diagonal transects were always completed), or the entire chamber floor had been counted and duplicates agreed ±10%. Biovolume was calculated by measuring average cell dimensions (at least 100 units for *Oscillatoria* trichomes or the entire chamber floor, and 25 units for unicellular and colonial species) and using the geometric formulae that most closely approximated the cell shape as given by Willén (1976). Where different size classes of a particular species were evident, these were treated separately. *Microcystis* and *Aphanocapsca* biovolumes were estimated from average colony surface areas multiplied by 2 times individual cell diameter. *Coelospherium* surface areas were multiplied by 1 times the cell diameter. Fortunately, problematic colonial forms of this kind were rarely significant. *Anabaena* biovolume was calculated by averaging the number of cells in at least 25 filaments or the entire chamber floor, and totals calculated assuming spherical cell dimensions. Heterocysts were only counted separately on a few occasions.

2.5.3 Zooplankton

Samples were collected by straining approximately 4.2 L (5*1.5 m integrated column samples) of pond water through a mesh of 100 micron diameter sealed over a plastic funnel. The animals were backwashed into a vial with distilled water and preserved with ~10% formaldehyde. Zooplankton sampling was carried out in conjunction with all other water sampling (no under-ice samples were collected). On ponds where more than one sample was taken, these were combined as one composite sample. In 1992, samples were only analysed on a dry weight basis after cursory inspection. Thereafter, enumeration and species composition was evaluated by counting subsamples under a binocular microscope using the key of Edmonson (1966).

2.6 Sediment Sampling and Experimentation

2.6.1 Sediment collection

In 1992, sediment samples were collected using an Eckman dredge sampler and carefully removing the surficial material (~1 cm). Samples were stored in acid washed plastic vials filled to allow no head space and refrigerated prior to freeze drying. From 1993 onwards, sediment samples were always taken as intact cores using a home-made device consisting of an extendable pipe section terminating in a light weight ball valve mechanism. The upper portion of the removable coring tubes (5-cm i.d. polyethylene) could be conveniently locked in position below the valve. The device gave manual control during coring, and provided effective vacuum for lifting the core. Intact cores with overlying water were transported in a dark ice-cooled container.

2.6.2 Sediment depth

Sediment depth was averaged from cores taken at representative locations throughout the ponds, and taken as the distance between the sediment surface and the visible clay plug of the pond liner.

2.6.3 Sediment analytical protocol

Effort was made to process all sediment samples within 24 hours of collection and usually within 6 hours. When overnight storage was necessary, cores were stored at 4 °C in the dark with the overlying water. After syphoning of the overlying water, cores were extruded into a matching section of graduated tube and sectioned into 1, 2, 3+4, 5+6, 7+8, and 9+10 cm horizons. Fractions were centrifuged for 5 minutes at 4000 rpm (Centra-4b IEC). pH was measured on raw interstitial water to prevent degassing. Interstitial water was filtered through Whatman GF/C filters in air prior to chemical analysis. Subsamples for metal analysis were acidified with 1 to 2% nitric acid. Moisture content was measured in duplicate by weight difference after drying to stable weight at 70 °C for 48 hours. Sequential extraction of sediment P was performed on freeze-dried samples (without removal of interstitial fluid), following the method of Hieltjes and Lijklema (1980). Alkaline and acid extraction solutions were neutralized prior to phosphate analysis. Where precipitation of humic compounds interfered with analysis, colour-developed samples were centrifuged prior to spectrophotometry. To correct for background colour, all interstitial and sequential extraction solutions were analysed in conjunction with sample blanks.

2.6.4 Nutrient uptake and release experiments with sediments

A number of intact core incubation experiments were carried out in batch mode, under aerobic and anaerobic conditions at various temperatures. Whatman GF/C filtered detention pond water was used as the reservoir media in all cases. Lakeview pond cores were incubated in replicates of 8 to 16 cores under anaerobic conditions at 5, 10, and 20°C in a temperature controlled growth room in the dark. Cores for these experiments were collected in the field at times when the ambient in-pond temperatures were close to the temperature at which the cores were to be incubated in the laboratory. Rochdale pond cores were incubated under anaerobic conditions at 20°C only. A single set of cores from Rochdale and Lakewood ponds were incubated aerobically at 10°C only. Experimental duration was between 14 and 21 days, and all cores were allowed to equilibrate for at least 24 hours before measuring.

Anaerobic conditions were maintained in each core by continuous sparging with N_2 gas of the overlying reservoir through a small diffuser stone, with gas flow to each stone controlled from a multiple valve manifold. The near-surface sparge was low enough to provide only very gentle mixing of the overlying water. Aerobic conditions were achieved in the same way with wetted pressurized air. Any evaporative losses were replaced with distilled water prior to sample removal. Release/uptake rates were measured by removing 50 mL daily or 100 mL every two days from a 300 mL reservoir and replacing this volume with filtered pond water of known composition. Prior to analyses, all samples were Whatman GF/C filtered. pH was measured in all experiments on each sampling day, and dissolved oxygen [DO] was checked in representative cores using a YSI oxygen meter. After two days from the beginning of each release experiment, four to six cores were sacrificed for analysis of interstitial phosphate and ammonium.

Aerobic/anaerobic P adsorption isotherms were calculated for Rochdale and Lakewood pond cores by taking the top 1+2 cm fractions from fresh cores without surface oxidation (assumed by a light tan colour); 0.5 to 1.0 g of wet sediment was incubated in triplicate in 50 mL of K_2HPO_4 solution (distilled water) across a range of 0 to 50 mg/L P. Sediment oxidation was achieved by rotating tubes for 48 hours exposed to air at 20°C, with vigorous manual shaking several times a day. Anaerobic conditions and suspension were maintained by sparging the incubation solution with N_2 gas for 24 hours. Replacement of weight loss with distilled water was used to correct for evaporation. Prior to analysis, solutions were centrifuged at 4000 rpm for 5 minutes and Whatman GF/C filtered.

2.7 Field Experimental Procedures

2.7.1 Inorganic nitrogen additions to Lakewood pond

Additions of granulated ammonium nitrate fertilizer (Sherritt 35-0-0) were made on a weekly and two weekly basis according to utilization rate, weather, and apparent phytoplankton response from May 10 to July 20 1994. Additions were calculated to give 2 to 3 mg/L or 3.4 to 5.1 g/m^2 of N, based on a 200 to 300 kg fertilizer addition. Fertilizer was dissolved by towing the chemical in a gunny sack behind a boat and placing powder on upwind littoral slopes in windy conditions. Rapid dispersal of the dissolved fertilizer was confirmed by visible plumes. Throughout the experimental period, water samples were taken at north and south locations before addition, approximately two hours after completion of additions, and again the following morning.

2.7.2 Aluminium sulphate additions to Rochdale pond

Phosphorus inactivation was carried out in Rochdale pond by applying 6600 L of liquid aluminium sulphate (4.44% as Al) to the prop-wash of a traversing boat outfitted with 10 hp outboard motor, while a second boat was used to maintain a gentle mixing behind the line of application. The chemical was pumped from the shoreline *via* 500 feet of 1-inch i.d. flexible hose using a gas powered propeller pump. That portion of the hose in the pond was supported by a series of 6-foot cylindrical foam floats at approximately 10-foot intervals to prevent sediment disturbance and hindrance of the boat. The chemical was applied in two batches, one from each end of the pond, and application time was approximately 2.5 hours. After 75% of the solution had been added, pH was monitored at representative locations. Water samples were collected and pH measured approximately 4 hours after completion and again the following morning, after which weekly monitoring was resumed.

2.8 Statistical Analyses and Software

Regressions were generated using SigmaPlot 3.0 and SigmaStat 3.0 programs (Jandel Scientific, California). Parametric and non-parametric statistical tests as given by Zar (1984) were performed using SigmaStat 3.0.

Modelling of stormwater runoff for the Lakeview pond catchment was performed using the runoff block of the USEPA PC-SWMM software, version 4

(Huber and Dickinson, 1988) by Blair Raymond, formerly a Masters student in the Department of Civil Engineering, University of Saskatchewan (Saskatoon), under the supervision of Professors Jim Kells and Gord Putz.

Chapter 3

PHYSICAL, CHEMICAL, AND BIOLOGICAL CHARACTERISTICS

3.1 Introduction

This chapter summarizes the physical, chemical, and biological characteristics of the Rochdale, Lakewood, and Lakeview stormwater detention ponds [SWDPs]. The data are from monitoring work carried out between the winter of 1991 and the autumn of 1995. Data for periods involving whole pond experimental work (N additions to Lakewood pond, 1994; aluminium sulphate additions to Rochdale pond, 1995) are included within some of the figures, however, this data is discussed later in Chapters 6 and 7.

3.2 Results

3.2.1 Physical and morphometric characteristics

All of the stormwater detention ponds in this study are serviced by separate storm sewers. Design characteristics for the ponds are given in Table 3.1.

Lakeview stormwater detention pond (Saskatoon): The catchment area for Lakeview pond is fully developed and has three storm sewers draining into the pond. One of these storm sewers drains only a small residential pocket, so that the bulk of stormwater loading enters *via* two sewers, entering at the north and south ends of the pond. There is potential for short-circuiting of inflows from the north storm sewer to the outlet weir, which is also located at the north end of the pond. The pond shape is essentially oblong. In 1994, the total catchment area was reviewed as part of a runoff modelling study, and was calculated as 78.36 ha, comprising 62 subcatchments of 0.18 to 4.32 ha, with ground slopes of 0.15% to 2.62% (Raymond, 1995). The area is primarily residential, with a small commercial development, a church, school, and parkland within the area.

The Lakeview detention pond is the deepest pond in both Saskatoon and Regina (Z_{max} = 2.75 m), and the only one that undergoes distinct temperature stratification during the summer. The pond is fairly well sheltered, and receives a variable amount of cool groundwater baseflows, so that the influence of wind induced mixing is decreased compared to the more open and shallow Regina ponds.

Rochdale and Lakewood stormwater detention ponds (Regina):

Rochdale and Lakewood ponds are neighbouring ponds of the same age and approximate surface area and depth, draining similar catchment types. The Rochdale pond service area is primarily residential, consisting of: 525 single family dwellings; 66 multiple family dwellings; 2 schools/recreation facilities; and 2 churches. The Lakewood pond service area is also residential, consisting of: 709 single family dwellings and 1 school. Both catchments also have parkland.

Important differences between these two ponds are that the Lakewood catchment is fully developed, while Rochdale has 16 ha currently (1996) under development, 10 ha undeveloped, and receives additional drainage from 20 ha of agricultural land. These differences have implications for slightly different nutrient loading characteristics and water residence times. Other differences include inflow pipe locations and shape: Rochdale is kidney-shaped and receives inflow from two ends, each equidistant to the outlet weir; while Lakewood is longitudinal in shape and receives stormflows from two pipes, each entering at the north end, with the outlet weir at the south end. Due to elevation and sewer networking, the outlet weir in Lakewood can also temporarily act as an inlet during high volume flood events. All three ponds have a single horizontal weir outlet housed within an underground concrete vault.

Table 3.1 Physical and morphometric characteristics of Rochdale, Lakewood, and Lakeview stormwater detention ponds.

		Rochdale	Lakewood	Lakeview
Catchment Area (ha)		89.7	68.4	78.4
Surface Area (ha)	LWL	2.1	2.1	2.2
	HWL	2.43	2.23	2.6
Pond SA /catchment ratio		0.021	0.031	0.028
Design runoff coefficient		0.63	0.63	0.65
Total % impervious		nd *	~35	61.74
% directly connected impervious		23 *	29	36.44
Number of inlets		2	2	3
Permanent Storage	Max Depth (m)	1.73	1.83	2.75
	Vol (m^3)	32500	30400	50700
Operating Storage (Max)	Storm Design	25 yr	25 yr	25 yr
	Depth (m)	3.6	3.6	3.9
	Vol (m^3)	80000	80000	93000
Littoral Slopes (LWL-HWL)		1/7	1/7	1/7
Year Constructed		1981	1981	1979

nd = no data
* development ongoing

3.2.2 Water temperature

Water temperature at 1 m for City of Regina ponds is represented by data from Rochdale pond for the study period (Figure 3.1). In these shallow ponds temperature maxima were in the range 18 to 24°C, being slightly higher in 1994 and 1995. Weak stratification was observed on a few occasions during hot and calm conditions, and the systems were predominantly well mixed.

Lakeview pond (Saskatoon) generally developed temperature stratification from late May until September (Figure 3.2). The average mixed layer summer temperatures were in the range 17 to 22°C, with periodic declines due to cooler stormwater runoff volumes. Bottom water temperatures in Lakeview pond were as much as 5°C higher in July and August of 1992, compared to the differences measured in other years.

Apart from 1992, the average summer temperature to which the sediments in Lakeview pond were exposed was 12 to 14°C, with a top and bottom water temperature difference in the range 5 to 10°C. Some influence on the temperature difference between top and bottom water, and on bottom water temperature stability in Lakeview pond may involve groundwater baseflows entering continuously below the thermocline.

Figure 3.3 shows an exemplary sequence of winter and summer temperature profiles from 1994. Increasing reduction of the mixed layer was typical as the season progressed, but stratification was variably disrupted according to the magnitude of stormflows. The thermocline was generally located in the 1.5 to 2.0 m region.

3.2.3 Dissolved oxygen

Open water: As a result of high photosynthetic oxygen production, all ponds frequently showed oxygen levels in excess of saturation in the upper water during daytime measurements. For the shallow Regina ponds, weak oxyclines from 0.5 to 1.5 m were only evident on bright days with little wind, and when the upper water temperature was high and Secchi disk depth low. Wind mixing, and a euphotic zone in the 0.5 m to sediment surface region tended to maintain an aerobic water column profile. Both Rochdale and Lakewood ponds displayed similar seasonal trends in oxygen peaks, indicating similar productivity ranges and controls. Exemplary oxygen data for these ponds from the 1994 season is shown in Figure 3.4.

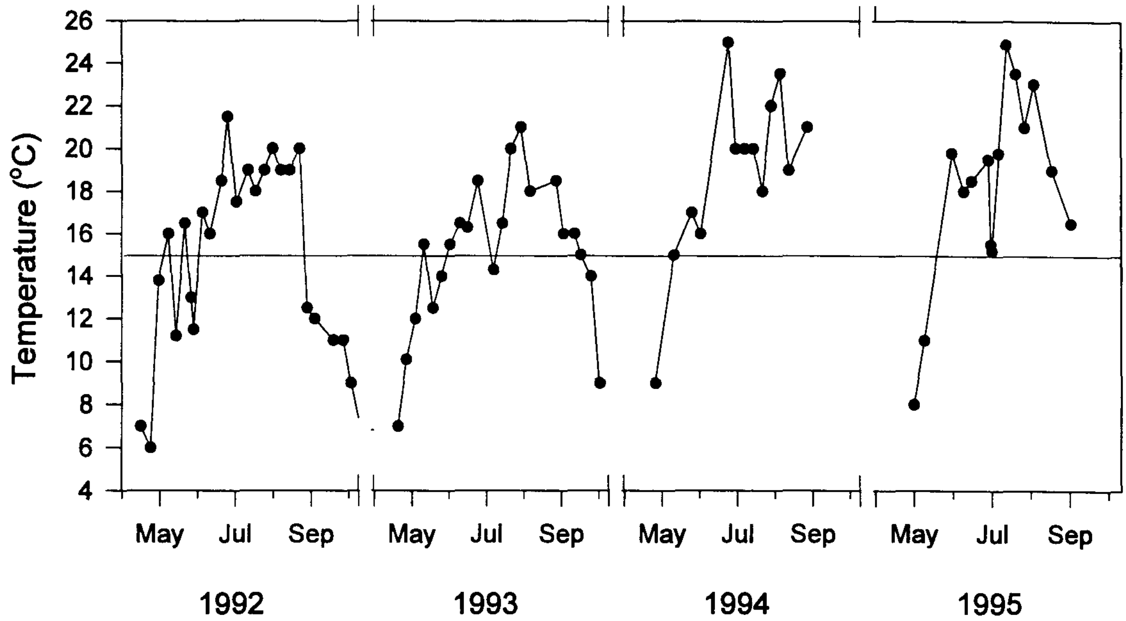

Figure 3.1 **Water temperature at 1 m in Rochdale pond (Regina), 1992 to 1995.**

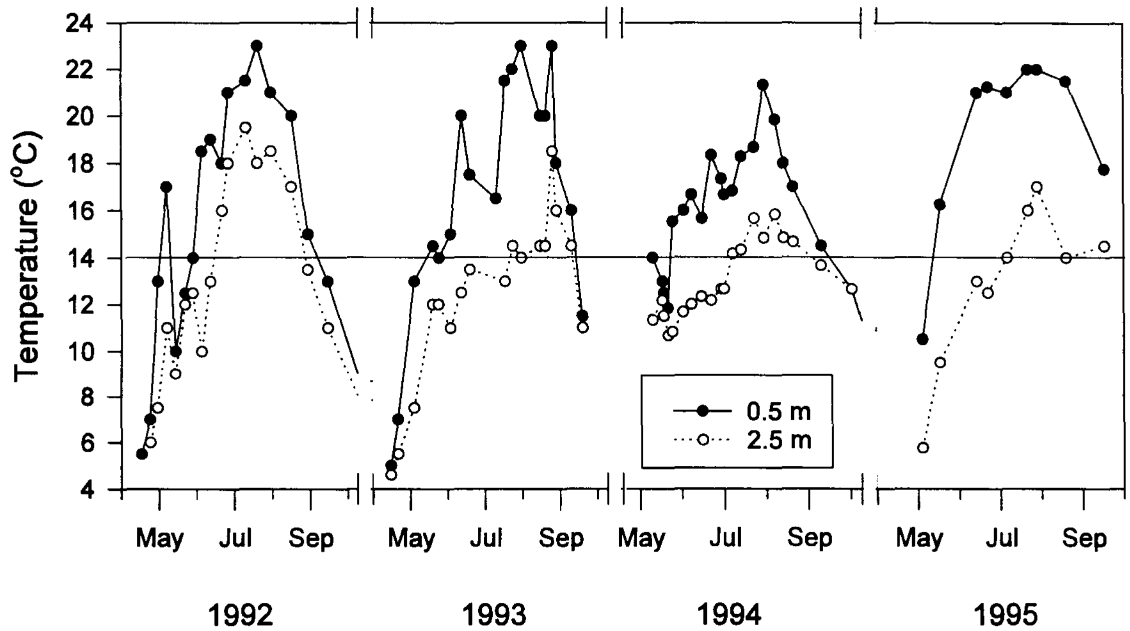

Figure 3.2 **Top (0.5 m) and bottom (2.5 m) water temperatures in Lakeview pond (Saskatoon), 1992 to 1995.**

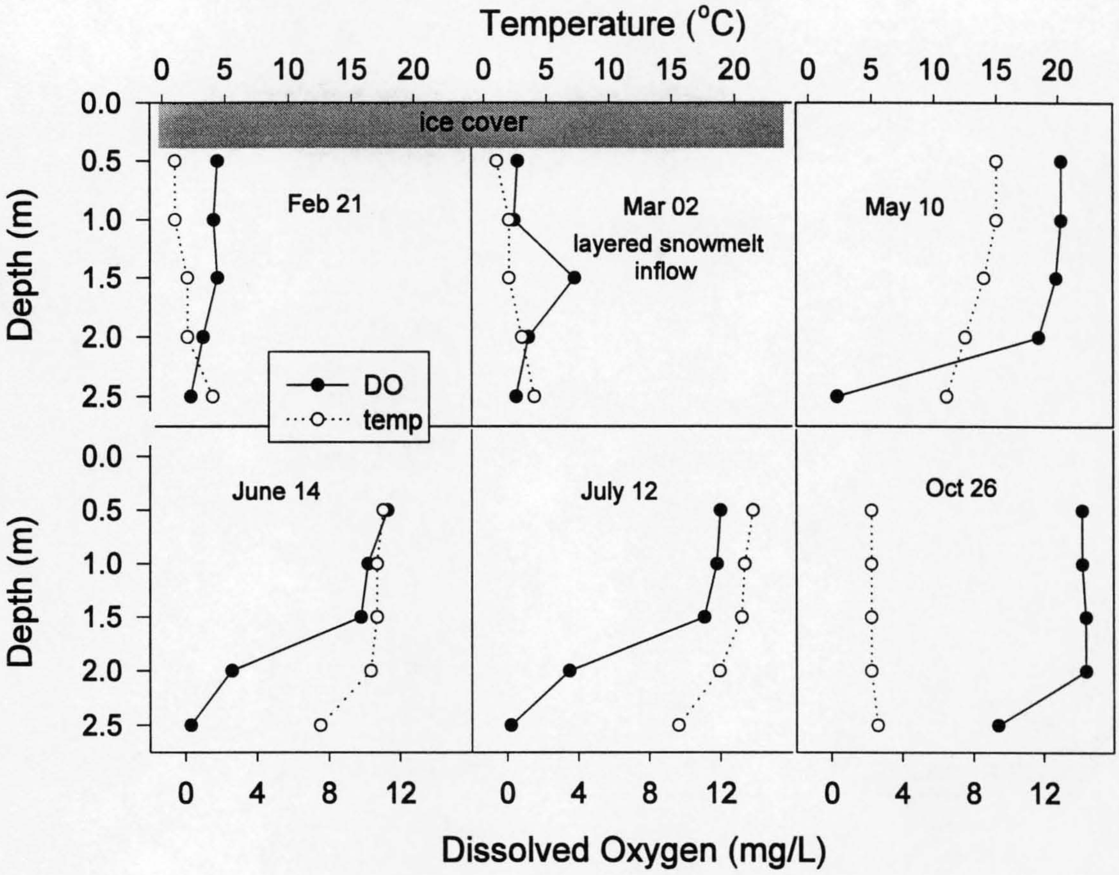

Figure 3.3 Typical seasonal sequence of oxygen and temperature profiles in Lakeview pond (Saskatoon), 1994.

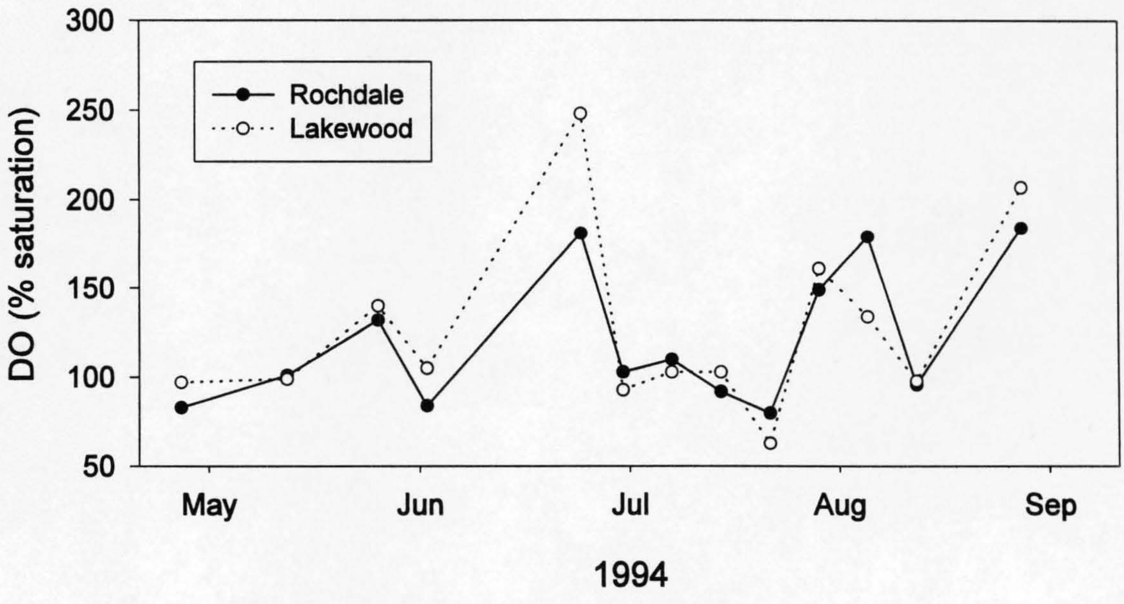

Figure 3.4 Oxygen saturation at 1 m depth in Rochdale and Lakewood ponds (Regina), 1994.

For Lakeview pond, a distinct oxycline was typical, and the sediments were overlain by anaerobic hypolimnetic layer (< 0.4 mg/L DO at 2.5 m depth). The typical sequence of oxycline formation for the 1994 season is shown in Figure 3.3.

Oxygen depletion under winter ice cover: Dissolved oxygen depletion under winter ice cover has consequences for biotic survival and diversity, and the potential for sustainable biomanipulation efforts (*e.g.,* Ellis and Stephan, 1989). Only a limited number of measurements were made under winter ice cover in Rochdale and Lakewood ponds. Since ice cover of 0.5 m is typical by mid-winter, and winter draw-down has also been practised, anoxia rapidly developed in these shallow ponds after freeze-over. Winter aeration of the ponds is not carried out since the ponds are used as public skating rinks.

For Lakeview pond, winter oxygen depletion rate [WODR] was measured in 1991/92, and 1993/94 (total of 10 measurements). Dissolved oxygen isopleths for the 1993/94 winter are shown in Figure 3.5, beginning three to four weeks after complete freeze-over (percent saturation at the time of complete sealing was not measured). Dissolved oxygen data from the two winters were pooled, and the depletion rate was plotted against the areal mass of oxygen measured at the start of each depletion period. Mass oxygen was calculated from integrated 0.5 m volumetric data, extrapolated to the sediment surface (as a mean value from two snow covered ice holes). A second order relationship gave a reasonable description of the mass oxygen depletion rate (Figure 3.6).

Net oxygen consumption in Lakeview pond appeared to reach a maximum rate at mass O_2 >20 to 25 g/m^2 in the range 0.31 to 0.35 g O_2/m^2/day within a 1 to 4°C temperature profile. As the areal mass of O_2 declined, the consumption rate declined. Using data from 13 lakes in central Alberta, Babin and Prepas (1985) also reported non-linear WODRs. Baird *et al.* (1987) also reported similar O_2 depletion relationships in two small Alberta ponds of similar dimensions to the Lakeview SWDP. One of these ponds was classified as hypereutrophic, with similar pond depth and organic matter content of the sediments to Lakeview pond (LOI ≈ 17 to 22%, see Chapter 4). The maximum O_2 depletion rates calculated in the Lakeview pond were slightly lower than those reported by Baird *et al.* (1987) for the hypereutrophic Alberta pond, but similar in magnitude to those reported by Barica and Mathias (1979) for shallow Manitoba ponds (Table 3.2).

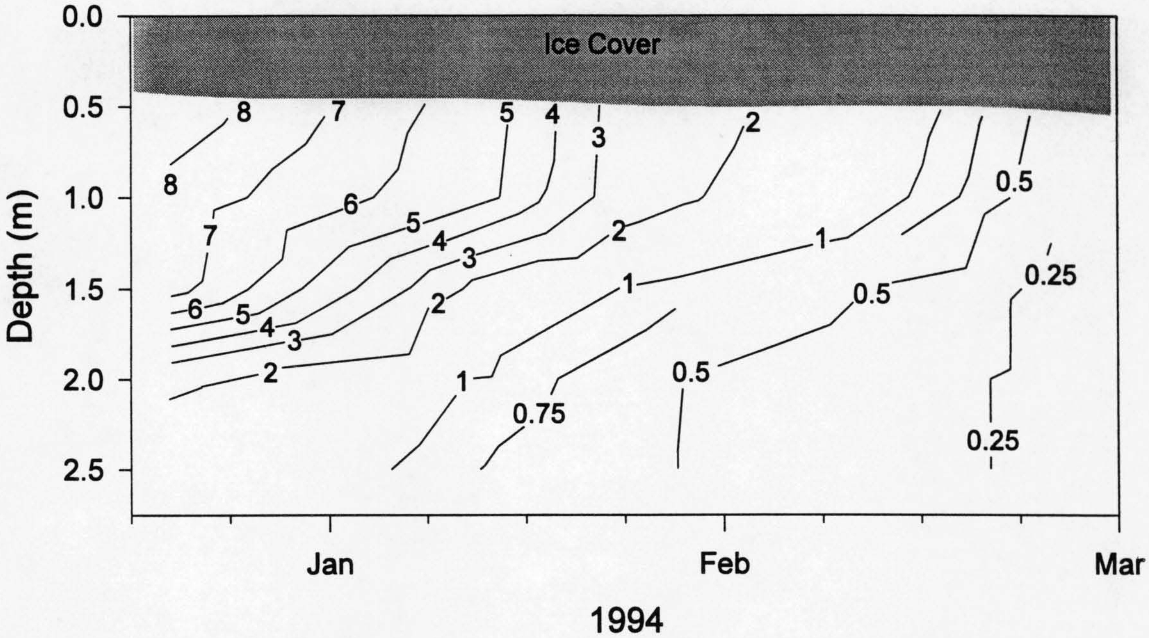

Figure 3.5 Isopleths of winter oxygen depletion rate in Lakeview pond (Saskatoon), 1993/94. (values are as mg O_2/L)

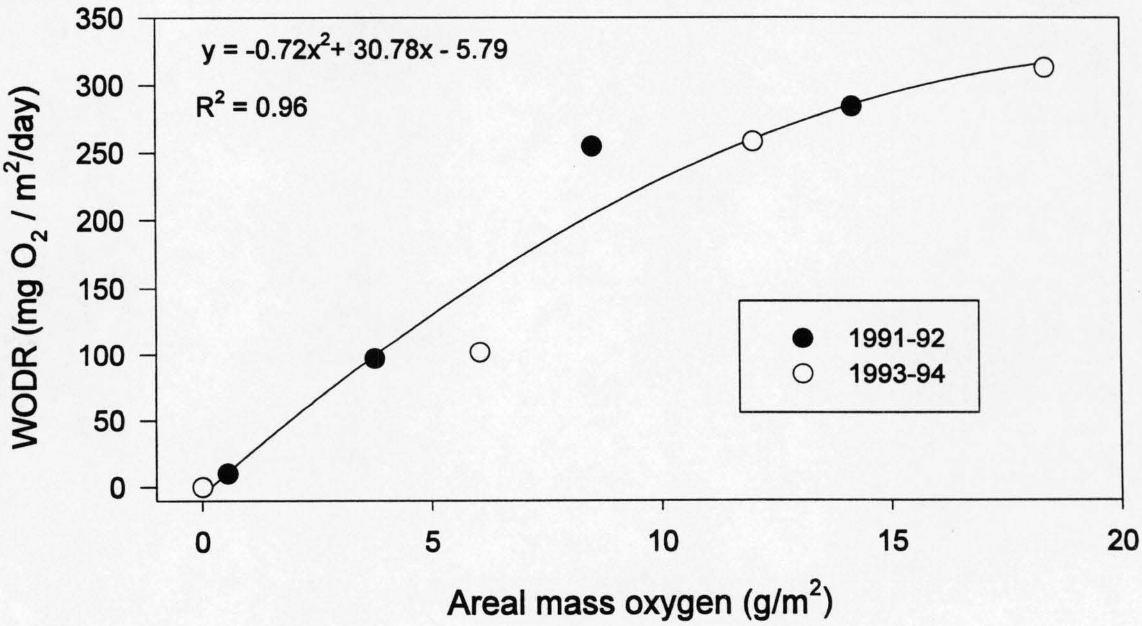

Figure 3.6 Winter oxygen depletion rate [WODR] as a function of the areal mass oxygen in Lakeview pond (Saskatoon).

Estimates of WODR calculated from the depth model of Barica and Mathias (1979) are in good agreement with the Lakeview pond value. Of the three Babin and Prepas (1985) models, regression #4, which includes summer TP as a productivity term, gave the best estimate compared to the models based on Chl *a* and depth, or depth alone (Table 3.2).

Table 3.2 Comparison of winter oxygen depletion rates [WODR] measured in Lakeview pond, with data from other prairie lakes of similar size, depth, and trophic status, including estimated WODR from published relationships. (adapted from Baird *et al.*, 1987)

		Maximum Linear WODR ($g\ O_2\ m^{-2}\ day^{-1}$)
Lakeview SWDP (Saskatchewan)	Observed maximum (40 day)	0.312
Manitoba pothole lakes	Baird *et al.* (1987)	0.617 - 0.516
Alberta pothole lakes	Barica & Mathias (1979)	0.21 - 0.42
Estimated WODR for Lakeview	Barica & Mathias (1979)[1]	0.307
	Babin & Prepas (1985)[2]	0.237
	Babin & Prepas (1985)[3]	0.614 - 0.436 **
	Babin & Prepas (1985)[4]	0.509 - 0.318 **

** based on 1992 to 1994 open water TP:Chl *a* average assuming euphotic zone of 1.5 to 1.0 m, or approximately 2 times the average Secchi disk depth, N=59

[1]WODR = 0.14 + 0.062z (z = mean depth)
[2]WODR = 0.214 + 0.0084z
[3]WODR = 0.093 + 0.00257 Chl *a* + 0.0127z.
[4]WODR = -0.101 + 0.00247 TP_{summer} + 0.0134z.

The observed maximum WODR for Lakeview pond was on the lower end of the ranges predicted by regressions #3 and #4, and the range calculated by Baird *et al.* (1987). This may be the result of: (1) the start of the initial depletion period was about three weeks past freeze-up, which might have increased the initial depletion period average had it been included; (2) partial clearing of snow from the pond for skating may permit some photosynthetic O_2 production, thereby decreasing the apparent depletion rate, since the above models are based on zero O_2 production.

The non-linear nature of the depletion process may involve a number of factors. Since the WODR is primarily due to sediment metabolic rate, the initial period of thermal equilibration between sediment and the water column would be expected to result in the highest WODR (Baird *et al.*, 1987). Also, the amount of easily oxidized organic material may decrease. As lower strata become O_2 depleted, the distance that O_2 must move by diffusion between remaining oxygenated water and the sediment surface increases, so that the O_2 flux rate may decrease (Babin and Prepas, 1985).

3.2.4 Alkalinity and pH

The statistical trends in alkalinity and pH for Rochdale, Lakewood, and Lakeview ponds are summarized in Table 3.3. The long-term trend for alkalinity in Rochdale and Lakewood ponds was 75 to 90 mg/L $CaCO_3$. For both Rochdale and Lakewood, higher than average alkalinity was observed in 1992, which was attributed to a lack of flushing combined with concentration *via* evaporative effects during the dry summer. Throughout the open water period, daytime pH levels were generally high (pH 9.3 to 9.7) in both Rochdale and Lakewood, on account of photosynthetic inorganic C removal; alkalinity varied inversely with pH. The frequency of observations in excess of pH 10 were higher in Lakewood and reflected the tendency of this pond to support a higher phytoplankton biomass. Mean and median pH and alkalinity values were similar for both ponds. The majority of pH values in all ponds were pH >9.2.

Table 3.3 Summary of detention pond pH and alkalinity values for the years 1992 to 1995. (alkalinity in mg $CaCO_3$/L)

		Rochdale** (n=75)	Lakewood (n=95)	Lakeview	
				top (n=64)	bottom(n=45)
range	pH alkalinity	8.4 - 10.3 26 - 139	8.4 - 10.7 10 - 132	8.1 - 9.9 32 - 192	7.9 - 9.0 110 - 210
mean	pH alkalinity	9.4 87	9.7 80	9.1 118	8.5 150
median	pH alkalinity	9.4 86	9.7 76	9.1 114	8.5 150
% obs pH 9.0 - 9.49		51	21	47	zero
% obs pH 9.5 - 9.99		32	52	12	zero
% obs >pH 10.0		4	18	zero	zero

** Rochdale data does not include period following 1995 aluminium sulphate addition

Alkalinity and pH differences between top and bottom water in the stratified Lakeview pond reflect processes of $CaCO_3$ precipitation in higher pH epilimnion water, and subsequent dissolution in the lower pH/lower temperature hypolimnetic water (*e.g.,* Köschel *et al.,* 1983; Lopez and Morgui, 1992). The inter-relationships reflect variable photosynthesis in the epilimnion, with degradation of sinking organic material and accumulation of reduced compounds and CO_2 in the hypolimnion. Periodic entrainment of higher alkalinity/lower pH bottom waters into the upper strata during stormflows were characteristic, and temperature profiles indicated that seiche mechanisms operate in Lakeview pond. The pH differences between the mean values of top and bottom samples were around 0.6 pH units, with frequent differences >1.0 pH unit. The mean top water pH of 9.1 was lower than those of both Rochdale and Lakewood ponds, reflecting generally lower photosynthetic biomass of this pond. Lakeview pond receives groundwater baseflow (approx. 0.0048 m^3/sec in the north inlet in 1995), with an average alkalinity of 510 mg/L $CaCO_3$ that, in addition to supplementing bottom water alkalinity, is probably responsible for this pond being slightly more alkaline than either Rochdale and Lakewood ponds (see Chapter 4 for sediment calcium comparisons).

3.2.5 Trends in nitrogen, phosphorus, and silica

Rochdale stormwater detention pond (Regina): Comparison of TP, DIP, and DIN shows that wide seasonal and annual variations existed (Figure 3.7 a,b). Total-P and DIP were sustained at the highest levels in 1992 when there was less than average precipitation. The extent of internal P loading can be clearly seen in the 1992 data, when concentrations of TP and DIP reached 0.61 and 0.35 mg/L, respectively. The remaining years sustained lower TP levels, in the range of 0.1 to 0.3 mg/L, for most of the monitoring periods. The very large increase in PP in August of 1992, and of TP and PP in August 1994, followed a series of moderate precipitation events that continued to supply DIN, but with only moderate biomass losses through flushing. Available DIP and high insolation resulted in massive particulate P increases (Figure 3.7 a).

Very low levels of DIN were characteristic (Figure 3.7 b). Periodic algal bloom collapses/reductions resulted in releases of NH_4, which subsequently allowed proliferation of replacement species and increases in particulate N and P fractions. A $CuSO_4$ application in August 1992 resulted in a pulse of NH_4 detectable by the following week. This was rapidly removed back into the particulate fraction and followed a week later by another NH_4 pulse, apparently due to the collapse of the $CuSO_4$ induced assemblage and a moderate precipitation event. External DIN loading tended to produce pulses of NO_3 rather than NH_4 within the pond. Although stormwater tends to be higher in NO_3, the effect may be more acute in part because NH_4 is preferentially taken up by the phytoplankton (Liao and Lean, 1978). The NO_3 peaks in Rochdale pond may be increased as a result of the agricultural runoff component previously described. Both NH_4 and NO_3 will be lost through coupled nitrification/denitrification, and *via* NH_3 volatilization processes at elevated pH and temperature (Bouldin *et al.*, 1974).

The overall comparisons showed that DIP accumulated in both dry and wet years, but higher levels resulted during extended dry weather periods. Accumulated DIP was periodically moved into the particulate fraction when short-term increases of DIN followed rainfall or algae bloom collapses. Examination of dissolved N:P ratios suggested that the system was usually N-limited, assuming a stoichiometric supply requirement of 7:1 (Figure 3.7 c). Uhlmann (1982) summarized literature values and gave $\leq$5:1 ratio (by wt) as the transition ratio indicating N-limitation. For the years 1992 to 1994, dissolved N:P ratios were <5 on 62 out of 67 monitoring days. However, frequent dominance of the phytoplankton by non-fixing blue-green algae (primarily *Oscillatoria* sp.) indicated that N must be rapidly recycled within the system. Total-N was measured in 1994 only, and the relationships between TN:TP and soluble N:P ratios and blue-green algal dominance are discussed in Chapter 6. The effects of the aluminium sulphate application are discussed in Chapter 7.

Silica was measured in 1992 and 1993 only (data not shown). In 1992, Si levels were high under ice cover (>9 mg Si/L) indicating dissolution processes. Si was depleted for about one to two weeks immediately after ice-melt (April), in concert with a brief diatom bloom. Thereafter, Si levels increased gradually to ~5 mg/L by July, as a result of mineralization of the spring diatom bloom. Silica limitation may have been a factor in the relatively early decline of the 1992 spring diatom bloom. However, the succession from diatoms to greens/blue-greens is also dependent on other factors such as temperature/light climate changes, and variable flushing influences (Bailey-Watts *et al.*, 1990; Reynolds, 1984). The pattern observed in 1993 was similar, although summer Si concentrations only reached about 2.5 mg/L. The winter Si pool was not measured in 1992/93, but more rainfall in 1993 may have been the cause of the lower summer values.

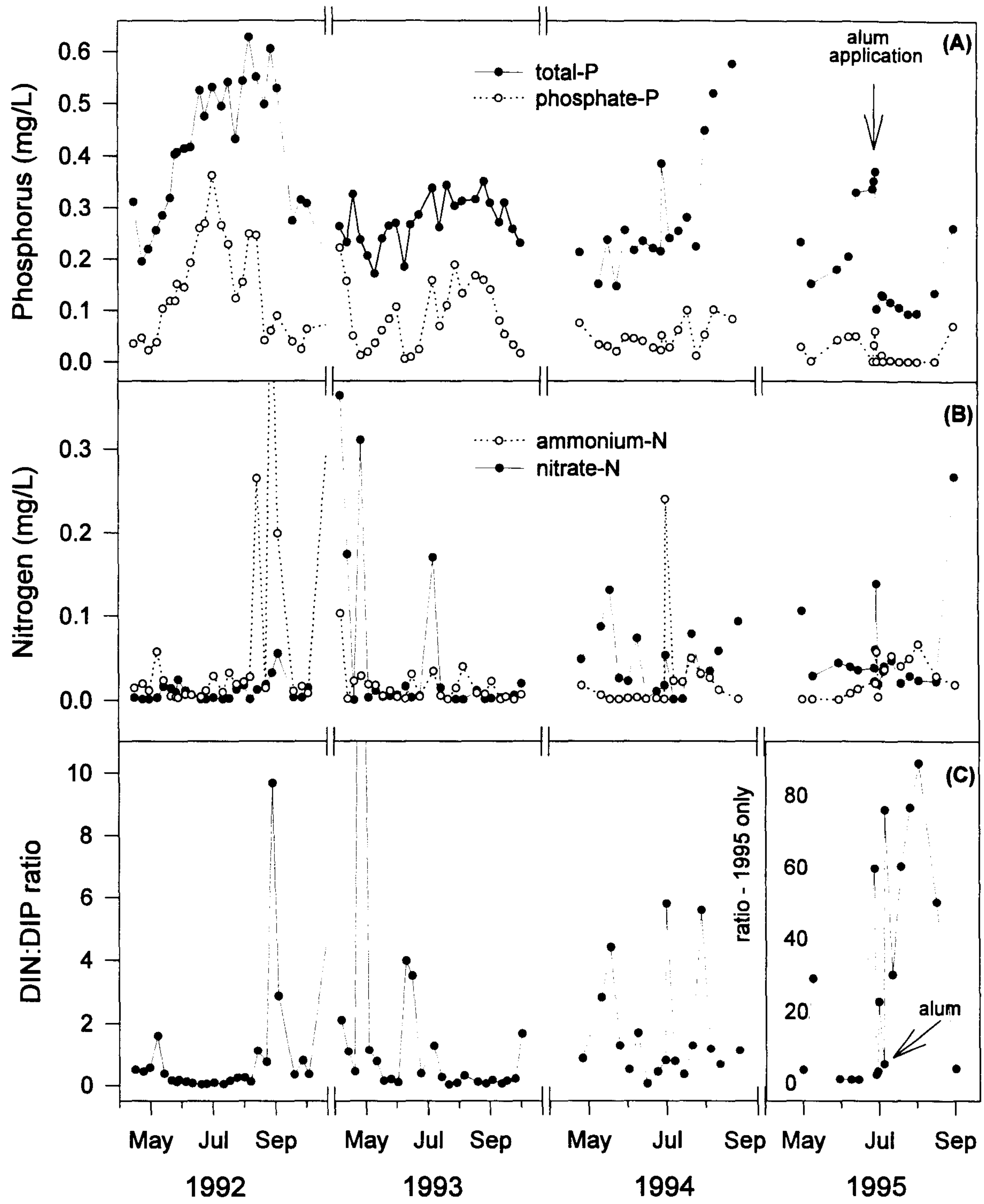

Figure 3.7 Total and dissolved inorganic phosphorus (A), dissolved inorganic nitrogen (B), and dissolved N:P ratios (C) in Rochdale pond (Regina), 1992 to 1995.

Lakewood stormwater detention pond (Regina): The long-term trend in Lakewood pond followed a very similar pattern to that described above for Rochdale pond. Peak mid-summer TP concentrations of 0.6 to 0.8 mg P/L were measured in all years. Dissolved inorganic phosphorus accumulations were highest in 1992 and 1993, and concentrations in excess of 0.2 mg/L were measured on 41 out of 48 monitoring days (Figure 3.8 a). The most sustained periods of high PP were during phytoplankton blooms in the latter part of both the 1994 and 1995 seasons, with maximum PP values of 0.470 mg/L in both years. The 1994 TP and PP increase was the same as that which took place in Rochdale pond at the time, and the transformations of DIP appeared to be facilitated by external loading of DIN (although Lakewood had been subject to experimental N additions between May and July of 1994). A large PP increase which occurred in Lakewood during July 1995 did not occur in Rochdale due to aluminium sulphate addition. The particulate increases were brought about by development of a dense N-fixing algal population, supported by high insolation, calm conditions, and internally loaded DIP.

Throughout the study, the trend was for very low DIN concentrations, with marginal increases measured after rain (Figure 3.8 b). Short-term DIN increases tended to be NH_4 and not NO_3, indicating a low external NO_3 load, and a dominance of internal mineralization processes to the DIN supply of this pond. Excluding 1994 data (experimental N additions), the open water DIN:DIP ratio was consistently very low - below 0.5 on 54 out of 61 monitoring days for the three seasons (Figure 3.8 c). These values are lower than those described above for the Rochdale pond during the same period. However, since DIP was very high, low N:P ratios may not necessarily reflect zero availability of N, and at high biomass levels, light may also become a co-limiting factor.

Inorganic nitrogen concentrations were lowest during the dry and hot summer of 1992. It was noted that although TP and DIP were very high, the PP fraction was relatively small in June. The PP fraction increased only once climatic conditions became suitable for N-fixing species to capitalize on low N conditions in July/August.

Silica levels were measured in 1992 and 1993 only (data not shown). In 1992, Spring concentrations of Si were never <1.0 mg/L, and concentrations began increasing from late May onwards. Peak summer concentrations (~4.0 mg/L) were slightly lower than those of Rochdale pond. The following year a similar pattern was observed, although summer concentrations were not as high (~2.5 mg Si/L). Si was therefore not assumed to limit the vernal diatom production in either 1992 or 1993.

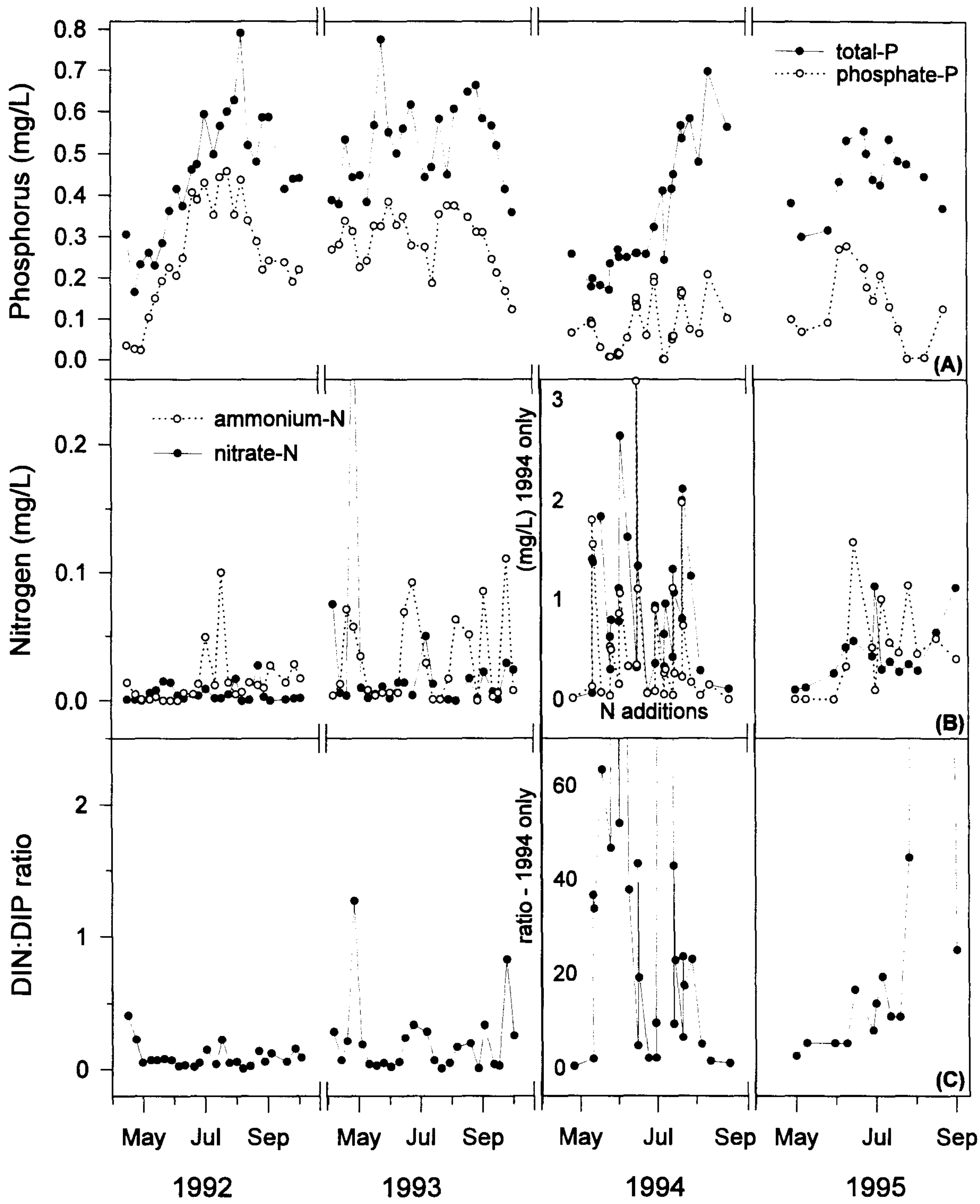

Figure 3.8 Total and dissolved inorganic phosphorus (A), dissolved inorganic nitrogen (B), and dissolved N:P ratios (C) in Lakewood pond (Regina), 1992 to 1995.

<u>Lakeview stormwater detention pond (Saskatoon):</u> Lakeview nutrient dynamics take on the added dimension of stratification and transfer between compartments. Total-P and DIP were almost always higher in bottom samples in 1993 to 1995 (no bottom samples were taken in 1992), due to the combination of seston mineralization and anaerobic P release from the sediment profile (Figure 3.9 a,b). Intermittent hypolimnetic DIP accumulation during dry weather was apparent in all years, especially 1993 and 1994. For the summer months of 1993 to 1995, bottom water DIP varied between 0.05 and 0.2 mg/L. June and July of 1995 had frequent rainfall and hypolimnetic DIP remained below 0.10 mg/L, presumably due to flushing and entrainment to the upper water (Figure 3.9 b).

The residual of P accumulated under winter ice cover was apparent on the first monitoring day of 1994, when maximum bottom water DIP was 0.35 mg/L. Dissolved inorganic phosphorus availability in the euphotic zone was variable, but was highest in spring before strong thermocline formation. Epilimnetic DIP was intermittently depleted to low levels by phytoplankton growth and perhaps to some extent by co-precipitation with $CaCO_3$ at high pH (Köschel *et al.*, 1983). Considering the proximity of hypolimnetic sources, recycling within the epilimnetic PP pool, and luxury uptake of P, it is unlikely that phytoplankton were P-limited for most of the study period, if an average half saturation constant (K_s) of ~10 µg DIP/L is assumed. However, values of K_s may vary considerably among species (Seip and Reynolds, 1995; Tilman *et al.,* 1982).

Within the TP:DIP regime, epilimnetic PP generally varied between 0.075 and 0.175 mg/L, which is similar to that measured in the shallower Regina ponds, despite their typically higher TP values. The 1993 data clearly showed that a greater proportion of the top water TP was in the particulate fraction for most of the open water period. The 1994 data showed less divergence between top and bottom PP values, and the 1995 data contrasted by having higher PP in bottom samples on most sampling dates (Figure 3.9 a,b,c). The dry summer of 1993 was more dominated by blue-greens compared to 1994 or 1995, implying buoyancy control and reduced sedimentation rates. More frequent rainfall in 1995, a lesser proportion of blue-green algae, and a slightly different pattern of zooplankton production may have been involved. The lower TP and PP values in the first half of 1992 were attributed to grazing by large-bodied cladoceran zooplankton.

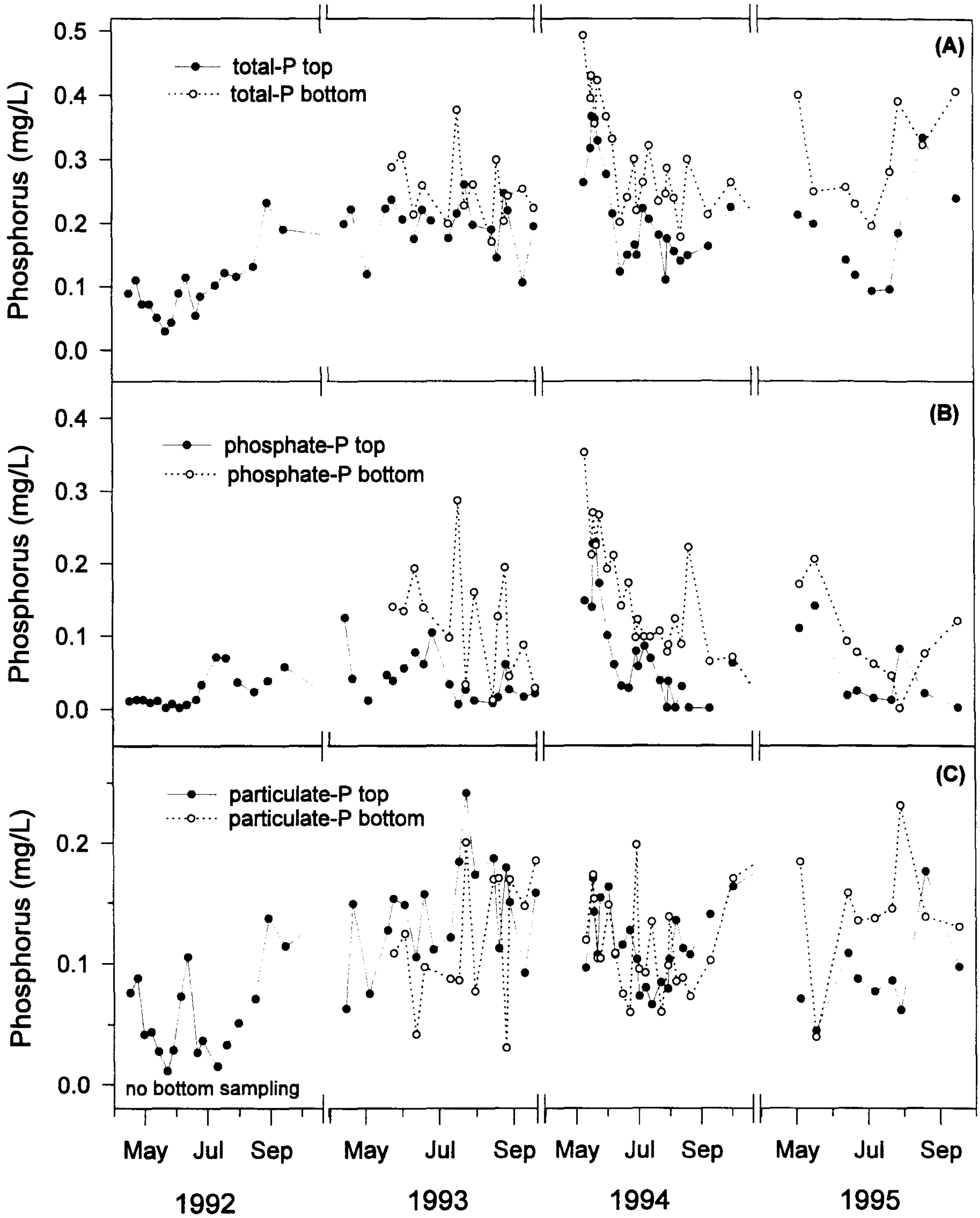

Figure 3.9 Total phosphorus (A), dissolved inorganic phosphorus (B), and particulate phosphorus (C) in top (0-1.5 m integrated) and bottom (2.6 m) waters of Lakeview pond (Saskatoon), 1992 to 1995.

Periodic equalization or simultaneous reductions of top and bottom P parameters reflected significant mixing/flushing events. Stormwater discharge volumes were measured in 1994 and 1995 in conjunction with measured and modelled nutrient loads (see Chapter 5). In-pond monitoring was more frequent during 1994 compared to 1995, so the short-term effects of some loading events were better described. An example of a mixing event which was picked up in the monitoring data can be seen in mid-May 1994, when an 9000 m^3 discharge occurred (approx. 20% exchange) (Figure 3.9 a,b,c).

Top and bottom water TP concentrations tended to follow each other in the pattern of peaks and troughs. Simultaneous declines of both top and bottom water TP most often resulted from stormwater flushing events in which a negative P retention occurred. In some instances, water column TP may be temporarily reduced, while the mass balance may actually produce a positive or neutral P retention coefficient on account of a high proportion of external TP being associated with particulates that settle out. The balance depends on the internal loading rate, the duration of the preceding inter-event dry period, and the stormwater inflow quantity and quality. Clearly, the P dynamics of the system are very unstable.

The preceding discussion accounts in part for the pattern of DIN fluctuations. Compartmentalization of both NO_3 and NH_4 was typical of all years that top and bottom samples were collected. Top water NO_3-N tended to vary between zero to 0.1 mg/L, with only a few occasions when peaks around 0.2 mg/L were measured (Figure 3.10 a). Bottom and top water equalization and transfers for NO_3 in the Lakeview pond system include the contributions of both stormwater loading and variable baseflow of groundwater. Monitoring in 1994 and 1995 showed that bottom samples were usually higher in NO_3, which must be attributed to groundwater, since low O_2 concentrations below the thermocline would reduce the potential for nitrification (Figure 3.10 a). However, nitrification of NH_4 may be significant within the metalimnion (Christofi *et al.,* 1980). Higher NO_3 levels in bottom water in 1994/1995 were assumed to result from increases in baseflow volumes from a higher water table and greater frequency of runoff events. Average baseflow NO_3-N concentration was 8.5 mg/L in 1994 (see Chapter 5).

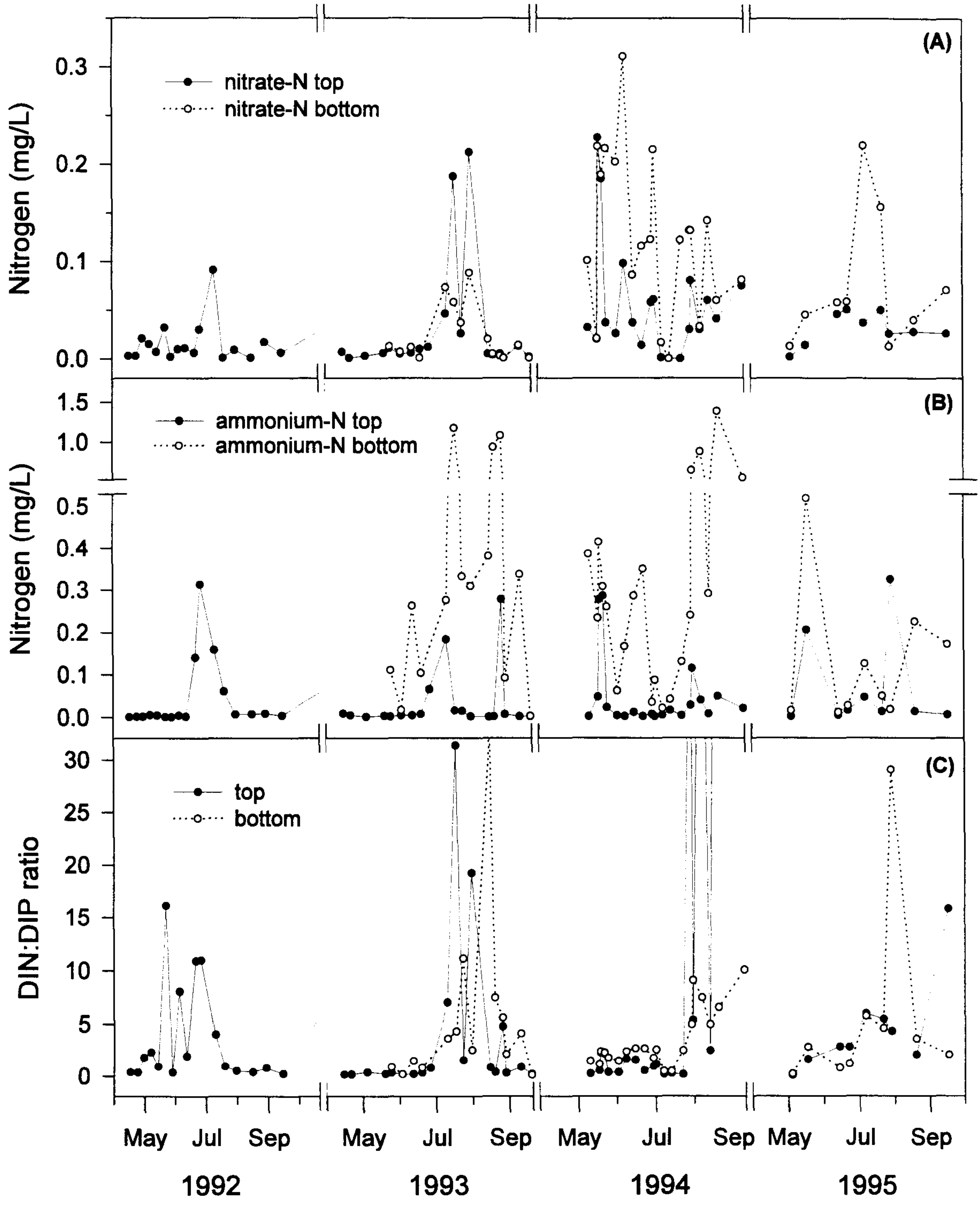

Figure 3.10 Nitrate (A), ammonium (B), and dissolved inorganic N:P ratios (C) in top (0-1.5 m integrated) and bottom (2.6 m) waters of Lakeview pond (Saskatoon), 1992 to 1995.

Periodic accumulations of NH_4 in bottom water were evident in all years, particularly in 1993 and 1994 (Figure 3.10 b). Maximum concentrations were in the range of 1.0 mg/L NH_4-N. Uptake of NH_4 by the phytoplankton tended to keep top water concentrations close to zero, resulting in a strong concentration gradient between top and bottom waters. As with DIP, periodic mixing and flushing events led to equalizations of top and bottom water concentrations, with rapid removal into the particulate fraction. Top water dissolved N:P ratios were ≤ 5 on 50 out of 65 monitoring days, suggesting a predominance of N-limitation, although greens comprised a large part of the phytoplankton community. The extent to which primary productivity may become severely N-limited was exemplified by the occurrence of a dense heterocystous *Anabaena* sp. bloom in mid-July of 1993. This bloom was sustained within close proximity to bottom water concentrations of 0.3 to 1.0 mg NH_4-N/L, indicating the extent to which transfer of hypolimnion nutrients may be limited. Although 1995 had relatively high rainfall, the pond was sampled less frequently than the previous three years, so that more short-term peaks in both nitrogen and phosphorus may have been missed.

Silica concentrations were measured in 1992 and 1993 only (data not shown). Silica levels reached ~4.0 mg Si/L under the ice cover in 1992/93. In April 1993, Si levels became depleted and remained <0.4 mg/L until mid-June, after which concentrations increased to 2.0 mg/L. Although no phytoplankton counts were obtained in 1992, the data suggested that diatom production was more extended in this pond compared to the two shallow ponds. In 1993, top water silica was reduced from 1.3 to 0.012 mg Si/L in the last two weeks of April, in concert with a spring diatom bloom. Diatoms often increased briefly throughout the season following inflows, which presumably acted to entrain higher Si bottom water and promote temporary suspension of cells. Concentrations of dissolved Si at these times indicated that subsequent declines of diatoms were not due to Si limitation (>0.05 mg Si/L). One low concentration was measured again in June, but depletion did not occur. Diatoms often increased in numbers for short periods after inflows, which presumably acted to entrain higher Si bottom water and promote suspension of cells.

3.2.6 Transparency, chlorophyll, and phytoplankton biomass

Rochdale stormwater detention pond (Regina): For the years 1992 to 1995, Secchi disk depths varied between 0.15 to 0.6 m, with most values around

0.3 m (Figure 3.11 a). The highest chlorophyll values were sustained in 1992 and 1993 and fluctuated between 0.1 and 0.4 mg/L Chl *a*. Chlorophyll values were lower for much of 1994, however, a peak to 0.36 mg/L Chl *a* occurred in early August. This sharp increase in Chl *a* was preceded by a number of runoff events, which were then followed by an extended period of hot sunny weather (as for nutrient changes previously described in Section 3.2.5). In 1995, chlorophyll was reduced to relatively low levels in mid-May following heavy rainfall, which exchanged close to 65% of the pond volume (Figure 3.11 b). Chlorophyll *a* levels then gradually increased to a seasonal peak of 0.3 mg/L by late June. The pond was then treated with aluminium sulphate which reduced algal biomass for the remainder of the season (see Chapter 7). For much of 1992 and 1993, chlorophyll *b* levels were relatively high with maxima reaching 0.3 and 0.2 mg/L, respectively. Following a $CuSO_4$ application in July of 1992, Chl *b* levels exceeded Chl *a*. On most sampling dates, Chl *b* fluctuated positively with Chl *a*. There was no clear pattern of Chl *a:b* ratio fluctuation with the proportion of blue-green algae. The general observation was that moderate blue-green algae dominance and biomass levels stimulated moderate to high Chl *b* synthesis in competing green algae and, either very high blue-green algae, or very high green algae dominance produced high Chl *a* and low Chl *b* levels.

Total cell volume varied between 6 to 65 mm^3/L for 1993 to 1995 (Figure 3.11 c). Spring phytoplankton populations were initially composed of diatoms (primarily *Cyclotella* sp. and *Fragilaria* sp.), small Chlorococcales, Euglenophytes, and Cryptophytes. In all years, these populations were replaced by *Oscillatoria* sp. with high blue-green algae dominance established by mid-May. For almost all of the 1993 season, *Oscillatoria tenuis* dominated the phytoplankton, accounting for 65 to 95% of the total biovolume. The 1994 season followed a similar pattern to 1993, until early July when *O. tenuis* gradually lost dominance to an emerging bloom of small lunate cells (<10 µm in length), tentatively identified as *Selenastrum minutum*. During the transition, *Cryptomonas ovata* briefly increased to about 20% of the biovolume. The green algae bloom persisted in dominance until the last sampling date in September, although, by that time the total volume was declining. *O. tenuis* dominated the phytoplankton from May until mid-June 1995, when a series of small precipitation events induced a variety of green algae (primarily *Chlamydomonas*, *Selenastrum*, and *Ankistrodesmus*). The pond was subsequently treated with aluminium sulphate (responses are discussed in Chapter 7).

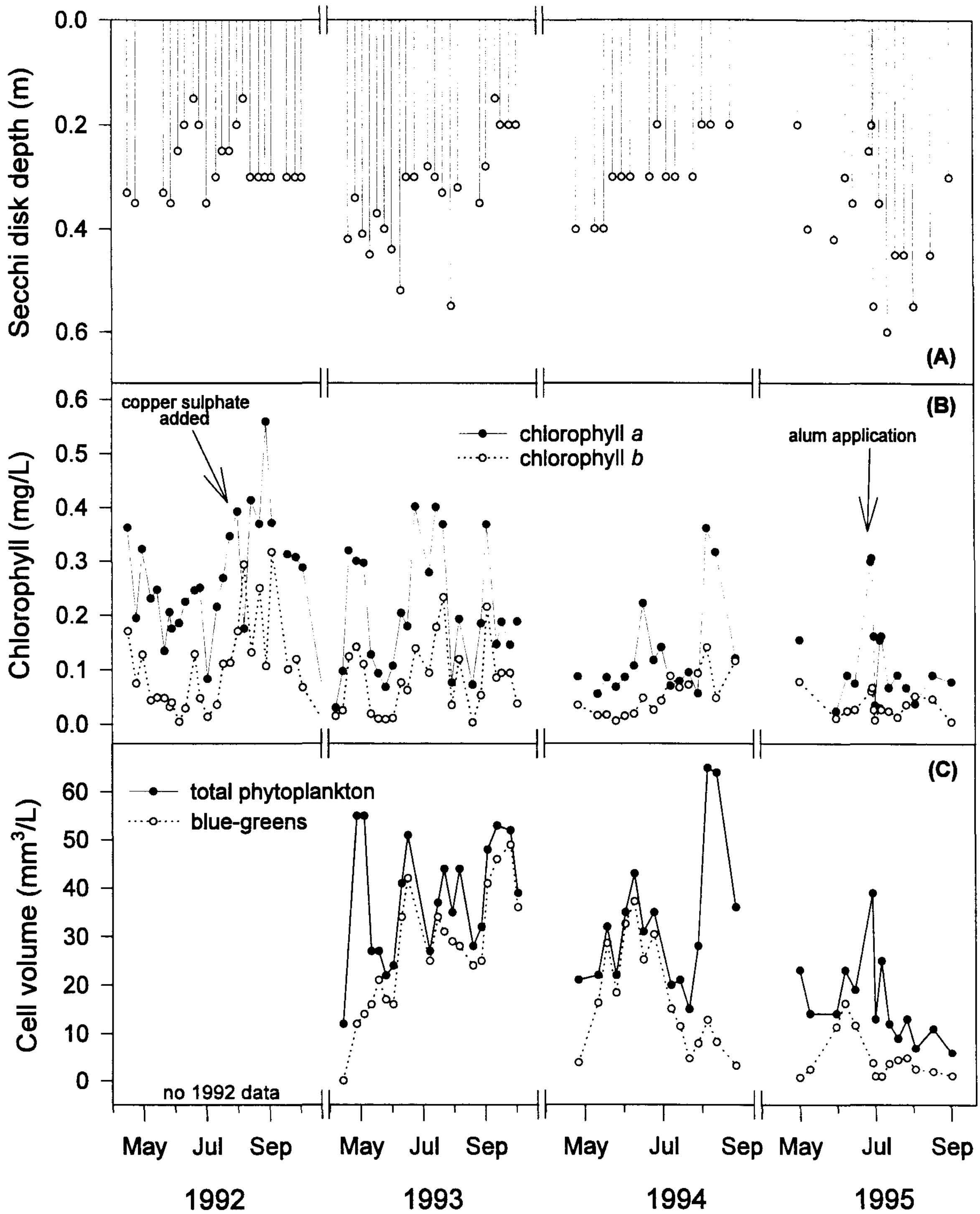

Figure 3.11 Secchi disk depth (A), chlorophyll (B), and algal cell volume (C) in Rochdale pond (Regina), 1992 to 1995.

Lakewood stormwater detention pond (Regina): Secchi disk depths in Lakewood pond tended to be highest in the spring, with a gradual reduction towards the latter part of the summer. Transparencies were within the same range as those of Rochdale pond, between 0.6 to 0.2 m, however, single sampling dates in the early spring of 1992 to 1994 had Secchi depths of 0.7 to 0.8 m. In 1995, minimum depths of only 0.1 m were recorded during a dense *Anabaena* sp. bloom in July (Figure 3.12 a,b,c). Compared to Rochdale pond, Lakewood sustained considerably less chlorophyll in 1992 at levels of 0.05 to 0.25 mg/L Chl *a*. Chlorophyll concentrations and temporal fluctuations in 1993 were similar to those of Rochdale, with three periods of chlorophyll maxima in May, July, and September of 0.3 to 0.6 mg/L Chl *a*. In 1994, Lakewood pond was subject to experimental N:P ratio manipulation - this data is discussed in Chapter 6. In June and July/August of 1995, two periods of increased chlorophyll were measured (0.3 to 0.42 mg/L, respectively) during dry weather periods. In each case, the declines followed stormflows. As in the manner described for Rochdale pond, chlorophyll *b* levels tended to positively follow Chl *a*. There were several occasions when Chl *b* exceeded Chl *a* concentrations. Very high Chl *b* levels occurred throughout July of 1994, indicating severe light limitation of greens during transition to blue-green bloom (see Chapter 6). In all cases, high Chl *b* was induced only during high biomass periods (Figure 3.12 b)

The early season phytoplankton were always dominated by diatoms, chlorophytes, and cryptophytes. The highest diatom biomass was measured in May of 1993, when centric species accounted for 73% of the biovolume (*Cyclotella* sp., ~ 6 to 8 µm diameter) (Figure 3.12 c). Filamentous blue-green dominance did not develop until mid-June in 1993, which was several weeks later than for Rochdale. Initially, the blue-green algal biomass was composed of a mixture of *Oscillatoria hamelli* (≈ 2.5 µm diameter, non-rigid trichome, identified as depicted by Prescott, 1962), *O. tenuis,* and non-heterocystous *Anabaena spiroides*. Throughout July of 1993, *O. tenuis* dominated, but from August onward, *Anabaena spiroides* with heavy heterocyst development were dominant. However, at the time of the highest blue-green algal biomass, it was noted that 20 to 30% of the total cell volume comprised a variable mixture of green algae including *Cryptomonas ovata, Trachelemonas* sp., *Scenedesmus* sp., *Peredinium* sp., *Golenkina radiata*, and *Pandorina* sp. These species appeared to be sustained by the N derived from that being fixed into the system by *Anabaena.* A very similar pattern of biomass volume and fluctuation in the species composition was observed in 1995. The blue-green algae bloom in August of 1994 was composed of non-fixing colonial species, primarily *Microcystis aeruginosa*, *Coelosphaerium* sp., and *Aphanocapsca* sp. (Figure 3.12 c).

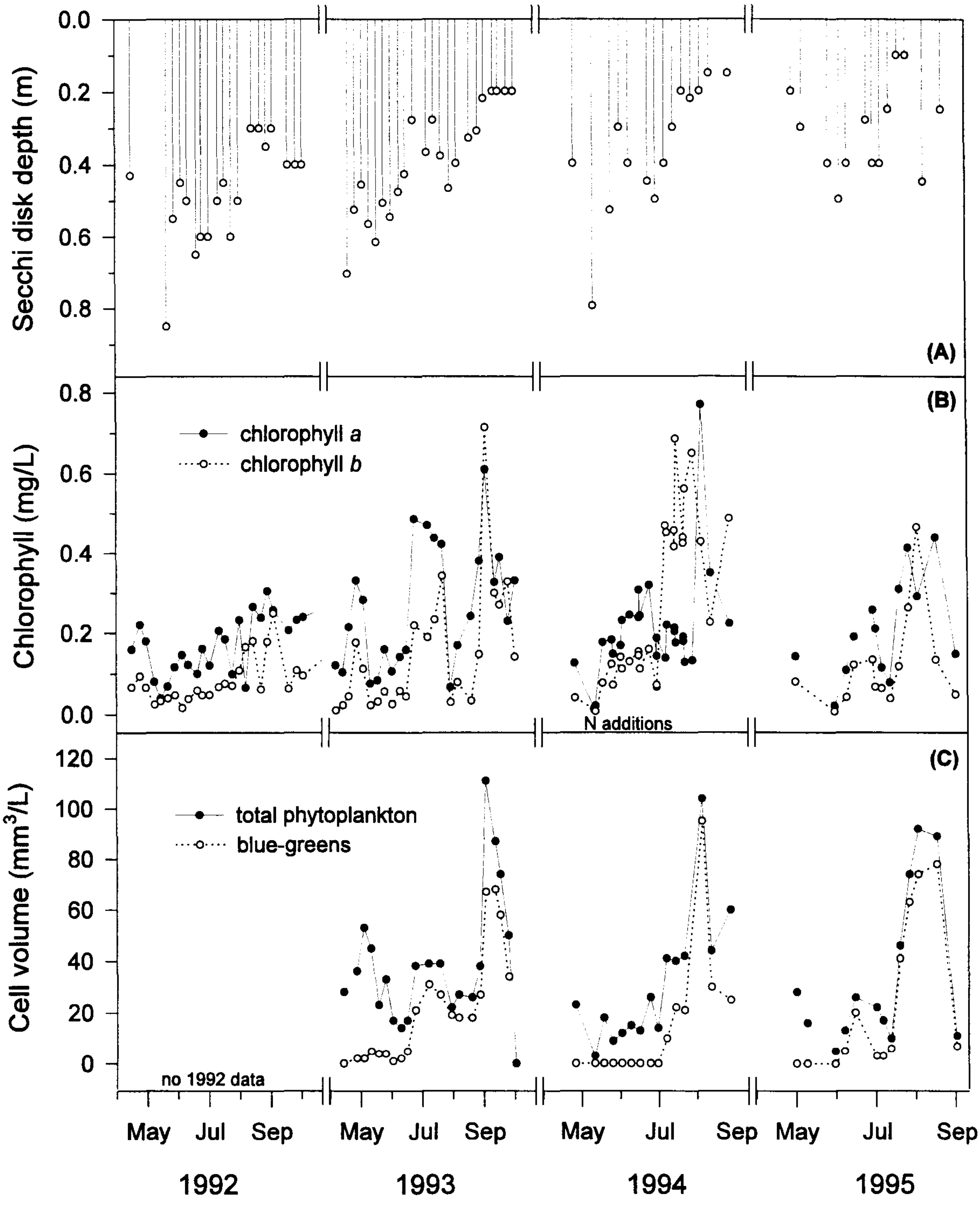

Figure 3.12 Secchi disk depth (A), chlorophyll (B), and phytoplankton cell volume (C) in Lakewood pond (Regina), 1992 to 1995.

Lakeview stormwater detention pond (Saskatoon): Maximum Secchi disk depths of 1.3 to 2.2 m were measured in July 1992, when a clear water phase was induced. This appeared to be brought about by a *Daphnia pulex* bloom. Minimum Secchi disk depths of 0.2 m were measured in 1993 to 1995 during peak algal biomass periods. Average Secchi disk depths were in the range of 0.5 m, which was greater than those recorded in the shallower Regina ponds (Figure 3.13 a). With the exception of the brief clear water phase in 1992 and four sampling dates in July/August 1993 (max Chl *a* = 0.45 mg/L; max Chl *b* = 1.0 mg/L), chlorophyll levels fluctuated within the range 0.02 to 0.22 mg/L Chl *a*, which gave a lower average than those measured in the Rochdale and Lakewood ponds. For the four year period, Chl *a* was $\leq$0.10 mg/L on 41 out of 66 monitoring days. The most extreme Chl *b* concentration of 1.0 mg/L in July 1993 was associated with an exceptionally high blue-green bloom, but still with a large green algal component (Figure 3.13 b,c). This pigment concentration was above what is theoretically sustainable in a mixed layer, due to self shading and respiratory losses (Bannister, 1974 a). Accordingly, the high biomass was not sustained.

In April 1993, a large biomass of *Cryptomonas erosa* accounted for 82% of the cell volume, with the remainder being *Cyclotella* sp. Throughout May and June, variable numbers of *Cryptomonas erosa*, *Chlamydomonas* sp., *Trachelomonas* sp., *Phacus* sp., *Elakothrix gelatinosa, Actinastrum hantzschii*, and a few *Oscillatoria hamelli* (possibly *O. limnetica*) filaments comprised the biomass. By early July, total biomass was increasing and a bloom of *Anabaena flos-aquae* was developing, but a significant volume of *Chlamydomonas* cells were also present (~8 μm diameter). This bloom peaked in the last week of July with total cell volume calculated to 120 mm^3/L with 67% *Anabaena*, 25% *Chlamydomonas*, and 5% *Cryptomonas*. Chlorophytes and diatoms resumed dominance from mid-August until the last sampling day in October 1993 (Figure 3.13 c).

The species composition from May to June 1994 was similar to that of 1993, being composed of variable Chlorophytes and Cryptophytes, and a few blue-green filaments. By early June, *Anabaena* sp. briefly increased to dominate the blue-green volume and by mid-June, *Oscillatoria hamelli* had been replaced by *Oscillatoria tenuis,* which was a typical succession. Throughout July, greens never accounted for less than 65% of the cell volume. *Oscillatoria tenuis* (predominant) and *Anabaena spiroides* increased from mid-August onward, but only accounted for 40 to 60% of the biomass. The 1995 cell volume and composition followed the same general pattern, except blue-greens were never more than 50% of the cell volume. In August of 1995, a brief increase in numbers of *Cyclotella*, *Chlamydomonas*, and *Cryptomonas* occurred, in addition to some *Oscillatoria tenuis* (Figure 3.13 c).

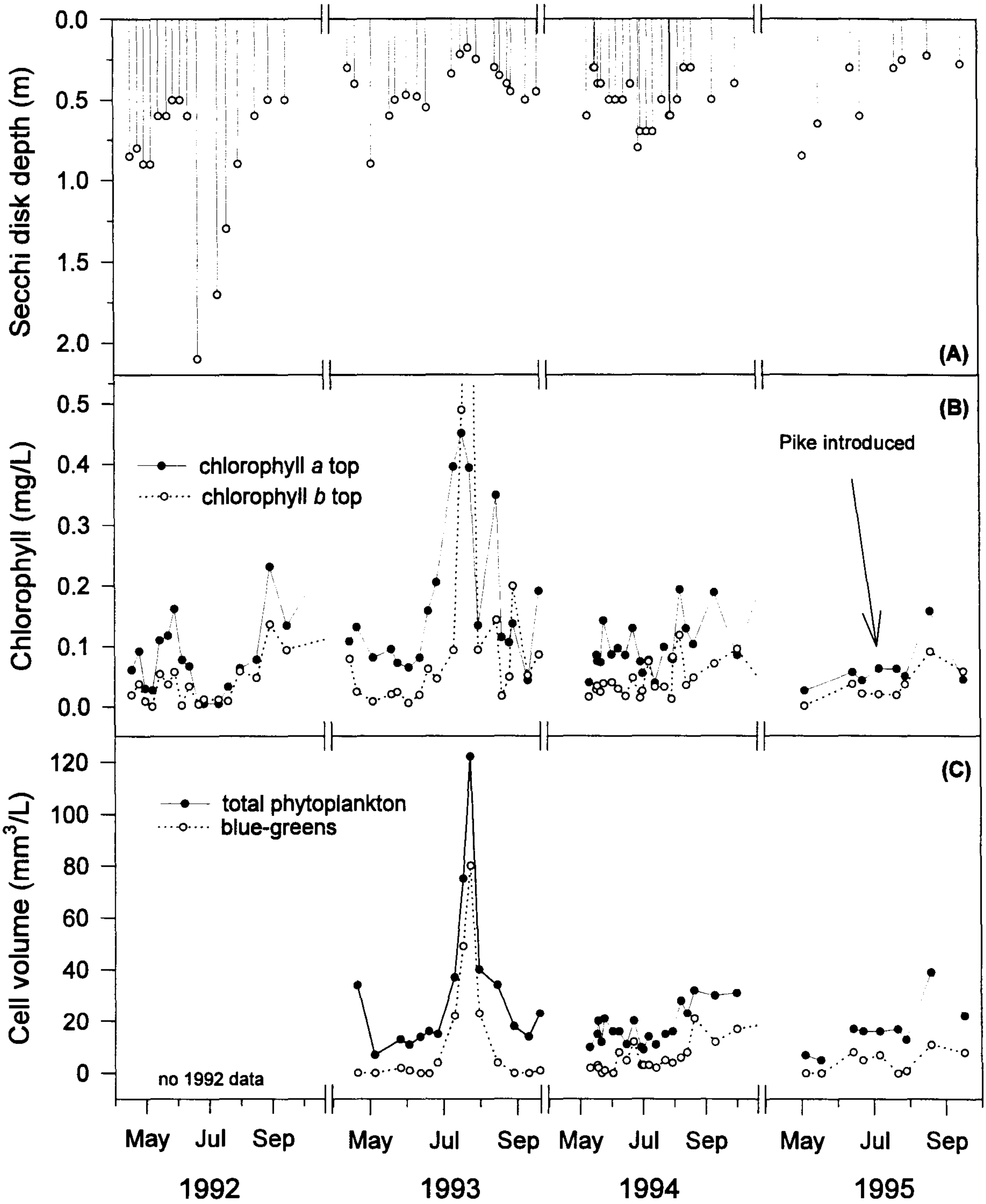

Figure 3.13 Secchi disk depth (A), chlorophyll (B), and phytoplankton cell volume (C) in Lakeview pond (Saskatoon), 1992 to 1995.

3.2.7 Zooplankton

Rochdale stormwater detention pond (Regina): From 1993 to 1995 the species composition was composed primarily of cyclopoid copepods (mostly *Mesocyclops edax*), with some calanoid copepods (*Leptodiaptomus siciloides*), and nauplii. The few cladocerans that showed up from time to time were *Bosmina longirostris* and *Daphnia parvula*, but they never dominated the total numbers. The trend in all years was for a late spring to mid-summer peak, with a secondary and smaller late summer increase. Maximum numbers were similar in all three years (Figure 3.14). Comparison of Figure 3.14 with the phytoplankton graph in Figure 3.11, shows that peak numbers of zooplankton tended to coincide with peak algal biomass. The findings suggest that algal grazing by zooplankton is of minor importance. Early spring numbers were always low, even with high diatom/green algal dominance. Temperature appeared to play a role in promoting the first annual peak. Rather than direct grazing, there appeared to be an intermediate food source involved in the synchrony of phytoplankton and zooplankton. This may involve detritus associated with increased phytoplankton turnover, bacteria, protozoa, or rotifer dynamics. The cyclopoid copepods may also be primarily predatory, and not herbivorous (Brandl and Fernando, 1975).

Lakewood stormwater detention pond (Regina): Similar to that described for Rochdale, the general cyclical pattern of zooplankton production in Lakewood pond was dominated by small copepods. In this pond, cyclopoid copepods included both *Ancanthocyclops vernalis*, and *Mesocyclops edax*. A slight divergence in the production pattern was seen during the period May to July of 1994 when the pond was manipulated with inorganic nitrogen additions. A prolonged first peak was extended into a closely associated second peak without the typical mid-summer decline. In 1995, the pattern was again slightly different, with the late summer peak larger than the spring peak. Both were associated with peak algal volume, in this case dominated by *Anabaena spiroides,* but still accompanied by a variable biomass of small greens (Figure 3.15). As in Rochdale pond, the effect of zooplankton on algal standing crop in Lakewood was indicated to be minimal and large-bodied cladocerans were completely absent.

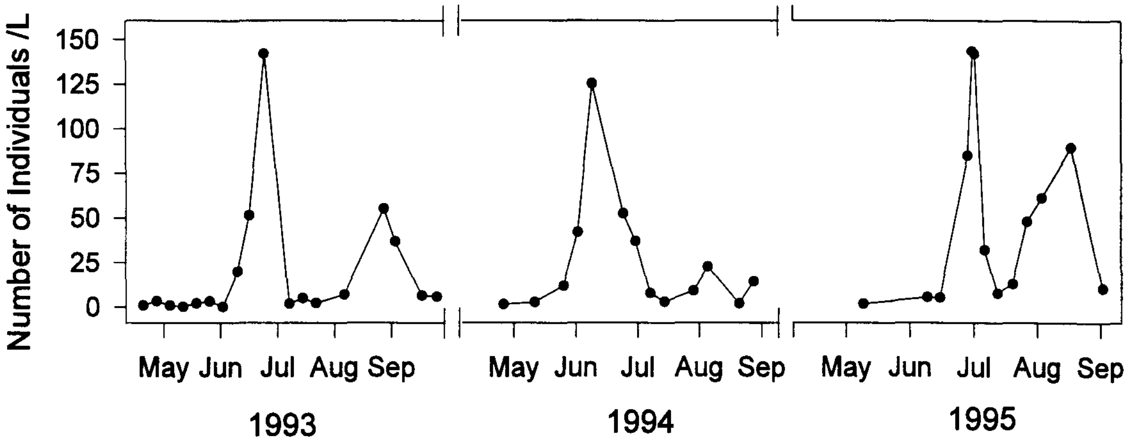

Figure 3.14 Total zooplankton numbers in Rochdale pond, 1993 to 1995.

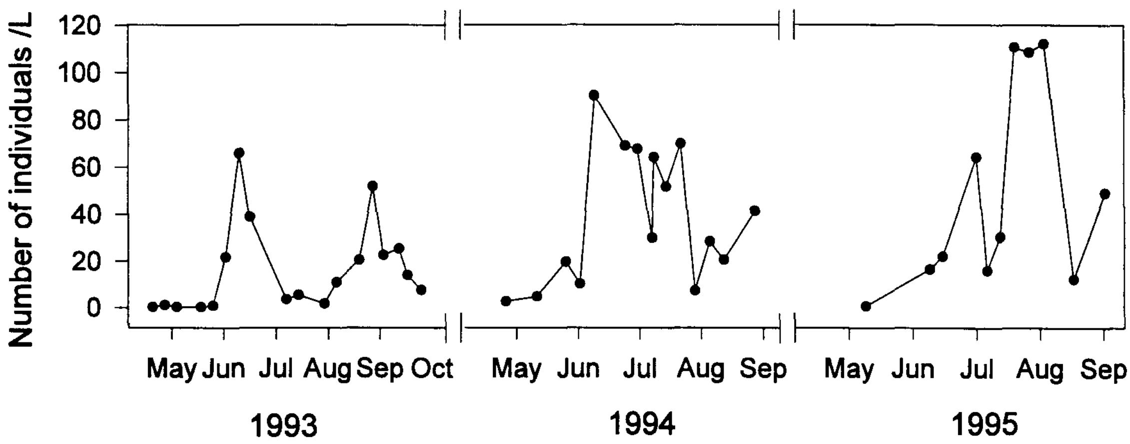

Figure 3.15 Total zooplankton numbers in Lakewood pond, 1993 to 1995.

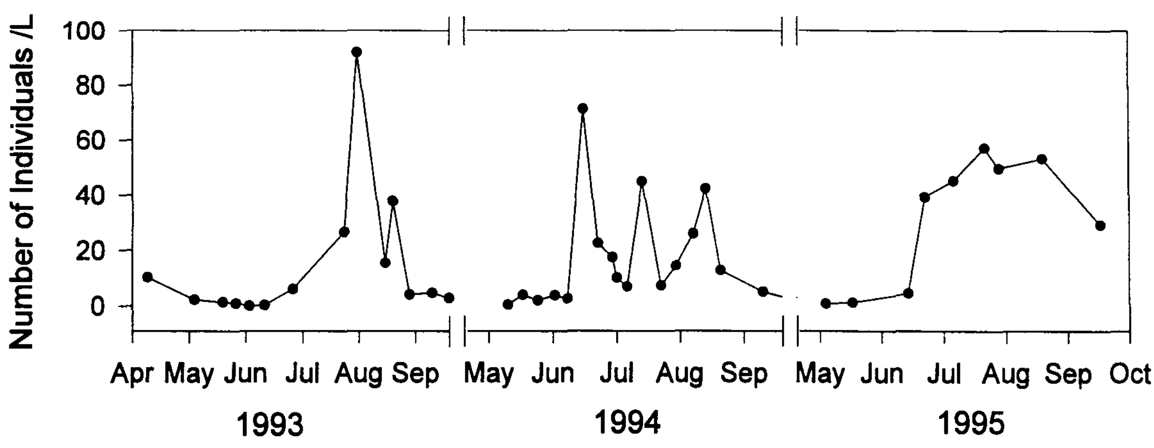

Figure 3.16 Total zooplankton numbers in Lakeview pond, 1993 to 1995.

Lakeview stormwater detention pond (Saskatoon): The species composition in Lakeview pond was generally similar to that of the other ponds, and comprised the same species in each year. Cyclopoid copepods (almost entirely *Ancanthocyclops vernalis*) and their nauplii dominated the community throughout 1993 to 1995. In this pond, the calanoid copepods were *Leptodiaptomus sicilis*. A few large-bodied cladocerans (*Daphnia pulex)* were recorded from time to time, so the potential to increase their numbers does exist. The pattern of seasonal peaks was slightly different between the years. In 1993, zooplankton numbers remained very low until mid-summer; 1994 had three peaks in numbers; while in 1995, a fairly steady population was sustained from June onwards (Figure 3.16). However, since the pond was sampled less frequently in 1995, periodic declines may have occurred. It was notable that the 1995 phytoplankton density was lower throughout the period when the zooplankton were sustained, but whether this was due to the dominating copepods was not clear. During the summer of 1995 (July), juvenile Pike (*Esox lucius*) were stocked into the pond, which may have affected minnows sufficiently to allow the zooplankton to maintain their density. Minnow surveys were carried out in 1996 and early indications were that the Pike had a significant effect on the minnow density.

3.2.8 Selected data relationships and data reduction

A number of linear regressions were performed on selected data sets of various parameters to examine inter-relationships, predictive ability, and methodological accuracy. Where appropriate, data sets were examined as grouped data from the three ponds for all years, or as individual data sets for individual years.

Chlorophyll, Secchi disk depths, and light attenuation: The relationships between chlorophyll and Secchi depth were examined to assess factors influencing primary productivity in the ponds. Light attenuation was estimated according to the Lambert-Bougier Law (Equation 3.1), where z = Secchi depth, I_o = Incident light, I_z = light at depth z, and K_w and α are the extinction coefficients for water and chlorophyll, respectively (Megard *et al.*, 1980).

Equation 3.1

$$z = \frac{\ln(I_o \;/\; I_z)}{k_w + \alpha C}$$

Data sets were linearized by plotting chlorophyll against the reciprocal of the Secchi depth. Regression of the complete data set for the three ponds gave a very poor correlation, a fact attributed to the shallow Regina ponds tending to sustain high algal biomass with consistently low Secchi disk depths, so that even significant fluctuations in chlorophyll produce little change in Secchi depth. Furthermore, Secchi disk and suspended solids *vs.* Chl *a* regressions indicated that the shallower detention ponds are influenced to a higher degree by detrital turbidity (see below). Predominantly low Secchi depths led to a scattering of data points at the high end of the scale (reciprocal of Secchi values) with a lack of adequate spread for significant regression (data not shown).

Lakeview had the greatest range of Secchi depths in 1992 and 1993, while none were greater than the maximum pond depth. Regression of 1992 data was significant: Chl *a* = $0.105(Z^{-1}) - 0.059$; $R^2 = 0.76$, $P=0.005$) (Figure 3.17). Secchi depth was assumed to represent 20% of the immediate sub-surface light intensity (Lorenzen, 1980) and from the intercept and slope average non-algal attenuation coefficient (k_w) and chlorophyll attenuation coefficient (α) calculated to 1.24 m^{-1} and 0.016 m^{-1} μg^{-1} Chl *a* L^{-1}. The calculated chlorophyll coefficient is close to frequently cited coefficients of 0.02 to 0.025 m^{-1} μg^{-1} Chl *a* L^{-1} (Ambrose *et al.,* 1993; Forsberg and Shapiro, 1982; Megard, 1980). Therefore, the proportion of light attenuated by phytoplankton would exceed background attenuation at Chl *a* >~80 µg/L, which was frequently the case during the study period. However, increased variance of regressions in other years and data from the shallower Regina ponds suggests background attenuation may vary considerably in these systems, whether by fluctuation of detrital/non-algal particulate or dissolved substances (Verduin, 1982).

<u>Chlorophyll, suspended solids, cell volume, and particulate carbon</u>:

Regression of chlorophyll on suspended solids (suspended solids were measured in 1994 only) was examined to assess the contribution of detritus in completely mixed *vs.* stratified ponds. The most relevant regression equations and correlation values discussed below are summarized in Table 3.4. In comparison to the stratifying Lakeview pond, Rochdale pond had higher suspended solids per unit Chl *a* in the 0.05 to 0.10 mg/L range, although the overall slope of the regression was steeper than that of Lakeview data as a result of several high Chl *a* concentrations (Figure 3.18). Summing of chlorophylls *a+b* did not significantly improve these correlations.

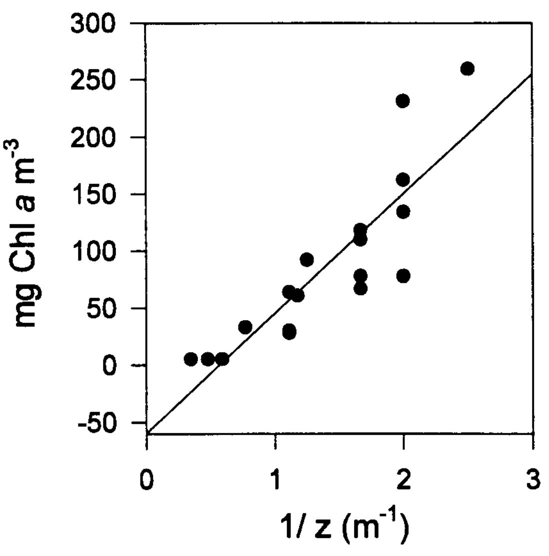

Figure 3.17 Regression of chlorophyll *a* *vs.* Secchi disk depth in Lakeview pond (Saskatoon), 1992.

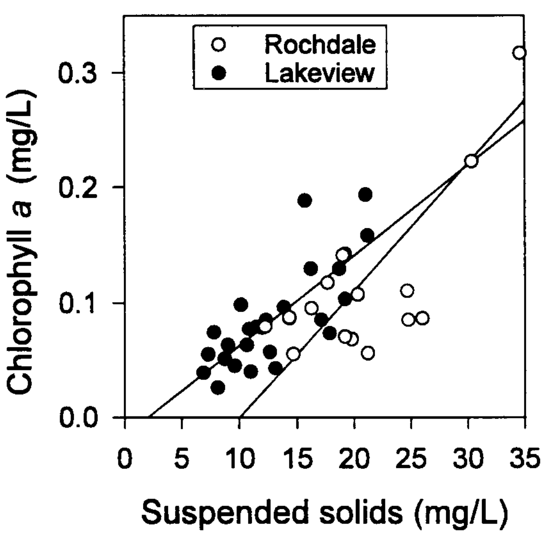

Figure 3.18
Regression of chlorophyll *a* and suspended solids in Rochdale pond (non-stratifying), and Lakeview pond (stratifying), 1994.

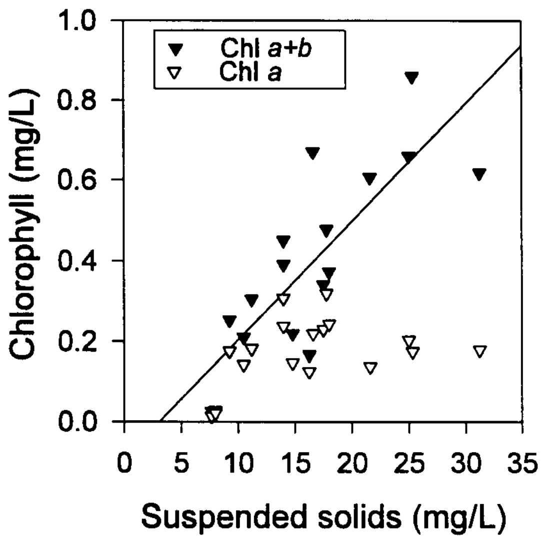

Figure 3.19
Regression of chlorophyll *a* and *a+b* with suspended solids in Lakewood pond (non-stratifying), 1994.

The correlation of Chl *a* and suspended solids for Lakewood pond was not significant, although the summing of chlorophylls *a+b* gave an improved correlation (Figure 3.19). The high variance of the Lakewood pond data, together with the spread and higher suspended solids residual (lower intercept) in the Rochdale regression, suggest a higher detrital component is maintained in the water column of the shallow ponds.

Regressions of Chl *a vs.* PC gave only weak correlations for all data sets, a fact attributed to the chlorophyll adaption mechanisms, and variable detrital components among the seston. Summing of chlorophylls *a+b* did not significantly improve these correlations (Table 3.4). Measurement of phycocyanin and phaeophytins may have provided some further insight into these discrepancies. However, PC did correlate better with suspended solids (Table 3.4).

Significant correlation of cell volume with suspended solids indicated that the counting and calculation technique was reasonably good, although some non-viable biomass may be included. Cell volume regressions on Chl *a* for both Lakeview and Lakewood data sets were improved by the summing of chlorophylls *a* and *b* as the pigment value (R^2 = 0.60 → 0.87, and 0.57 → 0.70, respectively - Rochdale R^2 was not improved). The improvement is attributed to the fact that both these ponds periodically sustained high Chl *b* levels in 1993 to 1995 (especially Lakewood in 1994). The relationships of cell volume on PC showed greater variance in the shallow ponds compared to Lakeview, which is presumably due to greater sinking losses of detritus in the stratified pond.

Table 3.4 Regression summaries of chlorophyll, carbon, phosphorus, suspended solids, and cell volume relationships.

Regression	Data Set	R^2 Value	Linear Equation
Chlorophyll *a vs.* suspended solids	(1994)		
	Rochdale	0.64	Chl *a* = 0.011(ss) - 0.113
	Lakeview	0.61	Chl *a* = 0.0078(ss) - 0.016
a	Lakewood	0.12	------------------------------
a+b	Lakewood	0.66	Chl *a+b* = 0.029(ss) - 0.09
Chl *a vs.* PC	all data (93-95)	0.38	------------------------------
Chl *a+b vs.* PC		0.42	------------------------------
PC *vs.* suspended solids	all data (1994)	0.72	PC = 0.362(ss) - 0.26
Biovolume *vs.* PC	Lakeview (93-95)	0.77	BV = 6.97PC - 13.8
	Rochdale (93-95)	0.57	------------------------------
	Lakewood (93-95)	0.50	------------------------------
	all data (93-95)	0.58	BV = 3.75PC +1.41
Biovolume *vs.* Chl	all data (93-95)		
a		0.57	Biovol = 126(Chl *a*) + 8.7
a+b		0.69	Biovol = 72.4(Chl *a+b*) + 10.4
Biovolume *vs.* suspended solids	all data (1994)	0.65	Biovol = 1.4(ss) - 1.17

Particulate carbon, nitrogen, and phosphorus ratios and phytoplankton: Variation of the cellular C:N:P ratios has consequences for nutrient uptake kinetics and growth of phytoplankton. Modelling of algal growth may require specification of population stoichiometry in units of nutrient uptake/mass of biomass synthesized (Ambrose *et al.*, 1993). Statistical summaries of the PC:PN:PP ratios (by wt) are given Table 3.5. The results show that different years within the same ponds produced varied means and median values for the ratios.

Table 3.5 Particulate C:N:P ratios measured in Rochdale, Lakewood, and Lakeview stormwater detention ponds, 1993 to 1995.

		Rochdale (No. obs)	Lakewood (No. obs)	Lakeview (No. obs)
1993	mean median	54 : 9 : 1 (21) 55 : 9 : 1	44 : 7 : 1 (22) 40 : 7 : 1	40 : 6 : 1 (19) 33 : 6 : 1
1994	mean median	38 : 6 : 1 (17) 37 : 6 : 1	34 : 6 : 1 * (22) 32 : 6 : 1	44 : 8 : 1 (20) 39 : 8 : 1
1995	mean median	42 : 7 : 1 (15) 42 : 6 : 1	37 : 7 : 1 (13) 35 : 7 : 1	50 : 8 : 1 (9) 51 : 8 : 1

* see Chapter 6

Divergence of C:N:P ratios from the "harmonious" 42:7:1 Redfield weight ratio (Vollenweider, 1985) appeared to be related to the phytoplankton composition in all ponds. The entire data set was pooled and analysed in relation to C:N, N:P, C:P and percent blue-green biomass expressed as 25% increments (Figure 3.20 a,b,c). Median values of C:N decreased significantly between the ≤25% blue-green band and higher bands (Mann-Whitney Rank Sum test, $P = 0.005$). Similarly, both the N:P and C:P ratios increased significantly ($P < 0.0001$; $P = 0.0009$) as blue-greens increased above 25% of the total cell volume. However, consecutive statistical differences between each of the upper three bands (>25% blue-green) were only significant for N:P, where each of the >25% to >75% bands gave $P<0.05$. A summary of ratio means for the highest and lowest blue-green percentages are given in Table 3.6.

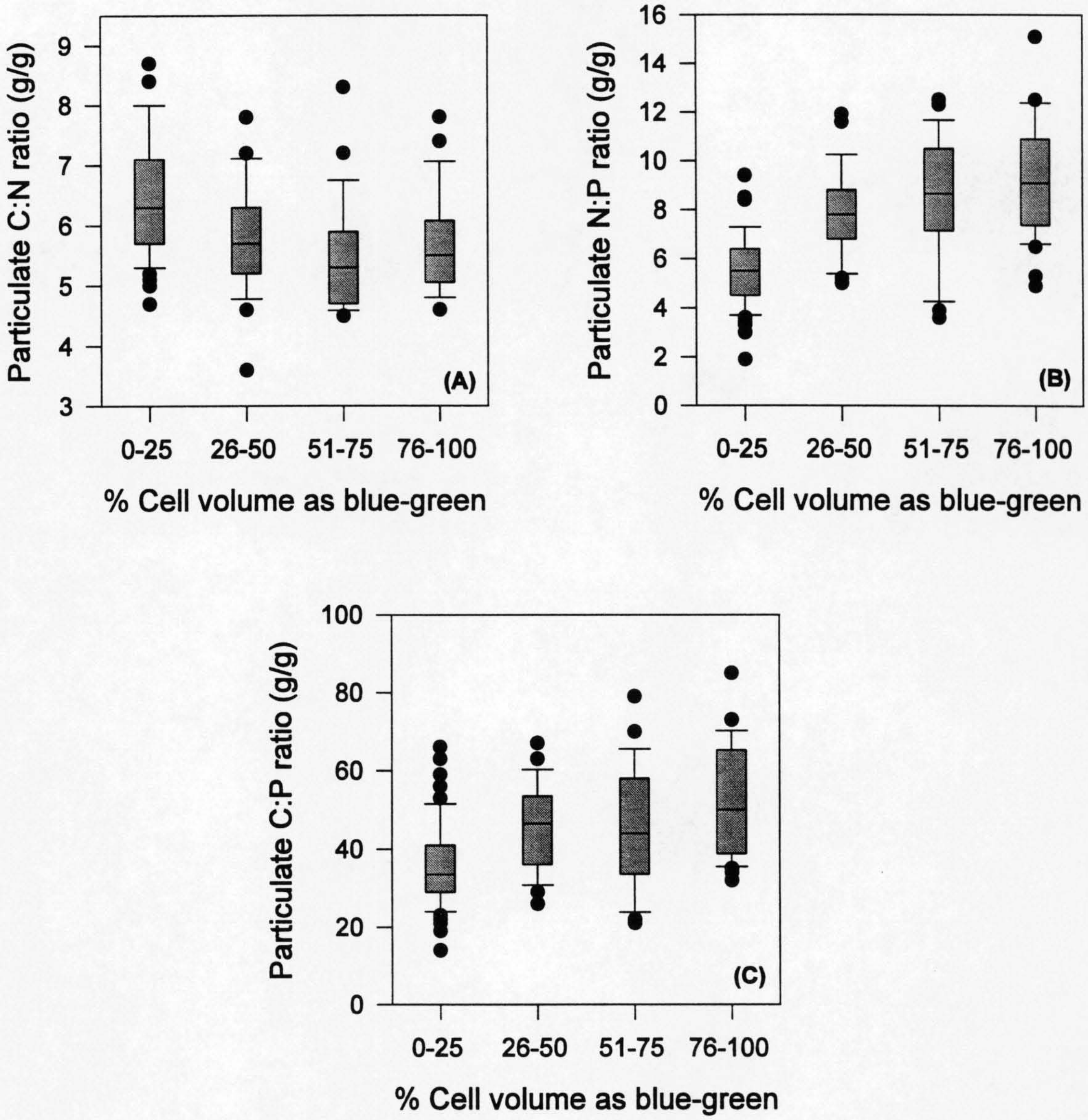

Figure 3.20 Box plots of seston carbon:nitrogen (A), nitrogen:phosphorus (B), and carbon:phosphorus (C), in relation to the relative dominance of blue-green algae. (Horizontal box lines are median values; bars represent the 95th percentile. Data are pooled from all detention ponds; N=136.)

Table 3.6 Total seston elemental ratios of carbon, nitrogen, phosphorus, and chlorophyll measured in stormwater detention pond summer samples from 1993 to 1995. (± 95% confidence intervals)

Data Set (% of cell vol.)		No. Obs.	C:N (mg/mg)	N:P (mg/mg)	C:P (mg/mg)	Chl *a*:C (µg/mg)	Chl *a*+*b*:C (µg/mg)
75-100%	range	59	4.7 - 10.9	3.0 - 9.4	19 - 66	6 - 65	7 - 113
green/diatom	mean		6.6 ± 0.3	5.6 ± 0.4	36.4 ± 3.0	25 ± 4	37 ± 6
75-100% total	range	29	4.6 - 7.8	4.9 - 15.1	33 - 85	5 - 45	
blue-greens	mean		5.7 ± 0.3	9.3 ± 0.9	52.0 ± 5.5	18 ± 4	
70-100%	range	11	4.5 - 5.7	7.1 - 12.5	34.0 - 65.0	14 - 47	
Anabaena spp.	mean		5.0 ± 0.3	9.9 ± 1.2	48.9 ± 7.5	25 ± 6	
total data set	range	136	3.6 - 10.9	3.0 - 15.1	19.0 - 85.0	5 - 99	7 - 129
	mean		6.0 ± 0.2	7.3 ± 0.4	43.1 ± 2.4	23.5 ± 2.2	36.6 ± 3.7

Dissolved inorganic carbon and phytoplankton species composition:

An examination of the relationship of increasing blue-green biovolume was made in relation to dissolved inorganic carbon availability [DIC]. Based on 25% delineations as given in the previous section, both total DIC and free CO_2 were reduced with increasing proportion of blue-green volume (Figure 3.21 a,b). Normality and variance permitted parametric t-test comparison for DIC. There was no significant difference between 0 to 25% and 26 to 50%, but increasing significance between 26 to 50% *vs.* >50% and >75% blue-greens (P=0.04 ⇒ 0.002). Free CO_2 data group distributions were skewed (except >75%) and medians were all low. Only the difference between the <25% and >75% groups were significant (P<0.001, Mann-Whitney Rank Sum test). Despite lack of consecutive significance, the increased range of the upper 50th percentile in the <25% blue-green group was notable in comparison to other groups.

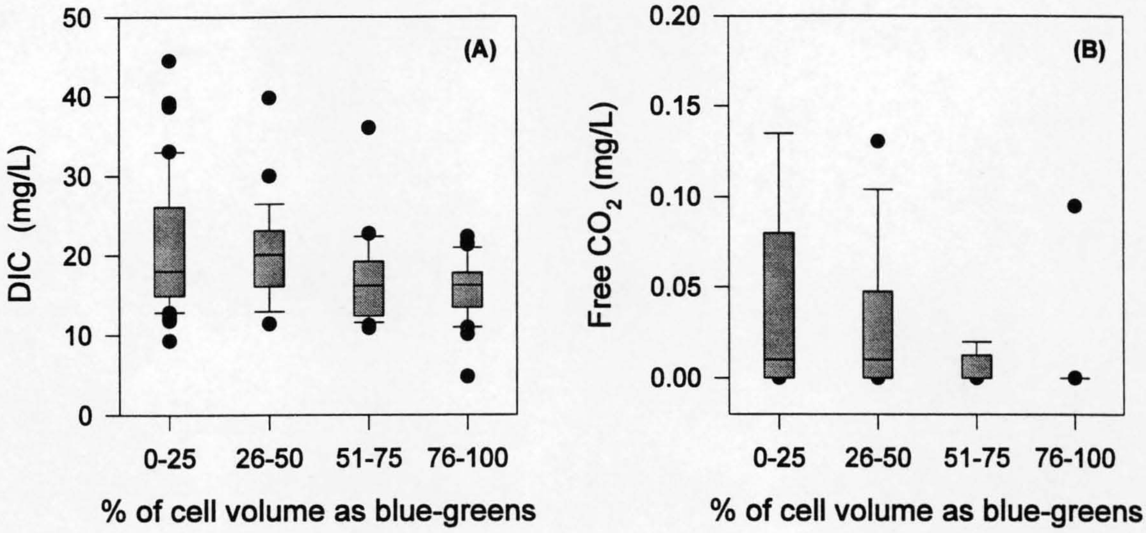

Figure 3.21 Box plots of dissolved inorganic carbon [DIC] (A) and free carbon dioxide (B), in relation to the relative dominance of blue-green algae. (Horizontal box lines are the median values; bars represent the 95th percentile. Data are pooled from all detention ponds, N=136.)

3.3 Discussion

3.3.1 Nutrient and phytoplankton observations

Throughout the study period, all three SWDPs sustained periods of very high phytoplankton biomass (>0.5 mg/L Chl *a*) and low Secchi disk depths ($\leq$0.2 m). Such levels can be considered to be approaching the maximum primary productivity achievable in natural systems due to self-shading light limitation (Bannister, 1974*a*).

The high P and low N conditions described for the ponds in this study resemble those reported for lakes that have been former recipients of sewage effluents, having had high N and P prior to diversion/nutrient load reduction. Following nutrient load reduction, such systems frequently maintain high P due to continued internal loading from P laden sediments, while N becomes increasingly diminished (*e.g.,* Forsberg and Ryding, 1980; Marsden, 1989; Ryding, 1978). As a consequence of a decreased N:P ratio, shifts in the phytoplankton composition to N-fixing blue-green species have been reported in some former wastewater recipients (Jensen and Andersen, 1990; Leonardson and Bengtsson, 1978).

Differences in the physical aspects of the ponds and their drainage basins resulted in slightly different nutrient characteristics. All three ponds were typically high in P relative to N, and DIP availability indicated that P was infrequently limiting phytoplankton growth. N availability was more variable. Depending on the extent of DIN limitation and the DIN:DIP ratios, the occurrence and density of heterocystous blue-greens ranged from severe (Lakewood) to moderate or low (Lakeview > Rochdale).

The fact that Rochdale was consistently dominated by *Oscillatoria* when the neighbouring Lakewood pond more often developed N-fixing *Anabaena* blooms, indicated N-limitation differences. While both ponds had very low dissolved N:P ratios (<3), higher N loads deriving from the 20 ha of arable land within the 90 ha Rochdale catchment is the most likely explanation for this difference. Also, in Lakewood the average hydraulic residence time will be higher since it drains only 68 ha of serviced land, resulting in a higher P retention coefficient, lower external N load, and a proportionately higher loss of N from denitrification relative to the external load.

As highly productive and shallow polymictic systems, benthic and water column mineralization of detritus/phytoplankton by bacteria are high. Although no evidence of an oxidized sediment surface was usually evident in the Rochdale and Lakewood ponds (see Chapter 4), decomposition was assumed to occur primarily under aerobic conditions. For the deeper Lakeview pond, sinking of algal cells out of the mixed layer resulted in a marginally lower average biomass compared to the shallow ponds, although it was not determined what differences in actual C-fixation rates exist between the systems. Variable degrees of restriction in the transfer of mineralized nutrients from the anaerobic hypolimnion to the euphotic zone occurred. Sedimentation rates and mixed zone to euphotic zone ratios would influence the share of aerobic and anaerobic decomposition within the system (Anderson and Jensen, 1992; Gunnison and Alexander, 1975; Jewell and McCarty, 1971; Molongoski and Klug, 1980; Ulén, 1978).

Leonardson and Ripl (1980) outlined three ecosystem models for hypereutrophic systems in relation to phytoplankton and availability of N and P: Type I systems are characterized by blooms of non-fixing blue-greens and, in particular, by *Oscillatoria* and *Microcystis* (Van Liere and Mur, 1980). These systems are described as having high internal recycling and availability of both P and NH_4-N. External inputs of P and NH_4 may also be significant. Type II systems show development of N-fixing blue-greens (*Anabaena* and

Aphanizomenon) in which P is high, but DIN supplies are insufficient to maintain non-fixing genera. In these cases, a lower input of N, lower recycling of N at the sediment surface, and a higher N loss *via* coupled by nitrification-denitrification may be responsible. Leonardson and Ripl (1980) further state that non-fixing blue-greens, greens, and diatoms may be associated in smaller amounts with the N-fixing blooms, assimilating NH_4 liberated from oxidation of nitrogenous organic matter derived from the fixed N supplement. Type III systems are those in which high external DIN loading (NO_3 in particular) permits a predominance of green algae. The Type III mechanism is also used to explain periodic shifts from both non-fixing and N-fixing blue-greens to green and diatom species in variably loaded Type I and Type II systems.

The phytoplankton dynamics of all three SWDPs in this study can be well described within the above framework. Rochdale can be predominantly classed as a Type I system, since temporary mid-summer shifts from an *Oscillatoria* dominated assemblage to green algae were observed on several occasions (*e.g.,* July/August 1994). In this case a *series* of runoff events increased DIN, reduced accumulated DIP, and reduced blue-green standing crops by flushing. Increased mixing, and reduced temperature and pH (affecting DIC availability) may also be involved. Ahlgren (1977) demonstrated that *Selanastrum* had a higher growth rate than *Oscillatoria* at high NO_3 supply in chemostats, which was the replacement scenario seen in Rochdale pond during 1994.

Lakewood pond showed strong tendency towards a Type II system during the summer months, with Type I characteristics following runoff series (excluding experimental 1994 data). Nitrogen-fixing blooms in Lakewood were always preceded by periods of very low DIN:DIP ratios, but no predictive indications of bloom formation could be made from these ratios alone, which were consistently low throughout the season (generally <1:1). High biomass N-fixing blooms occurred when dry sunny weather followed consecutively low N:P ratios and high preceding P levels. On the basis of DIN:DIP <5 (wt) implying N limitation to phytoplankton (Uhlmann, 1982), N-fixers may also have been expected in Rochdale pond, which never occurred throughout the study period. Barica (1990) gave comment on the predictive limitations of DIN:DIP ratios and suggested that the TN:TP ratio may be more useful since it includes the stabilizing influence of the particulate fraction within it. However, Barica (1990) was able to show that in four eutrophic lakes in Western Canada, which generally had DIN:DIP ratios >20:1 as a seasonal average, it was the preceding short-term ratio minima (<6-1:1) that was followed by development of N-fixers.

Lakeview pond tended to support a diverse phytoplankton community due to the complexity of stratification, intermittent and variable degrees mixing, and baseflows. The only large N-fixing bloom between 1993 and 1995 developed as stratification re-established after a series of mixing events in early July of 1993. It was notable that as the N-fixing bloom subsided, associated green biomass remained high, presumably sustained by N deriving from that fixed into the system over the previous weeks.

Although significant DIP concentration gradients were measured in subsequent seasons, algal biomass never reached that of the 1993 bloom, indicating that vertical diffusion of N and P across only a short distance was restrictive to growth. Substantial mixing followed by extended calm conditions appeared to be a prerequisite to dense bloom formation in this pond.

3.3.2 Particulate C:N:P ratios

Variations in the particulate C:N:P ratio may reflect relative amounts of carbohydrates, proteins, and lipids and, therefore, may reflect a physiological state (Vollenweider, 1985). Overall, the total data set mean ratios conformed almost exactly to those of the Redfield ratio.

It has been suggested that large C:N and N:P ratios correspond to that nutrient which is limiting growth and that small ratios reflect excess nutrients (Healey and Hendzel, 1978), although significant inter-species variations are known to exist (Hecky, 1988). Despite the consistently low DIN:DIP ratios (and high DIP), the seston ratios during periods of blue-green dominance had a slightly higher N:P ratio and lower C:N ratio than during green dominance.

Mineralization is reported to result in detritus with a higher C:N and C:P ratio than living cells (Andersen and Jensen, 1992; Jewell and McCarty, 1971; Uehlinger and Bloesch, 1987; Ulén, 1978). Therefore, although detritus is included within the seston ratios, the lower C:P and N:P ratio observed during green dominance cannot be explained by this mechanism. Inter-species variation is assumed responsible. Ambrose *et al.* (1993) gave similar seston ratios to those found in this study for general blue-green dominance (mean C:N =

3.9, C:P = 44, N:P = 11). Ulén (1978) also reported very similar ratios for *Oscillatoria* dominated seston from a Swedish lake (mean C:N = 4.8, C:P = 63, N:P = 13). The ratio change for green dominated seston was also similar to the mean value of this study (C:N = 5.6, C:P = 41, N:P = 7.3).

3.3.3 Chlorophyll - biomass relationships

Apart from the effect of nutritional status (especially N), adaption of photo-pigment content to light intensity is general among phototrophs, and may range over two orders of magnitude in freshwater phytoplankton (0.1 to 9.7% of fresh wt as chlorophyll) (Nicholls and Dillon, 1978; Reynolds, 1984). Substantial reductions in the Chl *a:b* ratios to values of 1 or less occurred during several high biomass periods. In these cases, competition for radiant energy induced high Chl *b* synthesis by greens when blue-greens were presumably dominating attenuation within the Chl *a* absorption spectra. Substantial light limitation was indicated at these times. These results suggest that modelling photosynthesis using quantum yield relationships for Chl *a* may underestimate C-fixation potential during periods when accessory pigments are high. For these hypereutrophic ponds, calibration of measured photosynthetic rate to a total chlorophyll attenuation coefficient may be useful, as would investigation of phycocyanin concentrations.

In this study, Chl *a+b vs.* biovolume gave a reasonable correlation, but improved regression relationships of both PC and suspended solids *vs.* biovolume show that pigment concentration, although convenient, may under- or overestimate biomass. Nicholls and Dillon (1978) reviewed data from 27 studies relating chlorophyll content to cell volume. Ranges for Chl *a* values (as $\mu g/mm^3$) for eutrophic Canadian prairie lakes and Swedish lakes were typically 6 to 25, and specifically for an N-limited lake, 1.4 to 16.5. In this study, the median seasonal value for Chl *a+b* was 10.6, which agrees well with other reported ranges.

3.3.4 Inorganic carbon, pH, and phytoplankton composition

The results of this study showed a tendency for a blue-green dominance to increase with decreasing DIC and CO_2 concentrations. In most lakes, DIC supply rates are usually not regarded as limiting to photosynthesis since diffusion of atmospheric CO_2 maintains a sufficient CO_2 supply (Schindler and Fee, 1973). However, in these hypereutrophic ponds high photosynthetic demand frequently drives the pH beyond 9.5 and in Lakewood, pH was often >10. The data shows that free CO_2 becomes completely depleted and, depending on the ability of different groups to utilize HCO_3^-, this may be a factor involved in dominance at high pH. Also, the high summer temperatures would make CO_2 less available *via* both dissociation and solubility changes. Shapiro (1990) reviewed the subject of DIC/CO_2 availability and utilization, and concluded from the evidence that blue-greens have lower K_s for CO_2 (Long, 1975; Talling, 1976), more efficient CO_2 concentrating mechanisms (Raven, 1985), a potential ability to directly utilize HCO_3^-, and potentially increased carbonic anhydrase activity.

Intense photosynthesis by diatoms and greens usually caused high pH shortly after ice-melt (pH >9.5). Although these early season populations were dominated by diatoms and greens, these pH conditions may subsequently favour the successful recruitment of blue-greens from the sediment surface (Reynolds, 1984). The greater frequency of high pH observations in Lakewood and Rochdale compared to Lakeview also coincided with more frequent dominance of the phytoplankton by blue-greens in the former ponds. Although the pH distribution may be more of a symptom rather than a cause, this would nevertheless be a reciprocal factor influencing the DIC equilibrium. It was also notable that during temperature stratification, DIC concentrations were higher and pH lower in bottom water, so that this additional generative "CO_2 reservoir" may further increase CO_2 supply to the euphotic zone in Lakeview pond.

3.3.5 Zooplankton

The crustacean zooplankton in all three ponds were dominated by cyclopoid copepods. Lakeview pond periodically supported a higher proportion of cladocerans compared to either Rochdale or Lakewood ponds. Studies on several other relatively new SWDPs, which are not reported in this thesis, showed that large-bodied cladoceran populations may initially develop, but are gradually lost within only a few years after construction. The declines coincide with increasing nutrient content of the ponds, and the increase in fathead minnow numbers (*Pimephales promelas*). Considerable grazing pressure may be exerted on large-bodied zooplankton by minnows (Morin *et al.,* 1991). The inhibition of large-bodied cladocerans with increasing abundance of blue-greens has been documented, along with several hypotheses on the reasons (*e.g.*, Fulton and Paerl, 1988*a*, 1987; Hessen *et al.*, 1986; Porter, 1977) The only cladoceran which occurred in any numbers in the shallow Regina SWDPs was *Bosmina longirostris*, a species reported to be resistant to blue-green algal toxins (Fulton, 1988*a*, 1988*b*).

Recycling of nutrients and phytoplankton loss mechanisms *via* zooplankton grazing was probably insignificant in the shallow Regina SWDPs relative to the otherwise high background nutrient levels (*e.g.*, den Oude and Gulati, 1988). However, Lakeview pond has the potential to undergo clear water phases due to *Daphnia* grazing, as seen in the early summer of 1992. The reasons why such phases did not occur in other years of the study is not clear. This may relate to the extent of oxygen depletion on winterkill of minnows and, in turn, may relieve grazing pressure on zooplankton until populations recover.

A suitable phytoplankton species composition is a prerequisite for effective grazing and would be required to coincide with the cyclical production of any large-bodied species. In these unstable systems, such a combination of appropriate environmental conditions and timing may vary from one season to the next. Furthermore, large stormwater exchange volumes, typical of shallow ponds with higher catchment:pond volume ratios, may flush greater numbers of zooplankton from the system and further limit grazing potential.

3.4 Conclusion

All three SWDPs are in a hypereutrophic condition, with summer phytoplankton assemblages frequently dominated by Cyanophytes. Winter anaerobiosis restricts food web diversity and biomanipulation potentials. Current P levels reach 5 to 10 times higher than that required to prevent bloom formation, and N is frequently the limiting macronutrient during the summer period. Stormwater loading and periodic N-fixing cyanophyte blooms may temporarily restore nitrogen sufficiency to the systems. Inorganic-C limitation may be a significant factor influencing cyanophyte dominance during high biomass periods. Throughout the study period, zooplankton grazing appeared to be predominantly ineffective as an algal loss mechanism.

Elemental ratio variation among algal groups indicate that these quotas should be taken into account for accurate calculation of N and P uptake based on C-fixation. For reasons of algal self-shading and pigment adaption to light limitation, variable chlorophyll *a* to C ratios and correction factors for accessory photo-pigments may also need to be adopted for modelling algal growth in these hypereutrophic ponds.

Chapter 4

SEDIMENT CHARACTERISTICS AND INTERNAL NUTRIENT LOADING

4.1 Introduction

The role and significance of sediments as sinks and sources for plant nutrients have been extensively described (*e.g.,* Boström *et al.,* 1988*a*, 1982; Forsberg, 1989; Marsden, 1989). Export of particulate and soluble nutrients from urbanized land may be greatly increased in comparison to an undeveloped watershed (Walker, 1987). Much of the TP in urban runoff tends to be associated with clays and mineral particulates so that settling of this fraction is characteristic during temporary storage, while most of the soluble component is immediately available for algal assimilation (Boström *et al.,* 1988*b*). Suspended particles in stormwater have been found to settle twice as quickly as predicted by Stokes Law, and is indicative of the aggregative tendency of the solids (Striegel, 1987). Therefore, P loading and retention in SWDPs may be very high according to pond design and operational characteristics. Phosphorus retention efficiencies of 80 to 90% have been reported for well designed wet-detention ponds (Walker, 1987).

Since SWDPs may exist as closed systems during inter-event periods (a few days to many weeks), with a relatively high sediment surface area/pond volume ratio, internal loading of nutrients may dominate water column chemistry. This chapter presents the results of sediment composition of the three SWDPs in this study, the distribution of P among various sediment fractions, and a basic quantification of the P release and uptake potential.

4.2 Results

4.2.1 General sediment composition

A compilation of elemental composition, organic matter, density, and porosity for the three SWDPs is summarized in Table 4.1. In all three ponds, surficial sediment taken in 1992 (one central location) and 1995 (mean of two locations) were very similar in composition of all parameters measured. However, it should be stated that the water, organic matter, and TP content of the surficial sediment fluctuated on a seasonal basis, while the total metal concentrations remained more stable. The variation of water content complicated interpretation of changes in the areal mass of elements. The two shallow ponds, Rochdale and Lakewood, had similar proportional characteristics for Al, Fe, and Ca in the surficial sediment, with Al>Fe>Ca by weight. Lakewood pond sediments contained more P than Rochdale, which resulted in atomic ratios of Al:P, Fe:P, and Ca:P being slightly lower in this pond. For both ponds, total Fe, Mn, and Ca tended to decrease in concentration with depth; however, when the increasing sediment densities are considered, the differences on a volumetric basis are diminished. Phosphorus, organic matter, and percent C all showed a fairly steep decreasing gradient from 0 to 6 cm, and especially in Lakewood. In this regard, Lakewood sediments are different in structure and depth compared to Rochdale. In Lakewood, the fine sediment is confined within a 6 cm upper layer, showing a steeper porosity gradient than Rochdale. Beneath this upper 6 cm, the sediment assumes a dense and gas pocketed structure. The Rochdale sediments are deeper, showing gradual compaction over the entire profile, and generally without gas pockets (Table 4.1).

The deeper Lakeview pond has markedly different sediment characteristics compared to the shallow Rochdale and Lakewood ponds. The sediments are much deeper and, like Rochdale pond, showed homogeneous fines with increasing compaction throughout the profile. The north end of the pond in the vicinity of the outflow pipe had deep anaerobic and organic rich sediment in excess of 30 cm deep, but at least 75% of the pond was in the 16 to 22 cm range. The organic matter content was higher than the other ponds, reflecting prolonged anaerobic conditions, and possibly a reduced tendency for resuspension. The TP, organic matter, and porosity profiles decreased more gradually with depth over an upper 12 to 14 cm layer. Aluminium and Fe content was lower than in the other ponds, while Ca was about twice as high. Therefore, the sediments have a low Fe:P and high Ca:P atomic ratio (Table 4.1).

Total-P and organic matter profiles decreased with depth and were positively correlated in all cases. For Rochdale, Lakewood, and Lakeview, the correlation coefficients were respectively: R^2 = 0.93, 0.96, and 0.90. The TP profiles were used to estimate the depth of the unstabilized layer, and the exchangeable P pool (Figures 4.1, 4.2, and 4.3). Neglecting compaction, the ratio of the exchangeable P:TP (F) for the three ponds is F=0.4 (Rochdale); F=0.5 (Lakewood); F=0.5 (Lakeview), so that about 40 to 50% of the P input to the sediments is mineralized/desorbed and released.

Table 4.1 Elemental composition and bulk sediment structure characteristics. (± standard deviations: elements, organic matter, and porosity, n = 9; sediment depth, n = 20)

Pond	Rochdale			Lakewood			Lakeview		
Year	1992	1995		1992	1995		1992	1995	
Sample	Surficial	Vertical Range		Surficial	Vertical Range		Surficial	Vertical Range	
Horizon	0-1 cm	0-1 cm	5-6 cm	0-1 cm	0-1 cm	5-6 cm	0-1 cm	0-1 cm	11-12 cm
Al (mg/g)	72.2 (1.9)	70.9	69.5	71.4 (7.0)	58.0	76.2	41.5 (2.9)	40.0	56.0
Fe (mg/g)	37.8 (1.6)	40.6	37.7	37.8 (3.0)	34.8	32.7	26.5 (1.7)	27.3	37.6
Mn (mg/g)		0.53	0.37		0.72	0.50		0.48	0.64
Ca (mg/g)	34.4 (2.4)	33.4	33.2	30.8 (5.6)	39.6	26.7	81.7 (14.2)	92.1	64.6
TP (μg/g)	970 (60)	1190	775	1020 (130)	1480	830	1260 (260)	1270	840
Al:P (atoms)	86 (7)	69	102	82 (17)	45	105	40 (9)	36	76
Fe:P (atoms)	22 (2)	19	27	21 (4)	13	28	12 (2)	12	25
Ca:P (atoms)	28 (3)	22	33	23 (4)	21	25	51 (6)	56	59
% LOI	11.3 (1.2)	13.3	10.7	11.4 (1.4)	17.5	10.1	17.9 (3.0)	17.5	13.4
% C		5.3	3.8		8.3	4.0		11.7	6.5
% H_2O	62 (4)	57 (2)	56 (1)	63 (3)	59 (2)	43 (4)	59 (6)	57 (3)	63 (3)
Density		1.17	1.22		1.16	1.33		1.15	1.35
Mean Depth (cm)	13 (2.2)			9 (2.0)			21* (3.6)		
Accum. Rate (mm/yr)	10			7			15		

*deep anaerobic sediment >30 cm in vicinity of north end outflow pipe

The deeper Lakeview pond sediments showed a more gradual reduction of TP and organic matter than those measured in the shallow non-stratifying ponds. A sequential P extraction profile was obtained for the Lakeview sediments, and showed that iron and aluminium-bound P accounted for a very small fraction of the TP. On the other hand, the calcium-bound P fraction was relatively high, increasing to a depth of about 7 cm. Unlike the shallow ponds, organic matter content was fairly stable down to a depth of about 7 cm, at which point it became progressively reduced (Figure 4.3). These characteristics resulted in interstitial P profiles which were markedly different from those of the shallow ponds.

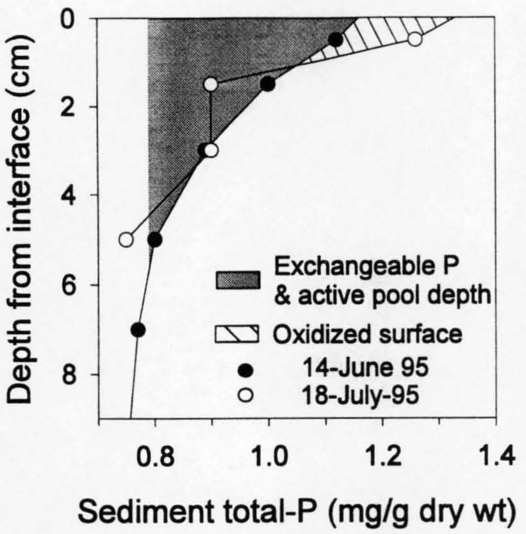

Figure 4.1
Sediment total phosphorus profiles in Rochdale pond with and without oxidized surface layer, 1995.

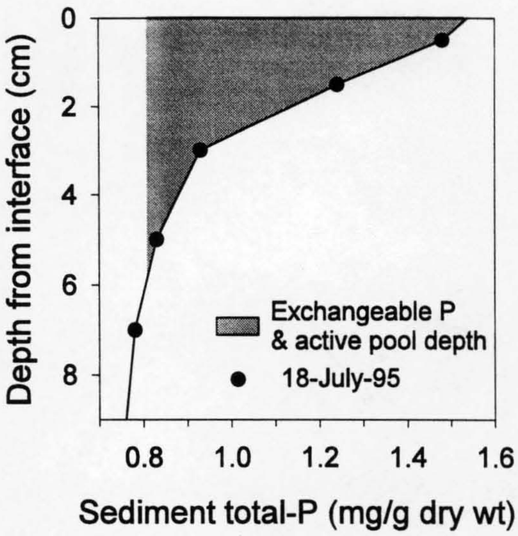

Figure 4.2
Sediment total phosphorus profile in Lakewood pond, 1995.

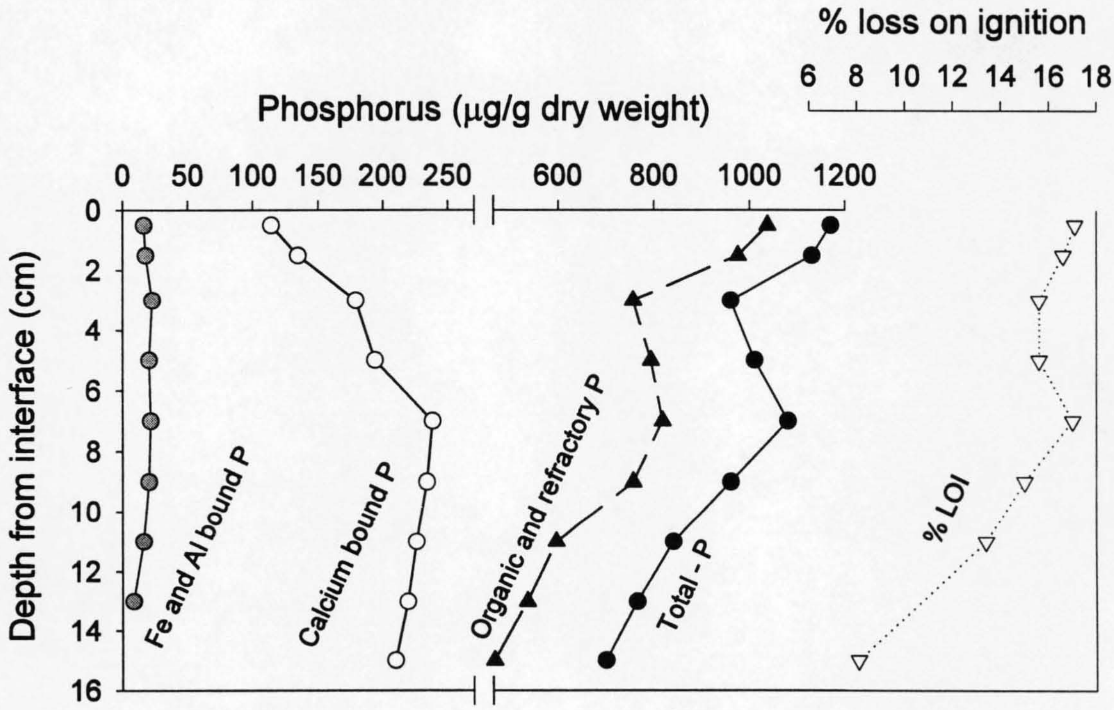

Figure 4.3 **Sediment total phosphorus profile and fractional composition in Lakeview pond, spring of 1995.**

4.2.2 Seasonal changes in phosphorus fractions

In 1992, monthly sampling of the active surficial sediments (≈1cm) was carried out in all three ponds. As temperatures increased, progressive internal loading of P was most evident in the shallow Rochdale and Lakewood ponds. Extended dry weather periods allowed net P loading to be calculated based on water column increases (Table 4.2). Evaporation was not regarded as significant over the 7 day periods, and precipitation of less than 4 mm was assumed not to generate runoff. Net release rates were higher in Lakewood than Rochdale, but considering phytoplankton sedimentation differences, the temperature based trend was quite similar.

Table 4.2 Net internal phosphorus loading in Rochdale and Lakewood stormwater detention ponds (Regina), 1992.

Date 1992	Total Precip*	Rochdale dTP/dt	Lakewood dTP/dt	Mean Water Temp 1.0 m
	mm	$mg/m^2/day$	$mg/m^2/day$	°C
23 Apr - 7 May	0	7	12	12
13 May - 20 May	2.2	8	13	13
23 Jun - 30 Jun	3.8	14	29	19
29 Jul - 4 Aug	0	24	46	20
15 Sep - 24 Sep	1.6	8	5	11

*Regina Airport, Environment Canada, AES

Sequential P fractionation, and measurement of sediment TP and organic matter content, provided insight into the relative contributions of organic and inorganic P pools to the net internal load. Unfortunately, bottom samples were not collected in Lakeview pond during 1992 and an areal TP mass could not be calculated. However, fluctuations in P availability were indicated by changes in the epilimnetic chlorophyll concentration.

The inter-relationship between the seasonal sediment changes and water column TP and Chl *a+b* concentrations are shown in Figure 4.4 a,b,c. Extractable inorganic-bound P fractions correlated positively with sediment TP content, and accounted for 11 to 15% of the pool in all three ponds.

Calcium-bound P (HCl-P) dominated the inorganic fraction in all ponds and was highest in Lakeview, reflecting the higher Ca concentrations of the sediment. Anaerobic conditions maintain a reduced P-binding potential by Fe and presumably led to a higher loss of Fe from the Lakeview system, with outflows drawn from an anaerobic hypolimnion. It was notable that, despite lower Fe concentrations in the Lakeview sediment, NaOH-P was similar between the ponds. Since the sediments were freeze-dried, the potential for movement of P to the Fe fraction by oxidation during drying should have been minimized. Therefore, P bound to Al may constitute a larger part of the NaOH extractable fraction in Lakeview. The maximum-minimum change within the surficial TP pool (~1 cm) for each pond, with the percent of the mass contributed by the inorganic extractable-P shown in brackets, was as dry weight: Rochdale, 175 μg/g (5.7%); Lakewood, 436 μg/g (9.8%); Lakeview, 700 μg/g (2.2%) (Figure 4.4 a,b,c).

Total P and organic matter content of the surficial sediment were positively correlated on most sampling dates, except in Rochdale, which showed a divergence from this trend in the latter part of the summer. This anomaly might be explained if these samples were inadvertently collected closer to inlets, where higher mineral particulates lowered the % LOI. The organic matter-TP relationship indicated that changes in the exchangeable-P, and hence P release, were primarily influenced by the balance between the rate of organic sedimentation and the temperature dependant mineralization rates. Associated changes in the inorganic adsorption equilibrium contributed less than 10% to the mass P decrease/increase under anaerobic conditions in these sediments (Figure 4.4 a,b,c).

Internal P loading was generally reflected by an inverse P balance in the surficial sediment. There was also an inverse relationship between sediment organic content and the water column Chl *a+b* concentration, with peaks in organic matter and TP content following significant water column reductions (Figure 4.4 a,b,c). In this regard, the pattern of productivity and sedimentation was slightly different in Rochdale compared to Lakewood during 1992. In Rochdale, the bulk of organic settling to the sediments followed the decline of a biomass peak in September. Over this period, there was greater oscillation of biomass in Lakewood, which led to a more continuous settling of organics, indicated by increasing sediment organic and P content (Figure 4.4 a,b). In Lakeview pond, the inverse relationship between sediment organic matter and productivity was remarkably clear, indicating lower sedimentation rates during phases of biomass development. It should be stated that the extent of the clear water phase, which produced the large peak in sediment organic matter and TP in 1992, was not observed in subsequent seasons and such wide fluctuations in sediment surface characteristics were probably not typical. The rate at which organic C and P were mineralized during August was remarkably high, considering an anaerobic hypolimnion (Figure 4.4 c).

The declines of sediment organic matter content between July and August 1992 were associated with periods of increasing blue-green biomass in all three ponds (*Oscillatoria/Anabaena* in Rochdale and Lakeview, and *Anabaena* in Lakewood). It was apparent that the increasing biomass of buoyant blue-green species initially resulted in reduced sedimentation rates, permitting mineralization to exceed sedimentation. The shallower slope of organic matter decline in Rochdale, compared to that of Lakewood, indicated a higher degree of sedimentation associated with the higher biomass (Figure 4.4 a,b). Assuming a hypothetical organic-C sedimentation rate of zero in the early stage of bloom development for the period June 18 to July 9 in Lakewood, a sediment organic C mineralization rate of 0.73 mg org-C/g/day at $\approx$20°C was calculated (assumed organic-C to be 50% of the ash-free dry weight). The associated rate for organic-P was 0.008 mg org-P/g/day. The values for the decline in Lakeview were slightly higher for both parameters - 1.04 mg org-C/g/day and 0.02 mg org-P/g/day at $\approx$16°C. Since the sedimentation rate was not known, comparison of these rates is limited. From these approximations, a gross P regeneration rate of 38 mg/m^2/day (Lakewood), and 90 mg/m^2/day (Lakeview) would result for the top 1 cm. Although these values are vague in the absence of sedimentation and sediment profile measurements, they certainly fall within a realistic boundary for release rates based on other measurements (Table 4.2). Søndergaard *et al.* (1990) reported gross release rates of 100 to 200 mg P/m^2/day for several weeks following bloom collapse in a shallow hypereutrophic Danish lake, values which were 2 to 3 times higher than the net internal load calculated from the nutrient budget.

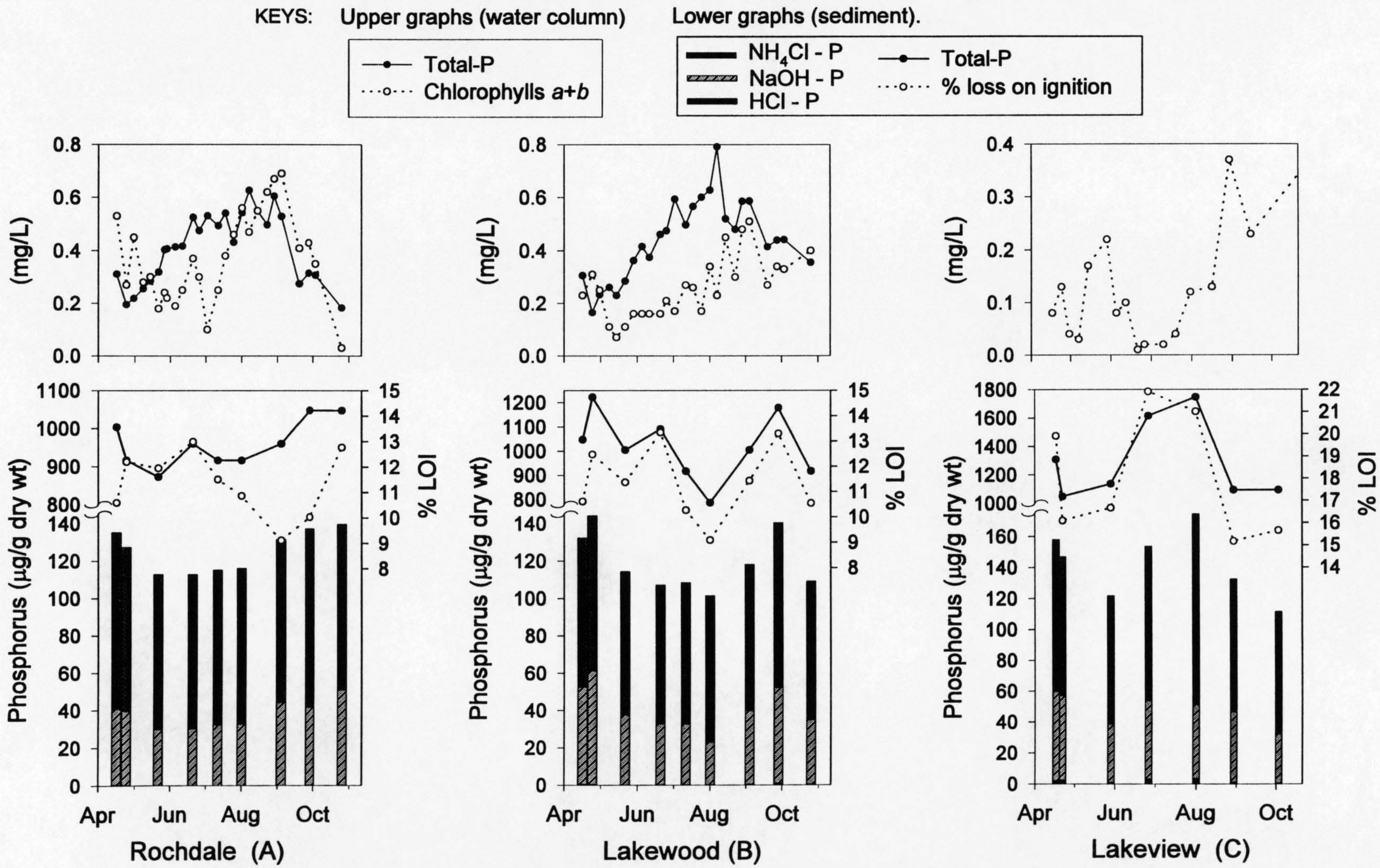

Figure 4.4 **Seasonal changes of surficial sediment P and organic matter content in Rochdale (A), Lakewood (B), and Lakeview (C) ponds in relation to water column concentrations of TP and chlorophyll, 1992.**

4.2.3 Interstitial phosphorus and ammonium profiles

As part of routine monitoring and as a complement to laboratory incubations, a number of interstitial profiles were measured between 1993 to 1995 for DIP and NH_4. Marked variation in interstitial concentrations and profile characteristics were measured across the seasonal temperature range. In addition, 1995 data showed that horizontal variation in concentrations and profiles exist within the ponds, although for the most part differences were relatively small (no profiles were measured very close the deltoid depositional areas around inflow pipes). Average interstitial P (P_{int}) for the top 0 to 1 cm and 1 to 2 cm fractions produced positive correlation with the overlying water temperature, although the variance was often quite large (Figures 4.5, 4.6, 4.7). For each Lakeview and Lakewood ponds, R^2 was 0.53 (n=8). In Rochdale pond, data obtained at times when the sediment surface was oxidized (following alum addition - see Chapter 7) was isolated from the remaining data and R^2 was 0.65 for reduced samples.

In Rochdale pond, sediment surface oxidation drastically reduced the interstitial P concentration (Figure 4.5), while the TP pool was increased in the upper layer due to adsorption (Figure 4.1). As with the Lakeview and Lakewood data, samples analysed in 1993 tended to be slightly higher in DIP than the 1995 samples within the same temperature range, although two sample locations were used in 1995. Sustained organic productivity and lower rates of flushing losses in 1993 may have increased the organic P pool of the surficial sediment, thereby increasing the P_{int}. Interstitial NH_4 concentrations were measured in 1995 cores and there was a much higher variance in the temperature regression than with P_{int} in all ponds (NH_4 data not shown). The results indicate the influence of both the substrate mass and the temperature coefficient.

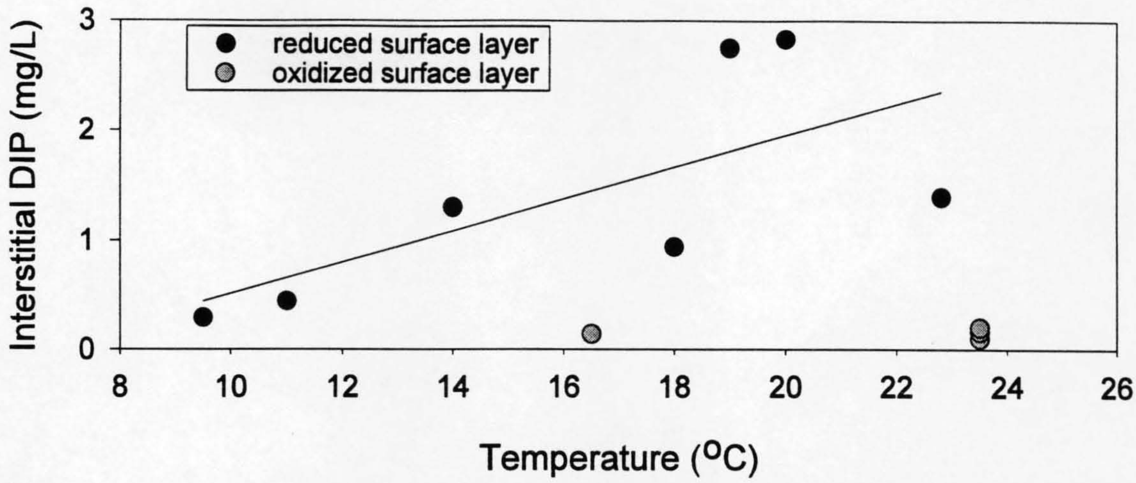

Figure 4.5 **Interstitial concentration of dissolved inorganic phosphorus in the surface sediments (1 cm) of Rochdale stormwater detention pond (Regina) with temperature.**

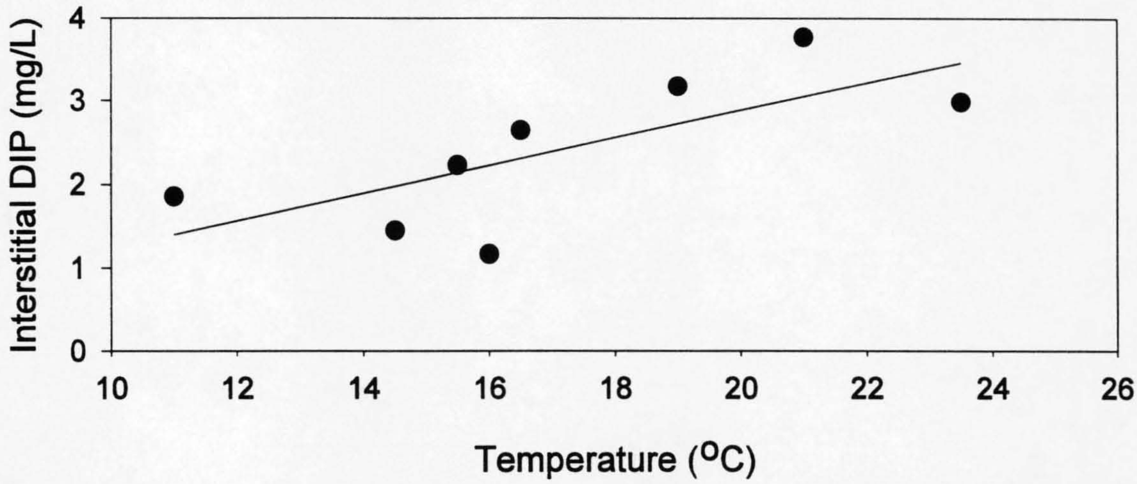

Figure 4.6 **Interstitial concentration of dissolved inorganic phosphorus in the surface sediments (1 cm) of Lakewood stormwater detention pond (Regina) with temperature.**

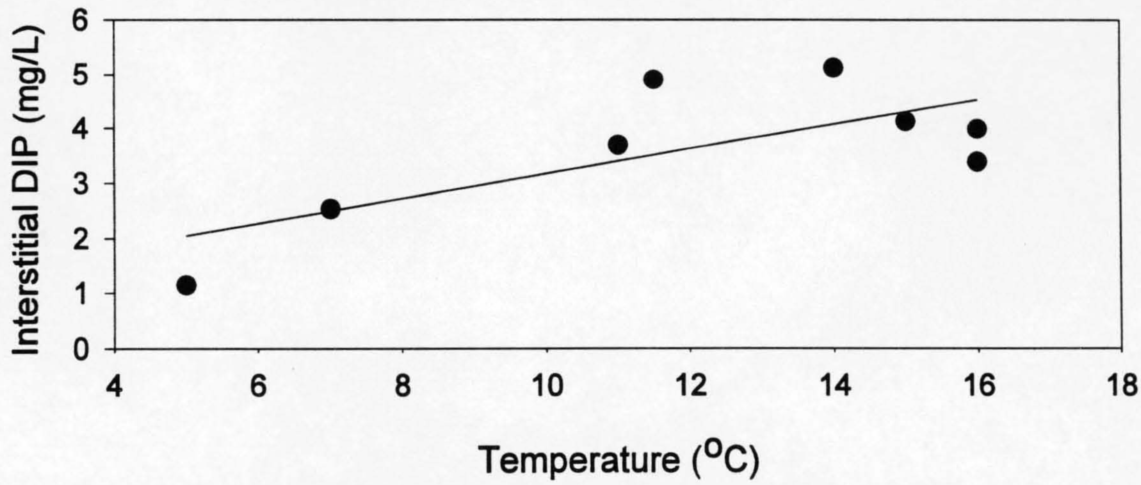

Figure 4.7 **Interstitial concentration of dissolved inorganic phosphorus in the surface sediments (1 cm) of Lakeview stormwater detention pond (Saskatoon) with temperature.**

The P_{int} profiles for the Lakewood and Rochdale ponds in 1995 are shown in Figures 4.8 and 4.9. Lakewood data clearly showed the effect of temperature, with the active layer depth gradually increasing as the season progressed; N:P ratios increased with depth in all Lakewood pond samples. The lowest N:P ratios measured shortly after ice-melt indicate a lowered organic content following the winter period. Large increases in NH_4 under winter ice cover occur in these ponds, while more PO_4 may be retained due to adsorption differences for the ions. Return of P to the sediment pool may also accompany Fe oxidation when the system becomes reoxidized in the spring, while more N may be lost due to coupled nitrification/denitrification. There may also be a greater proportion of organic-N which is refractory compared to organic-P (Jewell and McCarty, 1971). Formation of vivianite ($Fe_3(PO_4)_2{\bullet}8H_2O$), octacalciumphosphate ($Ca_4H(PO_4)_3$), and hydroxy-apatite ($Ca_5(PO4)_3OH$) are minerals which have been reported to control phosphorus and metals ion concentration in lake sediments (Matisoff *et al.*, 1980). Ion activity potentials were not calculated in this study.

The comparable spring profile for Rochdale pond was very similar to that of Lakewood. Although interstitial data for the June sampling is missing, the effect of sediment surface oxidation on the concentrations and profile shape were marked. What was notable was that the water column DIP and surficial P_{int} were much closer to equilibrium when the sediment was oxidized, indicating unsaturated adsorption capacity. The adsorption of P by the oxidized upper layer caused a large increase in the N:P ratios. The data also showed that diffusion of P_{int} from deeper layers (at least 6 cm) becomes more significant when an oxidized upper layer is present (Figure 4.9).

4.2.4 Anaerobic and aerobic incubation of intact sediment cores

A number of experiments were run in batch mode under constant temperature conditions using undisturbed sediment cores. A total of five anaerobic incubations were run in replicates of 8 to 16 cores. Four of these experiments involved anaerobic release from Lakeview pond cores (5 to 20°C), and one from Rochdale pond cores (20°C). Two aerobic incubations at 10°C were carried out with cores from the Rochdale and Lakewood ponds. In addition, adsorption isotherms were measured for surficial sediment suspensions from Rochdale (aerobic and anaerobic) and Lakewood ponds (aerobic only).

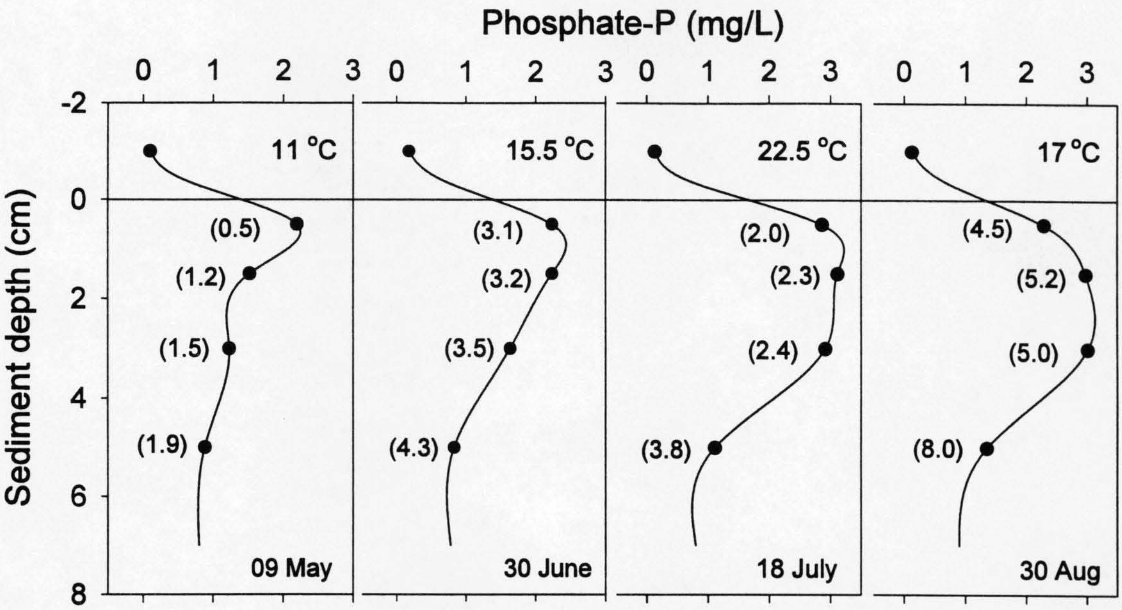

Figure 4.8 Seasonal development of interstitial dissolved inorganic phosphorus profiles in Lakewood pond sediments, 1995. (Numbers in brackets are the DIN:DIP ratios and lines are interpolated values to the water column DIP concentration.)

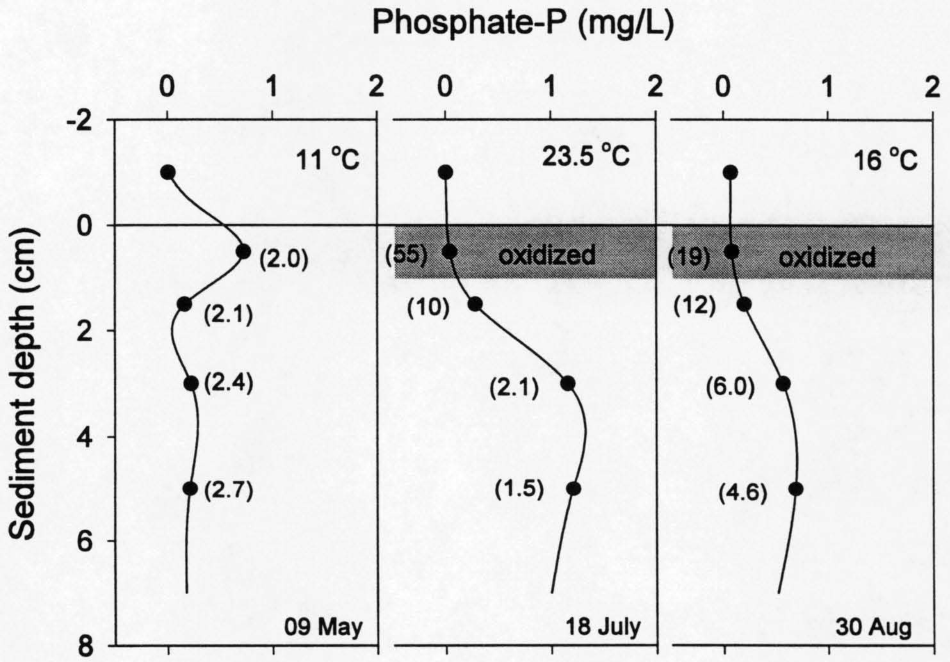

Figure 4.9 Seasonal development of interstitial dissolved inorganic phosphorus profiles in Rochdale pond sediments, 1995. (Numbers in brackets are the DIN:DIP ratios, and lines are interpolated to the water column DIP concentration.)

Anaerobic incubation of Lakeview cores: In all anaerobic experiments, considerable variation of DIP release rates were typical among the replicate cores for each of the release periods, and particularly the initial release rates. Cumulative release averages correlated positively with increasing temperature, and although the relationship was not quite linear, the Q_{10} was close to 2. Average DIP release rates at 12 days were 5.0 mg/m^2/day at 5°C (n=8); 9.2 mg/m^2/day at 10°C (two experiments, n=16; n=10; $P<0.01$); and 15.2 mg/m^2/ day at 20°C (n=8) (Figure 4.10). In all experiments, release rates tended to decline gradually over time. This effect was more apparent at 5 and 10°C, since these experiments were run for a longer period. The effect was presumed to result from decline of organic-P in the sediment surface, and not as a direct result of a decrease in the rate of diffusion: Removal and replacement of 15 to 30% of the reservoir volume on each sampling day tended to keep the DIP concentration within a consistent range throughout the experiments. Continuous flow experiments would have allowed better control of this parameter. In all experiments, pH was buffered by the sediments at between 8.2 to 8.5, which is close to the typical range of hypolimnetic values measured in this pond.

Only the 10°C experiment was carried out twice, and the pattern and magnitude of the cumulative P release was very similar in each case. The extent of the agreement was surprising in that the cores were collected nine months apart, although at the time of collection, hypolimnion temperatures were close to 10°C in both cases. Unlike the sediments from the shallow ponds, Lakeview interstitial profiles measured before each of the experiments (after 48 hours equilibration) show that diffusion from deeper horizons is more significant in maintaining the net release of P. The measured profiles show that average reservoir concentrations measured over the course of the experiment were never greater than about 30% of the pre-experimental interstitial concentration at 1.5 cm, and usually less than 20% (Figure 4.11).

The P_{int} concentrations measured prior to the 20°C experiment were only marginally higher than those recorded in both 10°C experiments, and this may go some way to explaining why the 20°C release rates were lower than might be expected from a doubling of temperature. In this case, the cores were collected at a hypolimnetic temperature of 16°C and then transferred to 20°C.

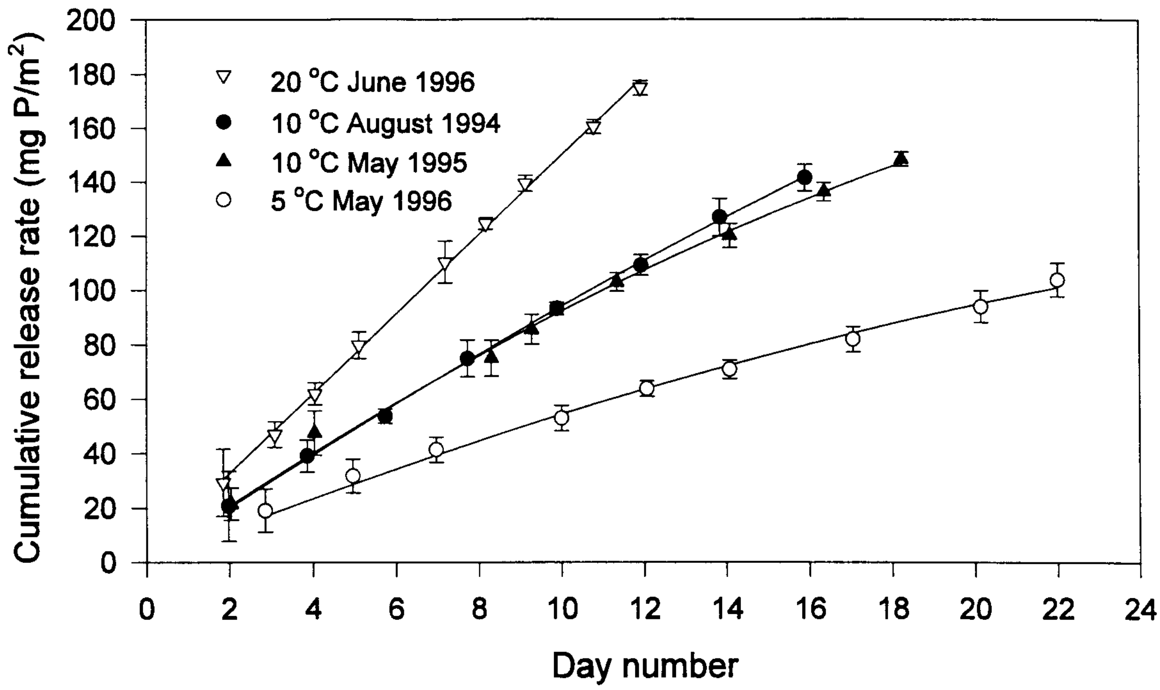

Figure 4.10 Anaerobic dissolved inorganic phosphorus release from intact Lakeview pond sediment cores, 5 to 20°C. (Error bars = standard error of the mean; 20°C and 5°C, n = 8; 10°C 1995, n = 10)

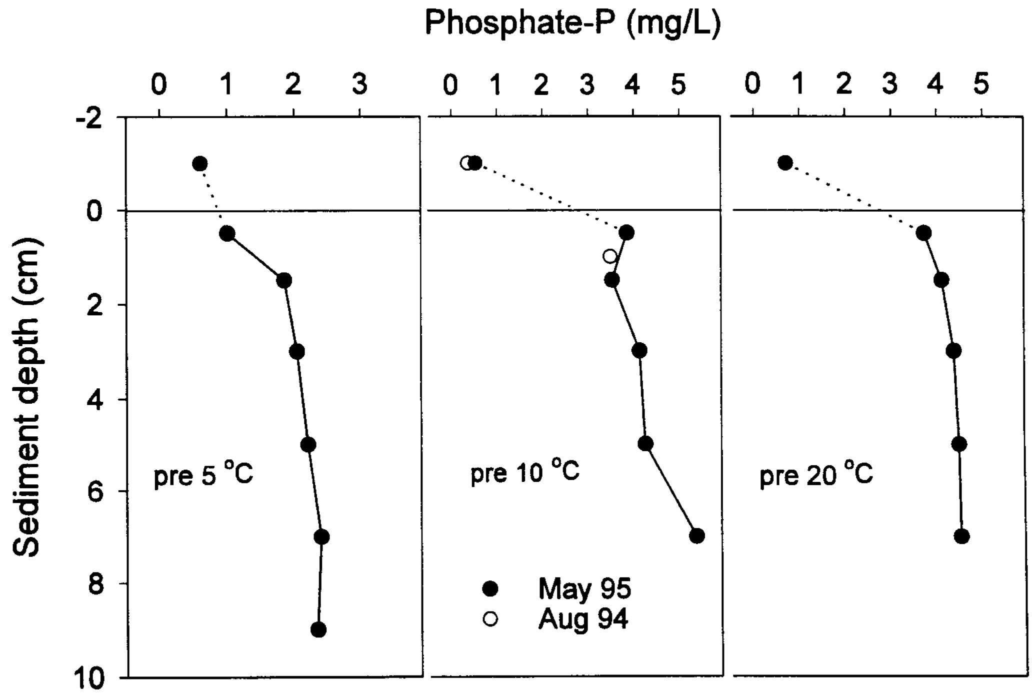

Figure 4.11 Interstitial dissolved inorganic phosphorus relative to mean reservoir concentration in anaerobic release experiments with Lakeview pond intact cores, 5 to 20°C.

Dissolved phosphorus concentrations may be increasingly affected by precipitation at higher concentrations, most likely by interaction with Ca^{2+}, as suggested by the increase in HCl extractable P with depth described previously. Alternatively, horizontal variation of sediment characteristics within the pond and/or lower organic-P content may have been involved. Unfortunately, the elemental composition of these cores was not measured to allow for comparison.

No groundwater seepage is assumed to occur in any of the detention ponds, so that the flux of P is determined by the balance of mineralization, diffusion, and chemical reaction (adsorption/precipitation) only. Bioturbation of the highly reduced bottom sediment is likely insignificant in the Lakeview pond.

Considering only the diffusive flux based on the measured interstitial profile as:

Equation 4.1

$$J_i = -\phi D_s \left(\frac{dC}{dx}\right)$$

where:

J_i = diffusive flux of component i in mass unit area^{-1} unit time^{-1}

ϕ = sediment porosity

D_s = whole sediment molecular diffusion coefficient

C = component concentration

x = distance

The molecular diffusion coefficient for phosphate was obtained from Li and Gregory (1974). Calculation of tortuosity was ignored since resistivity ratios of sediment and water would converge towards the sediment water interface, while porosity approaches 100%. The effect may be small relative to the uncertainty of the diffusion coefficient itself (Berner, 1980). At pH 7.2, the concentration of HPO_4^{2-} and $H_2PO_4^-$ species would be equal. This pH value is slightly below a typical value for the upper sediment (pH 7.7), but for simplicity an average of the diffusion coefficients for each PO_4 species was used (7.34 and 8.46 $*10^{-6}$ cm^2 sec^{-1}, Li and Gregory, 1974), corresponding to a value of 7.9 $*10^{-6}$ cm^2 sec^{-1}. Concentration gradients were estimated using the 1.5 cm (1 to 2 cm fraction) P_{int} concentrations, and the average concentration in the mixed reservoir throughout the experiments was assumed to represent the interface value.

Therefore, concentration gradients for the 5, 10, and 20°C experiments were 0.83, 1.99, and 2.28 μg P cm^{-4} and the corresponding flux rates calculated to 5.7, 13.6, and 15.7 $mg/m^2/day$, respectively. These values are remarkably close to the measured average release rates, considering the use of an average reservoir concentration in the surficial gradient calculation. Despite the convenience in the selection of the diffusion coefficient, the results show that a mineralization-diffusion based model explains anoxic release in this sediment.

Aerobic and anaerobic adsorption isotherms of the Rochdale and Lakewood sediments: In an effort to confirm the typical internal loading values calculated for the shallow ponds (Table 4.2), eight Rochdale cores were incubated anaerobically at 20°C, which is a typical mid-summer temperature overlying these sediments. Experimental pH was in the range 8.3 to 8.7, which is slightly lower than the *in-situ* pH of Rochdale during the summer. However, under anaerobic conditions the effect of ligand exchange at high pH may be less significant than in aerobic sediments (Boström, 1982). A further six cores from each of Rochdale and Lakewood were incubated aerobically at 10°C.

As in the previous experiments, substantial variation in the anaerobic release occurred among replicate cores, but the mean values produced an almost linear release of about 15 $mg/m^2/day$ after one week (Figure 4.12). One of the reasons for the variation in release may have been the effect of mixing by the variable gas flow being supplied to each column. Surface area differences may also exist among cores, not only due to the physical effects of containment within the tubes, but also by undulating surfaces and the presence of a few *Chironomid* larvae (Diptera) in some cores, which were initially able to survive the low oxygen conditions. Unfortunately, P_{int} concentrations were not measured within a sufficient time frame of extraction to allow reliable interpretation in this experiment.

Aerobic incubation resulted in net uptake of DIP from the reservoir and suppression of DIP release in cores from both ponds (Figure 4.13). A lack of DIP residuals in the reservoir meant that the uptake rates shown were not maximal, but nevertheless demonstrated the adsorption capacity of these sediments upon oxidation.

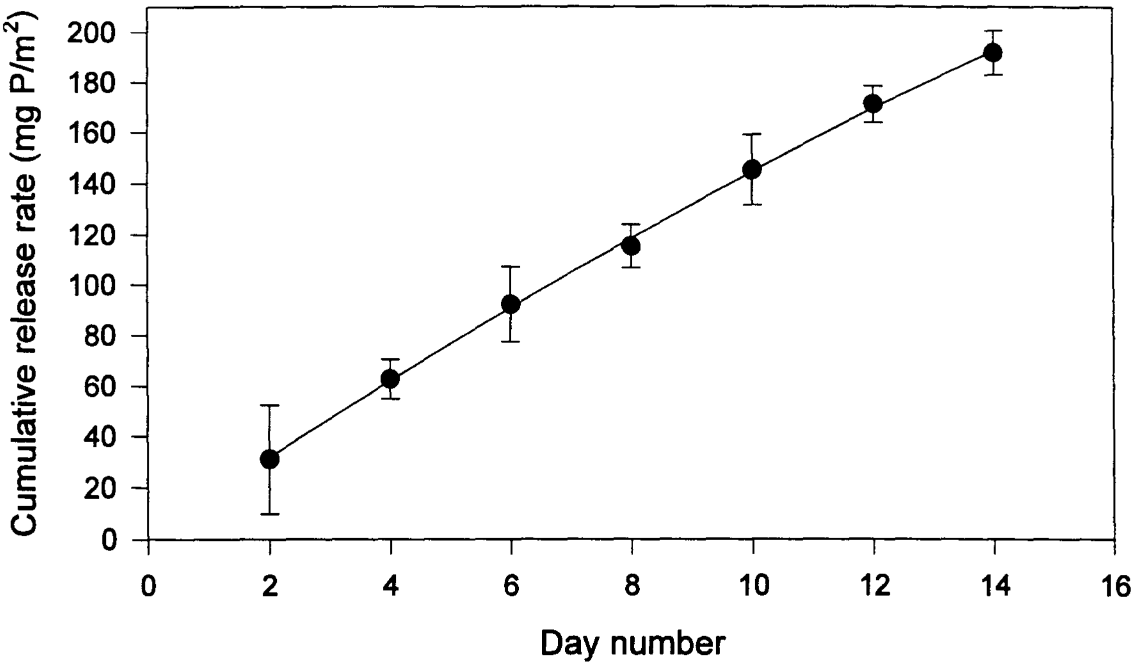

Figure 4.12 Anaerobic dissolved inorganic phosphorus release from intact Rochdale cores at 20°C.

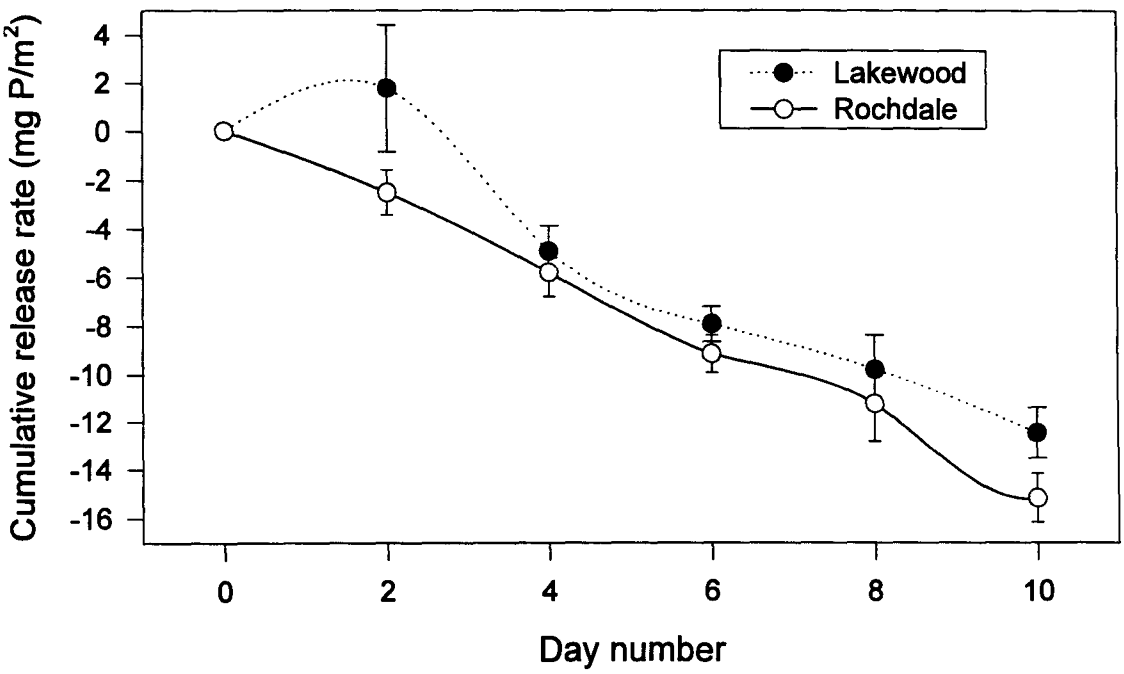

Figure 4.13 Aerobic dissolved inorganic phosphorus uptake in Rochdale and Lakewood intact cores at 10°C.

The uptake is presumed to result from adsorption of DIP on oxidized Fe-oxy-hydroxy complexes (Boström *et al.,* 1982). Aerobic and anaerobic adsorption isotherms were well described by both a Langmuir (Equation 4.2) and Freundlich (Equation 4.3) expression of the form:

Equation 4.2 $$PS + NAP = (PSC \cdot Ce)/(k+Ce)$$

Equation 4.3 $$PS + NAP = d \cdot Ce^{f}$$

where:

PS = adsorbed P (mg/g dry wt)
NAP = native adsorbed P (mg P/g dry wt)
PSC = P adsorption capacity (mg P/g dry wt)
Ce = equilibrium concentration of P (mg/L)
k, d, and f = constants

Correlations and equation constants are shown in Table 4.3. A slightly higher phosphorus sorption capacity [PSC] was found for the Rochdale sediment. The result agrees with the slightly higher Fe and Al content of the Rochdale sediment sample (Table 4.1). In this case, native adsorbed P [NAP] was not calculated separately from the P sorbed [PS] equilibrium value. Compared to the aerobic PSC, anaerobic adsorption and PSC is much less, which would be expected due to reduced Fe complexing.

Table 4.3 Langmuir and Freundlich adsorption constants and correlation coefficients for the Rochdale and Lakewood pond sediments.

Pond	Langmuir equation constants			Freundlich equation constants		
	PSC	k	R^2	d	f	R^2
Rochdale (aerobic)	4.465	1.026	0.933	1.680	0.264	0.956
Lakewood (aerobic)	3.566	1.719	0.927	1.203	0.287	0.967
Rochdale (anaerobic)	0.252	1.650	0.929	0.079	0.474	0.806

4.3 Discussion

The sediment TP and measured range of interstitial P and N were on the high end of most values reported in the literature and, therefore, reflect the hypereutrophic state of these detention ponds. Nurnberg (1988), Pettersson (1984), and Boström *et al.* (1982) summarized a large number of literature data on sediment total-P, extractable fractions, and P release rates from lakes of varying trophic status. Median values for release rates in hypereutrophic lakes were in the range of 20 $mg/m^2/day$, but may be as high as 50 $mg/m^2/day$. Therefore, the laboratory and calculated release rates from the field data are in good agreement with those reported for other hypereutrophic systems.

The relatively small changes in the inorganic-bound P fraction show that seasonal P release in these sediments is driven primarily by mineralization of organic-P. The correlation of extractable inorganic-P fractions with sediment organic-P showed that these fractions fluctuate in equilibrium with the interstitial-P concentration. Variations in the redox state of surficial sediments will have the greatest effect on the magnitude of adsorption due to transition of Fe^{2+} - Fe^{3+} (Boström *et al.,* 1982). The sediment Fe:P ratios in the shallow ponds are present in excess of those reported to be necessary to suppress P release under oxidizing conditions (*i.e.,* 20:1 wt) (Danen-Louwerse *et al.,* 1993), which was demonstrated when intact cores were incubated under aerobic conditions sufficient to exceed the oxygen demand.

The aerobic PSC values calculated here were similar to that given by Danen-Louwerse *et al.* (1993). In that study, PSC of 0.08 ± 0.007 mol P (mol Fe + Al)$^{-1}$ at pH 8 was given as 90% confidence intervals from a relatively large data set. The PSC values calculated here were slightly higher at 0.11 ± 0.016 (Rochdale) and 0.09 ± 0.015 mol P (mol Fe + Al)$^{-1}$ (Lakewood) (pH buffered by fresh sediment ~7). Here the PSC was calculated from the total Fe and Al content of the sediment, while Danen-Louwerse *et al.* (1993) measured the Fe and Al liberated in sequential extraction solutions. Differences in pH and ionic strength of incubation media, and sediment Ca and Mn content may also introduce variation. Based on the measured range of P_{int} concentrations in Rochdale pond, the anaerobic isotherm agreed well with the range of adsorbed P measured by sequential extraction *(i.e.,* 100 to 150 µg P/g dry wt).

In the absence of an *in-situ* oxidized layer, increased pH of the overlying water may be of lesser significance to release rates in the non-stratifying ponds

(Anderson, 1975; Boers, 1991). However, Löfgren (1987) discussed the effect of fluctuating redox microzones at the sediment-water interface in shallow aerobic lakes, so that the significance of high pH on P release will vary accordingly. For the calcium bound fraction, higher temperatures and enhanced microbial metabolism may depress pH within the sediment profile, leading to dissolution of Ca-P matrices (Bostöm *et al.,* 1982). Boers and van Hese (1988) formulated a steady-state P release model in which they showed that, after calibration, varying the value of both their adsorption constant and precipitation rate constant had only marginal effect on the calculated flux rate. Sedimentation and mineralization rates were the most important terms. Furthermore, it was the amount of organic-P contained in the sedimenting material and not the bulk quantity that determined the flux response.

The field data collected in 1992 clearly showed that sedimentation rate, or at least the supply of organic-P, is highly variable according to the dynamics of the phytoplankton. One of the greatest difficulties involved in formulating a recycling model for sedimented material is the estimation of the sedimentation and mineralization rates, and particularly so in shallow systems that may be predisposed to resuspension. The situation may be further complicated by the fact that variable amounts of organic-P may be released *via* autolysis before reaching the sediment and that different algal groups may contain more or less recalcitrant P (Gunnison and Alexander, 1975).

In addition to the above uncertainties, further complications in predicting the P release rate are due to transient oxidation of the surficial sediment in the shallow ponds. The effect of redox shifts was evident in the 1995 data from Rochdale pond when P_{int} was drastically reduced in the upper sediment layers. Since water temperatures were above 20°C at this time, sediment oxygen demand would have been high. Clearly, transition is brought about by the balance between the sediment oxygen demand, the organic sedimentation rate, and the oxygen balance of the water column. The untypically low biomass levels which were sustained throughout the oxidation period (50 to 100 µg/L Chl *a*) were the result of alum treatment (see Chapter 7), and obviously the combination of both reduced water column respiration and organic sedimentation rate were a prerequisite to the condition. In this regard, a description of the oxygen budget in terms of photosynthesis and respiration is required.

A simple sediment sub-model for anaerobic release in shallow systems has been described and applied by Jørgensen (1980). The formulations of such a model are a useful starting point from which to describe temperature dependant P recycling in the sediment surface under anaerobic conditions. Sedimented organic-P is mineralized similarly to detritus in the water column *via* a first order reaction, with subsequent transfer to the interstitial fluid. In this case, increase in sediment depth is neglected, and a fixed active surficial pool is used. For the shallow Rochdale and Lakewood ponds, the pool mass present in the top 1 cm would be representative, where:

Equation 4.4
$$\frac{dP_e}{dt} = P_e net - k \cdot ([Pe] \cdot M) \cdot Q^{(T-20)}$$

where:

$P_e$net = the sedimentation rate of exchangeable P (mg/m^2/day)

[Pe] = the concentration of exchangeable P in the defined active layer (mg P/g dry matter)

M = mass of dry matter in the defined active layer

k = mineralization rate constant (0.008)

Q = temperature coefficient ($\approx$1.05), where T is temperature (°C)

In these SWDPs, an approximate annual net sedimentation can be calculated since the ponds are of known age. Therefore, an average daily sedimentation rate of P could be estimated along the same lines by taking the fraction recycled (F) into consideration. However, since the ponds operate as intermittent sedimentation basins, this approach is inappropriate. From the estimates of annual sediment deposition given in Table 4.1, very approximate daily volumetric sedimentation rates (240 day operational season) for Rochdale, Lakewood, and Lakeview ponds would be: 0.04 mm/day, 0.03 mm/day, and 0.06 mm/day, respectively.

Normally the phytoplankton loss rate *via* sedimentation during dry weather would be low, and ranges of 0.02 to 0.2 m/day are typical values (Lijklema and Peeters, 1979). Buoyancy controlled blue-greens will undergo lower loss rates than diatoms and other species. Van Straten (1986) calculated a net sedimentation rate for PP in the shallow Lake Veluwemeer of 0.02 m/day, a low value on account of wind induced resuspension. The sediment surface organic matter variation described here showed that sedimentation rates are highly variable, as is resuspension. Clearly under unfavourable physiological conditions, such as increasing nutrient limitation, massive algal death may occur. Channelling of detritus production rate through two flows - one for the labile fraction and the other for the resistant fraction, with the degradable sedimented fraction being the difference between the sedimentation and water column mineralization rates, is a typical approach in modelling of nutrient cycles (Jorgensen *et al.,*1986).

The organic-P mineralization constant estimated from the 1992 field data of 0.008/day is similar in magnitude to that employed by Boers and van Hese (1988) in their steady-state mineralization model (0.006/day). Therefore, this may be a useful starting point for mineralization at the aerobic/anaerobic interface. Subsequent release of mineralized P (and N) from the sediment would depend on establishing criteria for a concentration gradient with appropriate diffusion coefficient and adsorption constant. For simplicity, direct release according to mineralized mass in the active layer could be assumed under anaerobic conditions. Lakeview pond P_{int} gradients indicate that diffusion from deeper layers may be important. Therefore, the measured P release rates may be considered typical as net internal loading values. The extent to which nitrate reduction enhances the mineralization rate in the hypolimnion is not known, but groundwater flows measured in 1994 and 1995 suggest this may be significant (see Chapter 5).

4.4 Conclusion

The sediments in the SWDPs are slightly different in character and reflect physical differences between the ponds and their respective catchment areas. Compared to the shallow Rochdale and Lakewood ponds, sediments from the 2.7 m deep stratifying Lakeview pond are deeper, lower in iron, and higher in TP and organic content. In comparison to the neighbouring Rochdale pond, Lakewood pond has a higher exchangeable P pool and slightly lower P adsorption capacity under aerobic conditions, which supports observations of the generally higher TP levels measured in the water column of this pond. The anaerobic release rates calculated from field and experimental data correspond well with ranges reported in the literature for hypereutrophic sediments. The current potential for internal loading of P represents a significant obstacle in management efforts to reduce phytoplankton biomass without sediment removal.

Chapter 5

NUTRIENT LOADING AND POND OPERATIONAL PARAMETERS

5.1 Introduction

This chapter presents the results of stormwater runoff loads to Lakeview pond in 1994 and 1995. These data comprise both measured and modelled loads that were obtained after calibration of the runoff block of the stormwater management model, PC-SWMM (Raymond, 1997; Raymond *et al.,* 1995). Further data related to water residence times and pond operational performance for average P removal efficiencies, according to the model of Walker (1987), are examined in the light of regional rainfall statistics and pond design characteristics.

The results of sediment traps used in Lakeview pond during the 1994 and 1995 seasons are given, and estimates of phytoplankton community productivity during selected periods have been made.

5.2 Results

5.2.1 Theoretical phosphorus removal performance

Eight years of tipping-bucket rain gauge data were examined for the period May to September, 1988 to 1995. Precipitation for Regina was recorded by Regina Airport (AES, Canada) located close to the ponds in the Northwest sector of the City. Saskatoon data was obtained from the Saskatchewan Research Council [SRC] Climate Reference Station, located about three miles from the Lakeview pond catchment. Precipitation events were grouped as <5 mm, 5<10 mm, and >10 mm events. Events were regarded as discreet if they were separated by more than six hours. Events of 5 mm or more were used in the statistical calculation for pond operational characteristics. It was felt that these data would better represent long-term performance, since the bulk of the water and nutrient budgets derive from these larger events. Depressional storage and lower particle washoff intensity of the <5 mm category may further justify their exclusion from statistical calculations.

Regina rainfall characteristics were different to those of Saskatoon. In the Regina area, 50 to 70% (mean - 60%) of the total seasonal precipitation (includes all numbers down to trace) was supplied in events >10 mm, except in 1992, which was very dry and recorded only one event >10 mm for the period. When combined with the >5 mm category, the bulk of the total seasonal precipitation was obtained (mean = 80%).

In the Saskatoon area, lower average seasonal precipitation was measured over the eight year record compared to Regina. Slightly more of the total precipitation occurs in the <5 mm category. Events >10 mm ranged from 30 to 55% of the total (mean = 41%), and events >5 mm ranged from 10 to 36 (mean 22%). Data input used to generate P removal performance curves for the ponds is given in Table 5.1. Equations used to calculate the mean P retention coefficient are as given in Chapter 1 (Equations 1.3, 1.4, and 1.5).

Table 5.1 Pond design and operational factors used to compute theoretical phosphorus removal curves. Precipitation statistics obtained for Regina and Saskatoon from independent regional records, May to September, 1988 to 1995.

Pond and Watershed Characteristics	Rochdale	Lakewood	Lakeview
A_w = watershed area (ha)	90	68.4	78.4
r_c = watershed runoff coefficient	0.4	0.4	0.4
A = pond surface area (ha)	2.1	2.1	2.2
Z = pond mean depth (m)	1.73	1.83	2.7
Precipitation Characteristics	>4.9 mm	>4.9 mm	>4.9 mm
P_m = mean storm size (cm)	1.57 (n=111)	1.57 (n=111)	1.12 (n=98)
T_e = mean time between event mid-points (hrs)	255	255	300
T_d = mean storm duration (hrs)	5.69	5.69	5.74
P_t = total seasonal precipitation (cm)	22.5	22.5	14.0
T_t = length of season (years)	0.416	0.416	0.416
Watershed Runoff			
V_m = mean storm runoff volume (m^3)	5634	4282	3522
V_t = total seasonal runoff volume (m^3)	81000	61560	43904
Pi = mean inflow TP concentration (mg/m^3)	650	650	650
Fo = mean DIP:TP ratio in inflow	0.3	0.3	0.3
Pond Performance Indicators			
V_{rel} = pond relative volume (cm)	10.09	14.05	18.94
Q_m/A = surface overflow rate during mean storm (cm/hr)	4.72	3.58	2.79
V_p/V_m = pond volume/mean runoff volume	6.45	8.98	16.87
T = mean hydraulic residence time (years)	0.187	0.260	0.563

Due to a lack of runoff data for the Regina pond catchments, the mean TP (Pi = inflow TP concentration) value used by Walker (1987) of 650 µg/L and DIP:TP ratio (Fo) of 0.3 was initially assumed. The overall average concentration of all measured and modelled TP inputs to Lakeview was about 40 to 50% of the above value, while the mean DIP:TP ratio was about 0.4 to 0.5.

Most of the storms measured during model calibration were relatively small, with short inter-event buildup periods; and loads associated with larger events and longer dry weather periods still require to be investigated. It is very likely that loads generated by larger events with longer inter-event buildup periods may be substantially higher. Two stormflow events entering Rochdale had TP and DIP mean concentrations of 0.406 and 0.168 mg/L (N=11, June 23, 1993 - 10 mm precipitation after seven days dry weather); 1.42 and 0.927 mg/L (N=24 June 30, 1994, 45 mm after two days dry weather). Hydrographs were not measured and so the true EMCs are not known. Relative to other uncertainties in the P model parameters, the value of 650 µg/L TP is deemed acceptable. Furthermore, approximate 95% confidence limits were estimated by varying Pi and Fo by a factor of 2 in each direction in the equations.

The average runoff coefficient was based at 0.4 for all ponds, on account of the portion of the record used to obtain the rainfall statistics. Raymond (1997) calibrated PC-SWMM with a runoff coefficient of 0.2 for the smaller measured events to Lakeview. However, the larger events drawn from the rainfall record will produce higher runoff coefficients, so that a value of 0.4 may be assumed. As with the variation in the Pi and Fo factors, the effect of altering total seasonal precipitation by a factor of 2 was examined, which is equivalent to altering the runoff coefficient. Total-P removal curves, indicating the mean hydraulic residence time and pond relative volume calculated from the rainfall record and physical characteristics, are shown in Figure 5.1 a,b.

Walker (1987) states that about 95% of the model errors in the model development data set could be accounted for by altering the P decay coefficient (K2) by a factor of two (*i.e.,* doubling or dividing by 2). The results of doing so produce a confidence range in the same order as those obtained by equivalent variation in both Pi and Fo. The effects for the Rochdale pond are shown in Figure 5.3. Altering the mean seasonal rainfall volume has the least effect on removal. Therefore, theoretical TP removal efficiencies are: Lakeview, 72 to 85%, mean 79%: Rochdale, 63 to 79%, mean 72%; Lakewood, 65 to 81%, mean 74%.

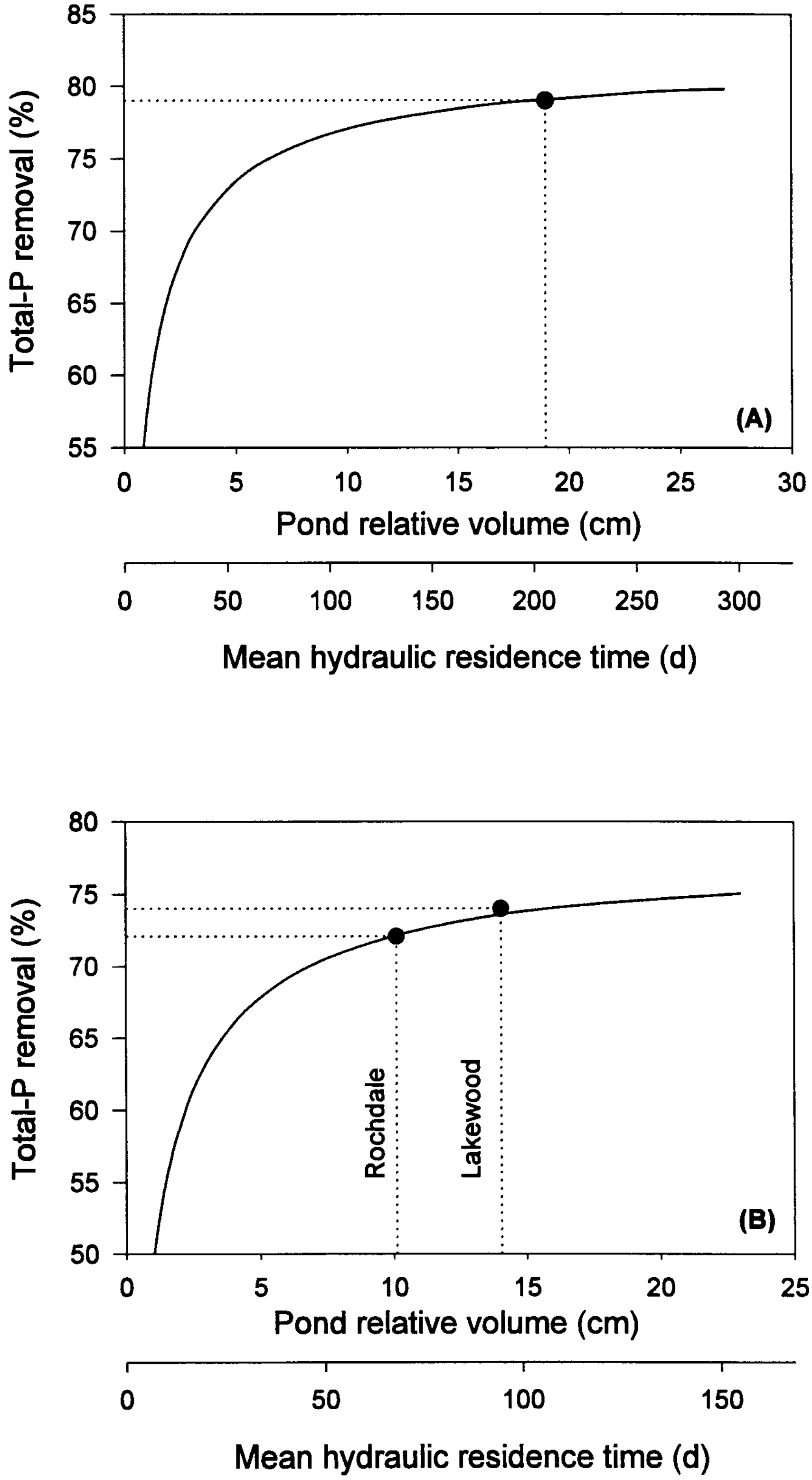

Figure 5.1 **Theoretical total phosphorus removal curves for Lakeview pond (A) and for Rochdale and Lakewood ponds (B).**

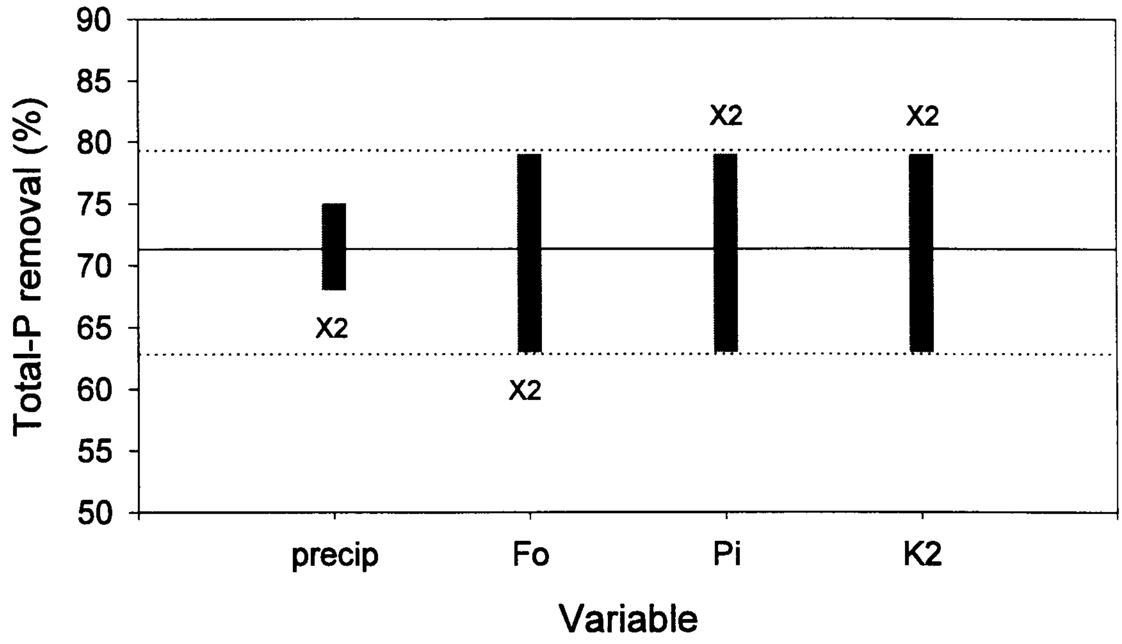

Figure 5.2 Example sensitivity analysis of variables used to generate total phosphorus removal curve for Rochdale pond. (Fo = DIP:TP ratio of inflow; Pi = TP concentration of inflow; K2 = P decay coefficient)

5.2.2 Volumetric and nutrient loads to Lakeview stormwater detention pond (Saskatoon), 1994 and 1995

While the loads of both nitrogen and phosphorus are of interest, P is ultimately the conservative element that must be controlled in order to reduce the phytoplankton biomass. In the absence of outflow data, there remains considerable uncertainty with regard to the N budget. The storm volume, mass and areal P loadings to the Lakeview pond for the period May to September, 1994 and 1995 are shown in Figure 5.3 a,b,c,d. In both years, the bulk of inflow volumes produced exchanges of less than 10% of the pond volume. In 1994, the largest precipitation event occurred in May. The intensity and duration of this storm resulted in high washoff from the catchment producing a load of close to 80 mg TP/m^2. The four events that closely followed gave minor nutrient loads due to the short inter-event duration. Total-P loads for the remaining summer months were mostly in the range of 5 to 20 mg/m^2/event, with DIP between 30 and 55% of TP load. There were no events in September. The pattern of P loading in 1995 was quite similar to that described for 1994, except that May had very little precipitation. Particle buildup factors throughout May resulted in a high load to the pond at the beginning of June from an exchange volume of only 5%.

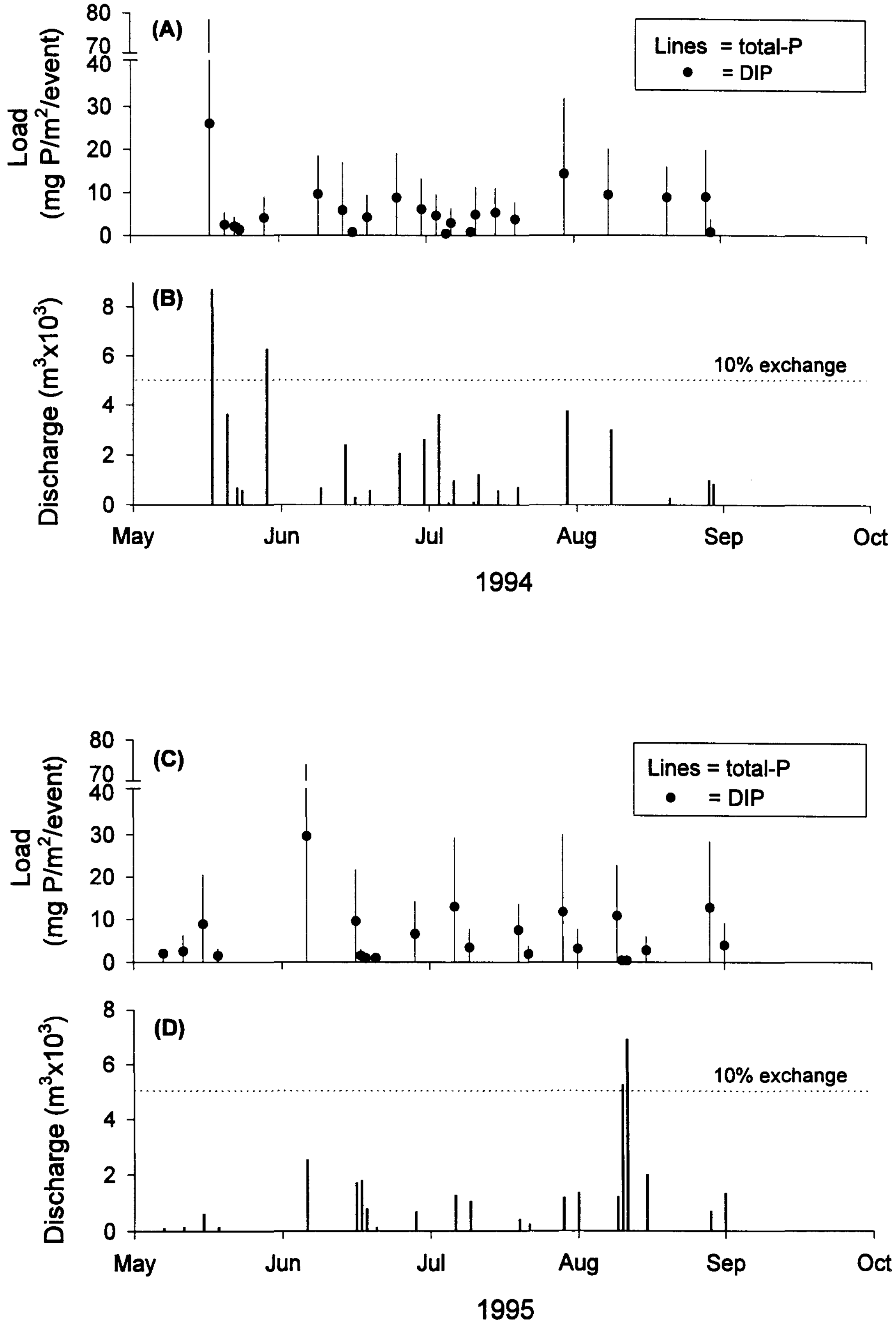

Figure 5.3 Areal phosphorus loading and runoff volume to Lakeview pond in 1994 (graphs A and B), and 1995 (graphs C and D).

Modelled mass nutrient and measured volumetric loads to the Lakeview pond for 1994 and 1995 have been summarized on a monthly basis in Table 5.2.

Table 5.2 Summarized monthly external loading data to the Lakeview stormwater detention pond (Saskatoon) generated by surface runoff. (no runoff was recorded in September of 1994 or 1995)

Month 1994	**Total Discharge (m^3)**	**TSS (kg)**	**TP (kg)**	**PO_4-P (kg)**	**TKN (kg)**	**NO_3-N (kg)**	**NH_4-N (kg)**
May	20062	2496	2.27	0.84	15.1	2.63	0.43
June	8714	1327	1.72	0.77	8.3	1.99	0.31
July	11187	1294	1.75	0.81	7.9	1.98	0.3
August	5146	811	1.31	0.61	5	1.7	0.21
Oct	2755	1195	2.52	1.13	6.6	3.33	0.3
Σ 1994	**47864**	**7123**	**9.57**	**4.17**	**42.94**	**11.62**	**1.55**
April	3648	580	1.17	0.56	3.4	1.58	0.54
May	1053	455	0.73	0.33	2.8	0.92	0.4
June	7740	2470	2.54	1.08	15.6	2.7	1.76
July	5581	1695	2.05	0.9	10.6	2.19	1.26
August	17536	1156	1.51	0.69	7.1	1.77	0.86
Oct	8321	1836	2.49	1.11	11.3	3.09	1.43
Σ 1995	**43879**	**7611**	**9.32**	**4.11**	**47.3**	**10.67**	**5.71**

For the period April to October, the stormwater volumetric loads resulted in water residence times of 230 days in 1994 and 250 days in 1995. These value are close to the value of 205 days calculated in Table 5.2. Event mean concentrations for dissolved nutrients were frequently lower than the pond background concentrations (dry weather internal loading), so that negative DIP retention coefficients may result: As an average of all storm events, approximately 50% of the TP in the inflows was in particulate form. Smaller storms will result in higher particulate retention efficiencies, so that although external loads may result in a net gain to the system as a whole, settling of particulates produces a net reduction of the ambient TP. Unfortunately, a shortage of suitable automated sampling equipment meant that outflows could not be measured.

Groundwater baseflow rates were measured in 1994 and 1995. The associated nutrient concentrations were measured in 1994 only between May and September. Flow rates in both years were similar, with flows tending to be slightly lower in the spring and autumn. Average flow rates in the north catchment storm sewer entering the pond were 0.0048 m^3/sec (Raymond, p. comm.), which would replace about 0.8% of the permanent pond volume per day. Average nutrient concentrations are given in Table 5.3.

Table 5.3 Groundwater baseflow nutrient concentrations entering Lakeview stormwater detention pond (Saskatoon) in 1994. (Values are as mg/L)

Parameter	PO_4-P	NO_3-N	NH_4-N	Alkalinity	pH	TDS
mean (n=15)	0.050	8.4	0.019	509	7.56	2809
s.d.	0.028	1.1	0.015	46	0.90	305

High values measured for alkalinity and TDS suggest the source of the baseflow to be groundwater. The relatively high NO_3 concentrations are most likely derived from agricultural leachate during aquifer recharge. Phosphate varied quite widely, suggesting that additional sources to the storm sewers may be involved, such as runoff from irrigated lawns. Based on the average flow rate, the baseflow would be loading a substantial or dominant part of both the annual nitrogen and phosphorus budget. The relative contributions of the seasonal nutrient inputs for the period when baseflows concentrations were measured are summarized in Table 5.4.

Table 5.4 Nitrogen and phosphorus inputs to the Lakeview stormwater detention pond (Saskatoon), April to October 1994. (Internal loading is estimated from anaerobic sediment core incubations 5 to 20°C and average interstitial N:P ratios)

Parameter	TP	PO_4-P	NO_3-N	NH_4-N	TKN
Σ storm loads (kg)	9.57	4.17	11.62	1.55	42.94
Σ baseflows (kg)	5.32	5.32	745	1.68	nd
mass load (kg)	14.9	9.5	756	3.23	42.94
averaged external load mg/m^2/day	3.17	2.02	160	0.69	9.12
internal load (gross) mg/m^2/day	5 - 15	5 - 15	0	15 - 45	nd

nd = no data

Baseflows are clearly supplying a very large amount of nitrate to the system, while the total seasonal load of DIP may also be dominated by baseflows. Assuming an anaerobic hypolimnion of 1 m, and that baseflow loads have a uniform distribution throughout the hypolimnion during stratification, the amount of nitrate-oxygen available for denitrification-based oxidation processes would be 880 mg O_2/m^2/day. Air saturation of the baseflow would supply a further 150 mg O_2/m^2/day to the hypolimnion. The predominantly anaerobic conditions will result in a large part of the nitrate being lost from the system *via* denitrification (*e.g.,* Stewart *et al.,* 1982). These findings support data given in Chapter 3, which showed that nitrate levels were frequently found to be higher in the anaerobic bottom water. It is not clear what the fate of the 400 m^3/day baseflow was, but it was certainly sufficient to replace evaporative losses (~5 mm/day, or 100 to 150 m^3/day, as an average from May to September, AES Canada, small water body evaporation data, 1992). Although elevations greater than the weir height were not recorded by the sonic probe during dry weather, trickle outflows may have occurred on account of an imperfect seal around the steel plate fitted at the start of 1994 for modification to a sharp crested weir. The only other explanation is that re-routing *via* infiltration was occurring. A tracer study may provide an insight into the distribution of the baseflow within the pond. Internal loading of P dominated the estimated DIP loads from both stormflows and groundwater flows as a daily average, whereas the bulk of the N is derived from the groundwater. Pulse loading by stormflows will shift the relative contributions during discreet time periods.

5.2.3 Lakeview sedimentation traps and algal productivity

During part of 1994 and 1995, sediment traps were used to intercept settling particulate material. In 1994, traps were usually replaced at weekly intervals, whereas in 1995 a less frequent monitoring program meant that traps were often in place for several weeks or more. There is a great deal of uncertainty with regard to mineralization within the traps, as well as the influence of resuspension. For this reason, only the results obtained during weekly exposure times were assumed to be useful. Short exposure times isolate samples affected by stormflows and reduce the changes of ongoing mineralization. Nevertheless, longer exposure times may provide information on the net supply to the sediments, but significant errors in gross sedimentation estimates will result from mineralization.

Table 5.5 summarizes the 1994 and 1995 data. The range of nutrient concentrations measured in the traps was similar between the two years, and tended to be highest in mid-summer (conceding mineralization in late summer samples with longer trap exposure times). It was notable that the lower C sedimentation rates during September of both years occurred during a month of no runoff, as shown previously in Figure 5.3. Therefore, no resuspension or interference by external loading should have affected these samples.

Although phaeophytins were not measured, the consistently low chlorophyll *a:b* ratios in the traps indicate the higher sedimentation rate of green algae. There were several occasions when external suspended solids (mineral) and probable resuspension clearly affected the traps (indicated by increases in the SS:C ratio beyond that of the seston: *e.g.,* July 6, 1994 and August 11, 1994). In an effort to isolate the proportion of trap material which may have derived from phytoplankton, the following approach was used: the ratios of SS:PC:PN:PP were obtained in top and bottom water samples. The ratio of SS:PC:PN in bottom samples (average of value measured at start and end of each period) was applied to the traps and the corresponding amount subtracted from the trap total. The difference between top and bottom water C:N ratio on each sampling date was plotted against temperature (Figure 5.4). Although the results predictably showed a large variance, the data indicated that N was more rapidly released compared to C upon autolysis and initial mineralization. This process has been similarly described by Ulén (1978) and Andersen and Jensen (1992). A somewhat arbitrary zero to 15% loss of PN between top and bottom water seston was estimated from Figure 5.4. Thus, carbon in the trap derived from phytoplankton was allowed to vary from the bottom water SS:C:N ratio of the seston (minimum) to the same ratio at 15% more N. Based on the C:N:P ratio of the top water seston, a substantial deficit in the trap PP was evident. For the shorter time periods, the calculated deficit varied from 30 to 70%, a fact that further indicates the uncertainties in estimating the fraction of trap material of algal origin. The average C:P ratios of the trap material were similar to those measured in the surface sediment (≈65:1, Chapter 4).

Table 5.5 **Parameter concentrations measured in Lakeview pond sedimentation traps during 1994 and 1995, with estimates of trap carbon deriving from phytoplankton.**

Date Emptied	Number of Days Elapsed	PC $mg/m^2/day$	Estimated % of Trap PC of Algal Origin	PN mg/m^2 /day	PP mg/m^2 /day	C:N Ratio	N:P Ratio	Chl *a* mg/m^2 /day	Chl *b* mg/m^2 /day	Chl *a:b* Ratio	Susp. Solids $g/m^2/day$	SS:C Ratio (g/g)
30-June-94	start											
6-July-94	6	3055	35 - 40	188	12	16	16	22	52	0.4	30.6	10.1
12-July-94	6	5630	85 - 97	791	28	7	29	204	513	0.4	32	5.7
21-July-94	9	4614	86 - 99	681	57	7	12	191	458	0.4	23.6	5.1
28-July-94	***											
5-Aug-94	7	3362	92 - 100	534	17	6	31	261	624	0.4	12.8	3.8
11-Aug-94	6	7152	61- 70	773	44	9	18	163	306	0.5	41.8	5.8
18-Aug-94	7	3560	80 - 92	475	18	8	27	93	180	0.5	15.1	4.3
7-Sept-94	20	2077	72 - 83	249	8	8	31	95	215	0.4	11.8	5.7
28-Sept-94	21	1483	91 - 100	270	9	6	29	63	148	0.4	6.1	4.1
17-May-95	start											
13-June-95	27	2564	87 - 100	374	8	7	46	102	218	0.5	10.8	4.2
21-June-95	8	10383	62 - 71	1012	27	10	37	238	438	0.5	57.8	5.6
5-July-95	14	5488	97 - 100	895	41	6	22	342	509	0.7	18.2	3.3
20-July-95	15	5745	77 - 88	757	29	8	26	312	500	0.6	28.3	4.9
27-July-95	7	7169	97 - 100	1019	61	7	17	546	1085	0.5	23.6	3.3
17-Aug-95	21	5093	77 - 88	590	31	9	19	279	560	0.5	26.4	5.2
14-Sept-95	28	2175	77 - 86	274	11	8	25	107	222	0.5	10.8	4.9

*** equipment tampered with - no data

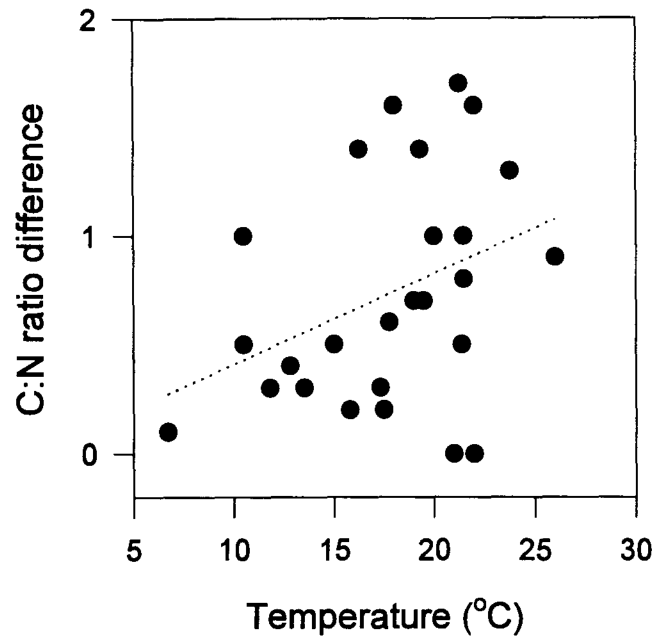

Figure 5.4 Particulate carbon:nitrogen ratio difference between top (0 to 1.5 m integrated) and bottom (2.6 m) water seston in Lakeview pond (Saskatoon), 1994.

Predicting mixed layer chlorophyll and carbon sedimentation: The standing crop of phytoplankton and the species composition are the result of the balance between gain processes (growth and recruitment) and loss processes (sedimentation, grazing, death, and washout). The amount of algae residing within the euphotic zone is related to the ratio of the euphotic zone depth to the mixed layer depth and the areal standing crop of chlorophyll in the mixed layer. In turn, the euphotic zone depth is dependent on the total extinction coefficient and the intensity of the surface irradiance. The relationship may be described by the expansion of the Lambert-Bougier Law given in Chapter 3 (Forsberg and Shapiro, 1982):

Equation 5.1

$$\frac{Z_e}{Z_m} = \frac{\ln(I_o \,/\, I_{ze})}{(a\varepsilon_c)/Z_m + \varepsilon_w} \bullet \frac{1}{Z_m} = \frac{4.6}{a\varepsilon_c + \varepsilon_w \bullet Z_m}$$

where:

I_o = surface irradiance (W m^{-2})

I_{ze} = irradiance at the depth of the euphotic zone (W m^{-2})

ϵ_c = partial extinction coefficient for chlorophyll (m^{-2} mg Chl a^{-1})

ϵ_w = residual extinction coefficient of the water (m^{-2})

a = areal standing crop of Chl *a* in mixed layer (mg Chl m^{-3})

Z_e = depth of the euphotic zone (m)

Z_m = depth of the mixed layer (m)

Gross and net algal growth were estimated for the period June 30 to August 18, 1994, during which sedimentation traps were operated at short term intervals. Productivity was calculated by working with the community as a C pool:

Equation 5.2 $$R = B - D$$

where:

R = carbon specific growth rate (day^{-1})

B = carbon specific production rate (day^{-1})

D - carbon specific loss rate (day^{-1})

For the purposes of assessing B in Equation 5.2, the daily integral rate of photosynthesis, (mg C m^{-2} day^{-1}) was set to the area of a rectangle with one side equal to the maximum daily volumetric rate of photosynthesis (p-max; mg C m^{-3} day^{-1}, redefined as mg C mg^{-1} Chl *a* day^{-1}), and the other side equal to the depth (Z') at which the irradiance half-saturation constant (I_k) for the phytoplankton community was achieved (Talling 1957). The approach assumes that irradiance is sufficient to cause photosynthetic inhibition in the upper layer, and that the mixed layer is greater than the depth of the euphotic zone. The parameter Z' was derived empirically from I_o, and I_{ze} for the time periods by selecting an average value of light saturation (I_z) for blue-greens and greens from literature values. The mixed layer was taken as the mid-point on the steep inflection of the thermal gradient (1.5 to 2.0 m).

Forsberg and Shapiro (1982) represented nutrient dependence with a Droop type term, which related light saturated specific rate of photosynthesis to a minimum TP subsistence quota per unit of chlorophyll:

Equation 5.3 $$P\text{-max} = P\text{-max}^{sat} \bullet \left(1 - \frac{Kq}{TP/c}\right)$$

where:

$P\text{-max}^{sat}$ = the maximum daily specific rate of photosynthesis under conditions of saturating nutrients (mg C mg Chl^{-1} day^{-1})

Kq = the subsistence quota of TP or the ratio TP/c, at which the value of P-max is zero (mg TP mg Chl^{-1})

Dissolved inorganic phosphorus was generally present in excess of 20 μg/L, so that if an average K_s of 10 μg/L is assumed, the phytoplankton were P sufficient (Tilman *et al.*, 1982). Nitrogen may have been periodically limiting, but the phytoplankton was dominated by greens for most of the 1994 period in which the productivity estimates were made. N-fixing *Anabaena* sp. began increasing toward the end of the period, but greens continued to dominate. The transition period may therefore have involved a degree of N limitation. Relative to other uncertainties no account of potential nutrient limitation was included (*i.e.*, Equation 5.3 = 1). Kq for P would require to be measured.

B was determined by dividing the daily integral rate of photosynthesis by the areal standing crop of carbon in the mixed layer: Θ•Zm•C (mg C m^{-2}), where Θ is the C:Chl ratio of the phytoplankton (mg C mg Chl^{-1}).

The community carbon/chlorophyll ratio and relative group biomass was estimated using cell volume determined from counts. While total seston carbon was measured, the influence of detritus is unknown, so that POC was estimated assuming a cell density of 1 and 50% of the cellular dry weight as C (Vollenweider, 1985). Average dry weight for both greens and blue-greens was assumed to be about 50% (Reynolds, 1984). The difference between seston and cellular POC was assumed to represent the momentary detrital pool. Chlorophyll *a* alone was used to calculate the C:Chl ratios, which were generally between 35 to 45. The complete growth model was:

Equation 5.4

$$R = \left[\frac{\ln(I_o/Iz')c\ P\text{-}max^{sat}}{c\varepsilon_c + \varepsilon_w}\right] \bullet \left[1 - \frac{Kqc}{TP}\right] \bullet \left[\frac{1}{Zm\Theta c}\right] - D$$

For the period estimated, water temperatures were between 18 to 23°C, so no temperature corrections were used to modify either the specified growth rates, or the endogenous respiration rates.

The sum of the various loss mechanisms in D did not include zooplankton grazing, since the population was composed almost entirely of raptorial *Ancanthocyclops vernalis*, and juvenile cyclopoids (Brandl and Fernando, 1975). Mortality produced detritus, which was mineralized according to a first order reaction during residence time in the mixed layer (Jewell and McCarty, 1971;

Ulén, 1978). The amount of detritus in the mixed layer without resuspension was described by:

Equation 5.5 $$M = (M_o - fM_o)\ e^{-k't} + fM_o - (M_s)t$$

where:

M = mass of detritus at time t+1(m^{-2})
Mo = mass of detritus at time t (m^{-2})
Ms = detritus sedimentation loss rate (t^{-1})
f = fraction resistant to mineralization (-)
-k' = rate constant for aerobic mineralization of detritus C

Sedimentation losses of detritus and intact cells were described with the following equation (Reynolds, 1984):

Equation 5.6 $$N_t = N_o\ e^{(1-v'/Z_m)^t}$$

where:

Nt = standing crop remaining at time t+1 (m^{-2})
No = standing crop at t=0 (m^{-2})
v' = sinking rate (m day^{-1})
Zm = mixed layer depth

The parameters used in the assessment of the various components of Equation (5.4) are summarized in Table 5.6. Irradiance data was obtained as Global Solar Radiation from the SRC's Climate Reference Station, about three miles from the site. The integrated daily irradiance was assumed to comprise 43% PAR 400 to 700 nm, daily reflection was 0.1, and a full rate of photosynthesis was assumed to occur from sunrise to sunset minus 3 hours. Attenuation coefficients were calculated from Secchi disk readings given in Chapter 3. Averaged parameter ranges for green and blue-green algae were estimated from literature sources (Boers, 1991; Carr and Whitton, 1982; Reynolds, 1984; Seip and Reynolds, 1995). Where appropriate, the values given below were adjusted linearly according to the species composition obtained from the group cell volumes. No account of biomass losses through flushing was made on account of the relatively minor exchanges involved during the period.

Table 5.6 Parameters used to predict mixed layer chlorophyll concentration and daily sedimentary losses.

Parameters	Units	Blue-greens	Greens
optimum light intensity	W m^{-2}	23	40
chlorophyll extinction coefficient	m^{-1}/mg Chl *a*/m^3	0.016	0.016
background extinction coefficient	m^{-1}	1.2	1.2
maximum growth rate	day^{-1}	1.7	2.25
respiration	day^{-1}	0.05	0.2
sinking rate live cells	m day^{-1}	0	0.5
sinking rate detritus	m day^{-1}	0.75	0.75
mortality	day^{-1}	0.05	0.05
aerobic rate constant for detritus mineralization in Zm	k'	0.05	0.05
fraction degradable		0.7	0.6

The specified parameters in Table 5.6 and the light limited model satisfied the trends for chlorophyll *a* reasonably well. However, chlorophyll was slightly underestimated on the last two dates, during which *Anabaena* sp. was increasing, so the maximum photosynthetic yield of blue-greens was adjusted in the model by 20%. The justification was that buoyancy control resulted in a higher proportion (20%) of blue-greens resident in the light saturation zone than the specified "homogenous" mixed layer otherwise suggested. The measured and predicted chlorophyll values for the period are shown in Figure 5.5.

The calculated gross and net C productivity are shown in Figure 5.6. The integrated net productivity gave an average over the 49 days of 2.5 g C/m^2/day or 6.7 g O_2/m^2/day. For the most part, the daily areal mass sedimenting to the hypolimnion was lower than the estimates obtained from the traps (Table 5.7). Externally loaded particulates, sediment resuspension, and the interception of debris deriving from zooplankton and fish, *etc.* are all factors that complicate the interpretation of differences (some runoff entered the pond on all weeks except the last, as shown previously in Figure 5.3). The calculated chemical oxygen demand (1 mg organic material = 1.42 mg O_2, Jewel and McCarty, 1971) associated with the estimated organic sedimentation was about 75% greater than oxygen input to the hypolimnion from baseflows described above (assuming 30% of the material was refractory).

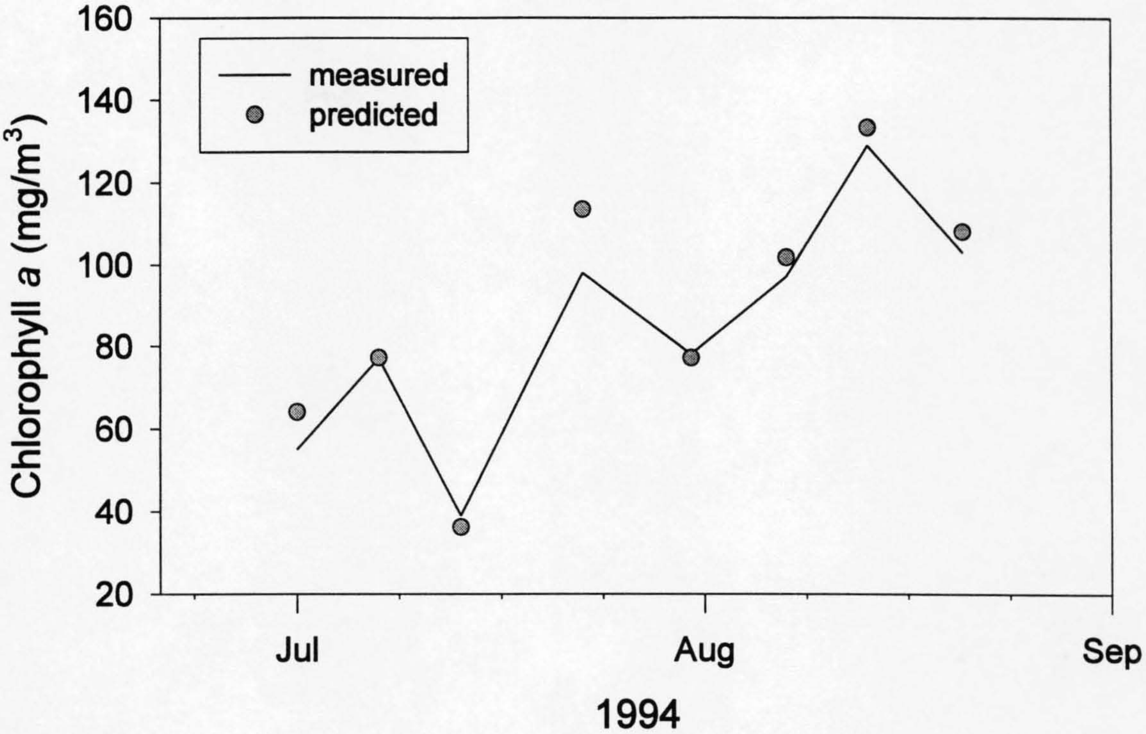

Figure 5.5 Measured and predicted chlorophyll *a* in Lakeview pond (Saskatoon), June 30 to August 18, 1994.

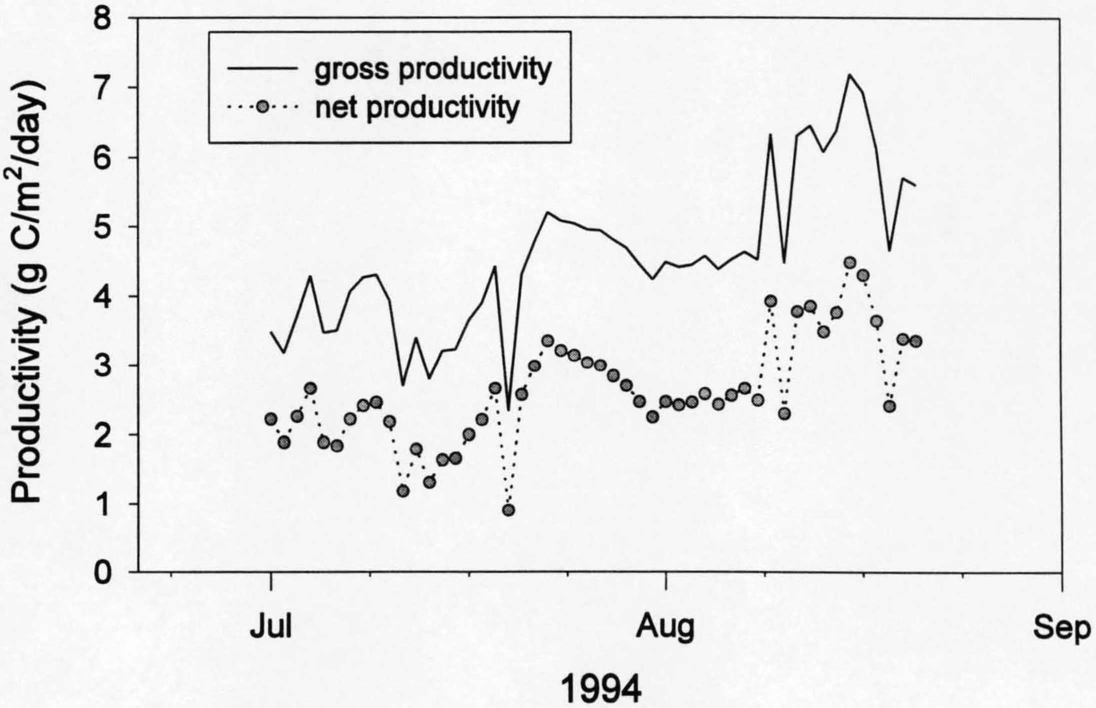

Figure 5.6 Estimated gross and net phytoplankton productivity in Lakeview pond (Saskatoon), June 30 to August 18, 1994.

Table 5.7 Comparison of sedimentary loss of organic-C from the mixed layer with that estimated from sediment traps in Lakeview pond, June to August, 1994. (mg/m^2/day and calculated chemical oxygen demand)

Date	Estimated Trap PC of Algal Origin	Predicted POC Sedimenting From Mixed Layer	COD ($gO_2/m^2/day$)
July 6	1096 - 1222	2168	6.16
July 12	4786 - 5461	2127	6.04
July 21	3968 - 4568	2179	6.18
August 5	3093 - 3362	2595	7.37
August 11	4363 - 5006	3356	9.53
August 18	2848 - 3275	2935	8.34

5.3 Discussion

All three detention ponds have V_{rel} values, which bring their theoretical P removal efficiencies well up onto the flatter portion of the TP removal curves, and especially Lakeview pond. For this reason, the sensitivity analysis showed that only relatively small changes in the efficiency resulted from variation of factors such as rainfall and the inflow TP concentration by a factor of two (Walker, 1987). All three ponds now have significant sediment reserves of P and may undergo substantial accumulation of P in the water column through extended periods of internal loading. As discussed in Chapter 4, about 50% of the sedimented P pool is ultimately recycled. Clearly, exchanges involving flushing of high [P] pond water will significantly reduce the theoretical efficiency of the detention ponds for net P retention in the long-term. Nevertheless, their potential as an environmental protection feature is clear and the theoretical analysis is of particular interest to the effect of linking ponds in series.

The role of groundwater flow to the water and nutrient budget of Lakeview pond suggests that fluctuations in the height of the water table may result in significant differences from year to year, especially in the load of inorganic N. Relative to the existing internal load of P, and periodic stormflow loading, it is questionable whether interception of P arriving in the baseflows would make much difference to the productivity of the system. A basic Vollenweider P loading plot classifies the system as highly eutrophic in both cases.

Considering the seasonal mass loads of P to Lakeview pond in 1994 and 1995, mass loads per hectare of impervious catchment calculated to about

0.27 kg TP/ha/0.58 yr (~40 to 50% dissolved), with seasonal precipitation of about 250 mm (about 50% of the catchment is impervious). This value lies within similar ranges to those reported in the NURP study (Athayde *et al.,* 1983) for low density residential catchments (see Chapter 1, Table 1.2), although variation in the mass export may be quite different in wet and dry summers.

Sedimentation traps have been regarded as having the potential to provide a realistic measure of sedimentation rates provided they are used in systems with little turbulence (*e.g.,* Bloesch and Burns, 1980; Reynolds, 1984; Trimbee and Prepas, 1984). Attention has also been drawn to the sediment "focusing" effects of lake morphometry when assuming the areal mass deposition, and to the aspect ratio of the traps themselves. Their use in Lakeview pond was somewhat questionable on account of the way the system operates. In the absence of detailed measurement of the phytoplankton productivity, it is impossible to estimate to what extent resuspension or the external load affected the traps. Specification of higher maximum growth rates and increased sinking rates would have increased the predicted sedimentation rate, but the chosen upper limit of the sedimentation rates seemed reasonable according to literature values. Productivity should be measured and calibrated for these systems. Furthermore, the extent of mineralization was unknown. Uehlinger and Bloesch (1987) calculated that less than 10% C and P mineralization occurred in traps exposed at four day intervals (aerobic).

Cappenberg and Verdouw (1982) used the sum of the PN plus the TDN in the water in their traps to calculate C flux based on the seston ratio. Mineralization rates estimated for organic-C and P in the sediment surface (Chapter 4) should theoretically apply to the traps, but in view of the various uncertainties the process was ignored.

Lake models, such as those used to predict the course of lake restoration, or biomanipulation measures need precise information on the links between the C, N, and P pools and their transformation rates. A number of key parameters need to be evaluated in order to link these rates. These would include $P\ max^{sat}$ and nutrient limitation parameters of the dominating phytoplankton species, and some form of predictive statement concerning factors involved in the ebb and flow of blue-greens. In particular, increasing water column stability, and the N:P ratio are factors which appear to predispose development of *Anabaena* biomass.

Overall the complex nature of variable loading/flushing and nutrient transfers associated with thermocline disruption in this pond would make prediction of the P fluxes very difficult.

5.4 Conclusion

The physical design characteristics of the three detention ponds provide theoretical P retention coefficients of 63% or more. Runoff studies in the Lakeview pond catchment averaged to a value of 0.25 to 0.3 kg TP/ha impervious area/0.58 yr. As a daily average, internal loading of DIP dominated the mass load to the system. The DIN (nitrate) load to the Lakeview system appears to be heavily dominated by groundwater seepage into the storm sewer network. It is conceivable that these flows also contribute to the thermal stratification which develops, and that the displacement volume is an additional mechanism for nutrient transfer to the upper water. The use of a basic light limited phytoplankton growth model predicted settling of organic-C, which was generally lower than that estimated in sediment traps. The use of traps over short dry-weather time intervals only are useful in this situation, and alternative methods of estimating this loss process would be preferable. In terms of pond operational performance and nutrient budgets, outflow quality must be measured in conjunction with future runoff studies.

Chapter 6

ADDITION OF INORGANIC NITROGEN TO ALTER THE N:P RATIO IN A NITROGEN LIMITED SYSTEM - EFFECTS ON PHYTOPLANKTON BIOMASS AND COMPOSITION

6.1 Introduction

Forsberg *et al.* (1978) generalized trophic state transitions in relation to N:P ratios and proposed that ratios <10 (by weight) are characteristic of nitrogen limited systems, while ratios >17 imply P limitation to algal growth. Smith (1983) demonstrated a borderline value of about 29 (as TN:TP), above which the development of N-fixing blue-green species are disfavoured. Clearly, the best option to increase the N:P ratio is to reduce the levels of P and so limit the yield potential. However, in enclosure and whole lake experiments several authors have demonstrated that favourable alterations in the phytoplankton composition were induced by manipulation of the N:P ratio with inorganic N additions. High ratios favoured development of green algae and non-N-fixing blue-greens (Barica *et al.,*1980; Leonardson and Ripl, 1980). However, some whole lake additions of N have been reported to stimulate the existing blue-green population, and failed to promote the intended shift to desirable species (*e.g.,* non-toxin producing, grazable) (Lathrop, 1988).

This chapter presents the results of whole-pond N additions carried out at one and two week intervals during May to July, 1994 in the City of Regina's shallow Lakewood pond. The pond typically develops a dense N-fixing *Anabaena* sp. bloom every summer, which may persist for up to one month depending on climatic conditions. Factors which predispose the extent of the high P and very low DIN:DIP (usually <0.5) environment of this pond have been previously discussed (high V_{rel}, high water residence times, internal loading, *etc.*). The aim of N additions was to determine the algal community response, the effect on DIP, and whether any alteration in the zooplankton community resulted. It was of particular interest to see whether blue-greens could be disfavoured, and the extent to which the vernal species composition might be extended by increased N:P ratio. Data was examined in relation to that observed in Lakewood pond in other years, and to that observed in the neighbouring Rochdale pond, in which *Oscillatoria* usually dominated.

6.2 Results

6.2.1 Climatic conditions during 1994

Water temperatures for 1994 were given in Chapter 3 (see Figure 3.1). Depth records from Rochdale pond were used to assess times at which Lakewood pond would have been affected by evaporation and stormwater runoff (Figure 6.1). Depth records throughout July showed that evaporation was a partial concentration mechanism; the accumulated loss was 8 cm (~6 to 8% of the permanent pond volume). During the first and last weeks of June, there was substantial rainfall. The period between the first week of July until the first week of August was dry.

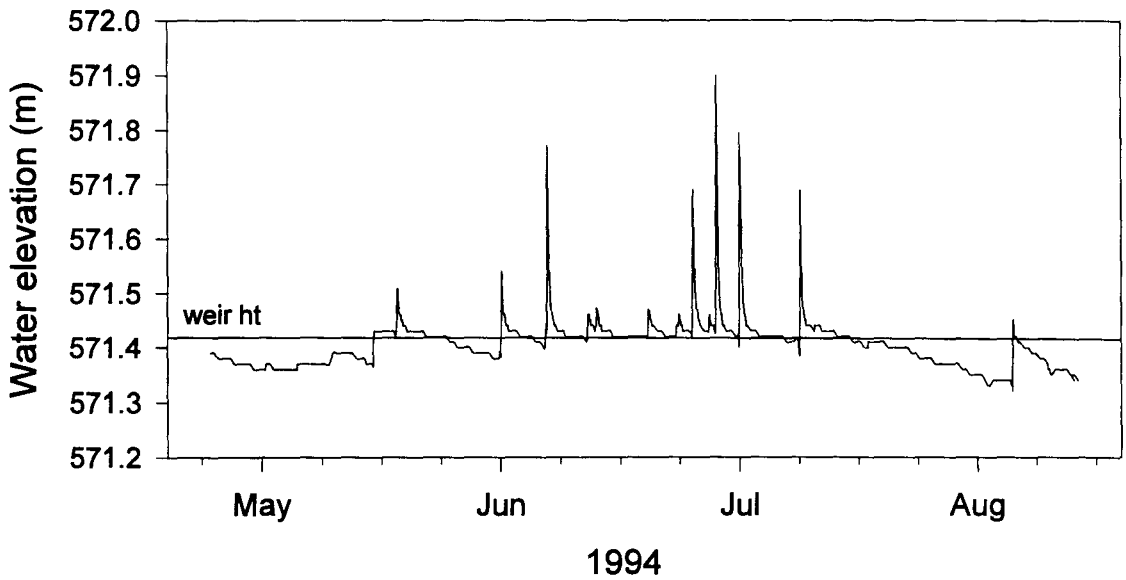

Figure 6.1 Pond depth record for the Rochdale stormwater detention pond (control), 1994.

Wind speed data was obtained from Regina Airport (AES, Canada) and expressed as the number of days/week on which average wind speed was less than 15 km/hr. The data shows that lower than average daily wind speeds were sustained from July until mid-August. The quiescent conditions (water column stability) throughout July were a contributing factor to blue-green bloom development (Figure 6.2).

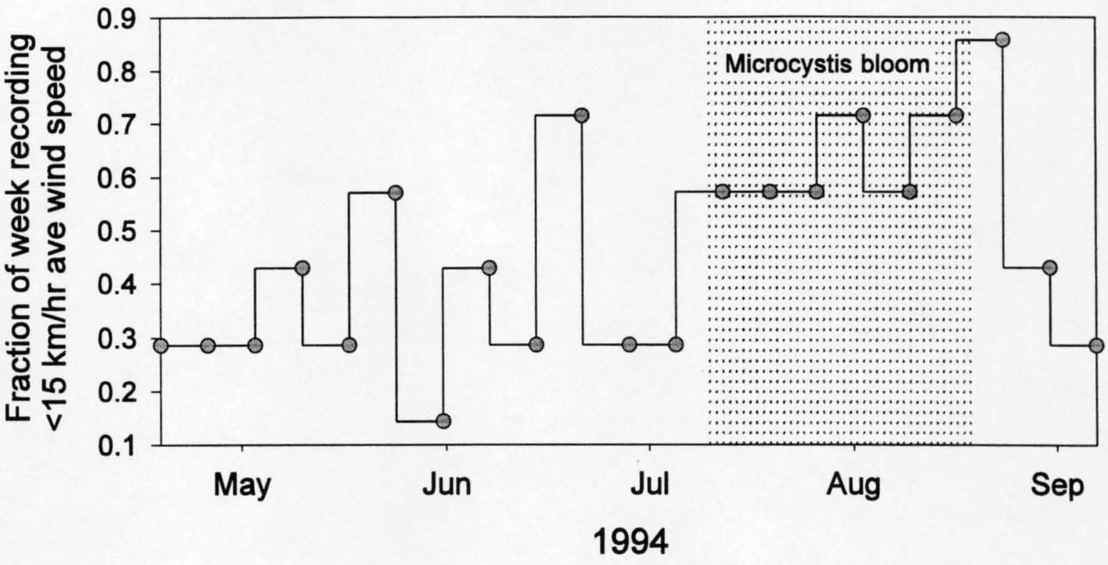

Figure 6.2 Daily wind speed record for the Regina area. (expressed as the number of days per week on which wind speed was less than 15 km/hr)

6.2.2 Chemical responses

Quantities of N fertilizer (ammonium nitrate 35-0-0, Sherritt) and the application dates are given in Table 6.1. A 300 kg application resulted in a concentration of about 3.25 mg/L DIN, or 5.5 g/m^2.

Table 6.1 Nitrogen fertilizer additions to the Lakewood stormwater detention pond (Regina), 1994.

Date	May 11	May 25	June 1	June 15	June 29	July 6	July 13	July 20
kg 35-0-0	300	200	300	300	300	200	300	300

Increases in DIN were measured after all additions, but some applications resulted in more or less recovery than others. Post-addition NH_4 and NO_3 increases were usually of the same magnitude, but recoveries on June 1 and July 6 had less NH_4 than NO_3 (Figure 6.3 a,b). The rapid disappearance of NH_4 was particularly evident after only one day in the July additions. Sampling within one week of additions showed that both NH_4 and NO_3 had been reduced on some occasions, while on others NO_3 had not been exhausted.

Prior to beginning N additions, DIN was barely detectable, while DIP was between 0.06 to 0.10 mg P/L, indicating the phytoplankton were N-limited at this time (Figures 6.3 b and 6.4). Nitrogen additions throughout May and early June resulted in only a small increase in PN and PP. Immediately after the first N additions in May, DIP was reduced from 0.09 to <0.01 mg P/L, with a corresponding increase in the PN and PP fractions. The rapid removal of DIP indicated that productivity was stimulated, but the dominance of the phytoplankton by diatoms/greens combined with cool water temperatures resulted in a fairly steady biomass following additions until the end of June. Increased rates of P loss by sedimentation and moderate mineralization rates in the sediment surface were assumed responsible. Residual NO_3 levels and low DIP indicated that the phytoplankton may have been P-limited for short periods throughout this time.

PN and PP levels declined slightly over the first two weeks of June, but DIP was simultaneously increasing. The N addition on June 15 facilitated removal of this DIP back into the particulate fraction. By the last week in June, DIN had reached low levels and sedimentation had caused PN to decline. As before, this decline was matched by increasing DIP, which proceeded to a seasonal maximum of 0.20 mg P/L. Following the N addition on June 29, there was a large rainfall event, so nutrient transformations were not attributed solely to in-pond exchanges. Nevertheless, a substantial DIP decline was matched by a proportional increase in PP and PN. On July 6, post-addition samples failed to recover increased NH_4 comparable to NO_3, although only a 200 kg application was made. It is unlikely that this was due to simultaneous flushing (loading NO_3 while flushing NH_4), since depth records for Rochdale showed the pond had finished releasing stormwater on the day of application. Sampling on the following day (July 7) also suggested flushing through a reduction in PP and PN, but there was no change in DIP. However, these particulate differences could also have been due to wind influences.

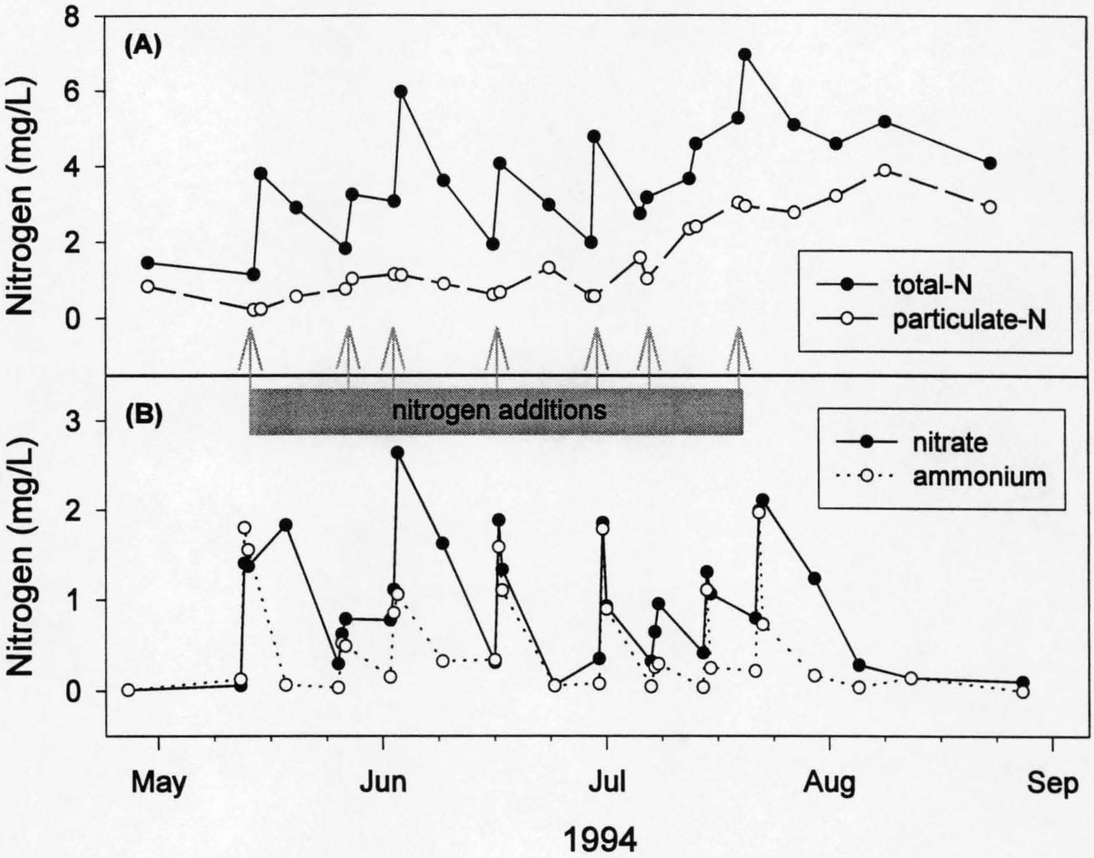

Figure 6.3 **Total and particulate nitrogen (A), and dissolved inorganic nitrogen (B) in Lakewood pond, 1994.**

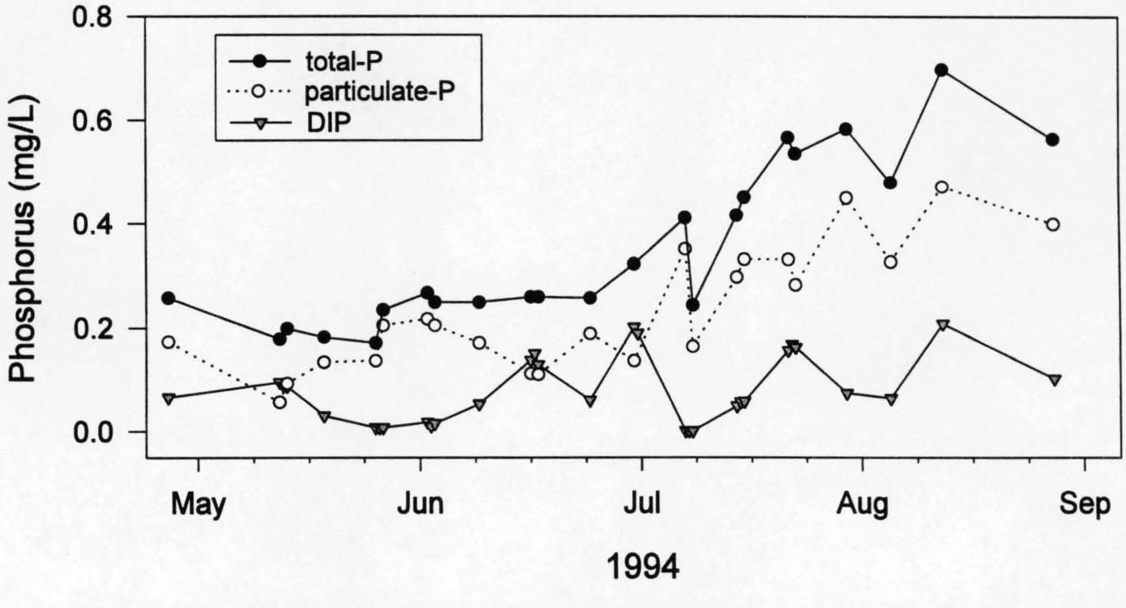

Figure 6.4 **Total, particulate, and dissolved inorganic phosphorus in Lakewood pond, 1994.**

By July 13, all P parameters and PN had increased from the previous week and the phytoplankton were probably neither N- or P-limited. The net internal loading rate increased with increasing water temperature, although 26 mm of precipitation was recorded on July 10, so external loading may also have contributed. The final addition on July 20 again clearly showed the rapid removal of NH_4 within one day. The increased N availability promoted removal of some of the DIP which had accumulated, but P never became limiting (Figure 6.4). No significant precipitation was recorded until the first week in August, after which DIP, PP, and PN increased. DIN remained very low until the end of the sampling program.

6.2.3 Soluble and particulate nutrient ratios and nutrient limitation

During the N addition period, DIN:DIP ratios fell below 10 on only three occasions, and these were always measured in samples taken before additions, after one week or more had elapsed since the previous addition (see Chapter 3, Figure 3.8, for DIN:DIP ratios). Assuming DIN:DIP ratios above 10 to imply zero N limitation (Uhlmann, 1982), the phytoplankton were potentially N-limited on only eight out of 23 sampling dates. Phosphorus limitation was probable during only two periods - initially following the first N additions (May 18 to June 2) and during the first week in July.

In relation to the TN:TP boundary (>29 by wt) proposed by Smith (1983) to disfavour blue-greens, ratios above this threshold were never achieved on account of the high TP content of the system. The highest TN:TP ratio achieved during N additions was 23, with all other values between 5 to 15. The minimum, maximum, and average TN:TP ratios and DON concentrations for Lakewood in 1994 are given in relation to the other ponds in Table 6.2. Although DON was not measured in other years, it seemed clear that concentrations were considerably elevated following N additions, when compared to Rochdale (control) pond. Average top water DON (assumed to be primarily autochthonous) levels in the Lakeview pond were relatively high, and may be explained by the lesser influence of flushing in this system, as discussed in Chapter 5. In the shallow ponds, DON levels fluctuated more than might be expected in natural systems on account of periodic flushing losses. The average TN:TP ratios in both Rochdale and Lakeview were below the threshold of 10 given to imply the predominance of N limitation, and always below the P limitation threshold of 17, as given by Forsberg *et al.* (1978). The average TN:TP ratio in Lakeview was higher than that of Rochdale pond, and coincided with the lesser seasonal abundance of blue-greens in the former pond.

Table 6.2 Comparison of total nitrogen, dissolved organic nitrogen, and the total nitrogen:total phosphorus ratios in Lakewood, Rochdale, and Lakeview ponds during 1994.

Pond		Total-N (mg/L)	DON (mg/L)	TN:TP Ratio
Lakewood	Min	1.14	0.44	5.6
	Max	6.96	1.23	23.9
n = 22	Avg	3.62	0.89	11.2
Rochdale	Min	1.21	0.31	4.7
	Max	2.69	0.64	8.2
n = 17	Avg	1.78	0.47	6.4
Lakeview (top)	Min	1.25	0.49	4.5
	Max	2.19	0.92	14.9
n = 20	Avg	1.66	0.72	8.9

A summary of the seasonal C:N:P seston ratios for Lakewood pond has been given previously (see Chapter 3, Table 3.5). The data showed that N additions to the pond resulted in marked changes to the average ratios observed in other years (less so in 1995), and in the control pond during the period. The notable alteration was a reduced C content, which acted to give a seasonal mean of 34:6:1 as C:N:P. In this instance, luxury uptake of both N and P may have been facilitated. The Lakewood pond data was examined according to the scheme of Healey and Hendzel (1978) for the 1993 to 1995 seasons (Figure 6.5). Previous discussion was made in relation to resuspension and potentially high detrital content of the seston in these shallow systems. For this reason, the seston is likely to contain more C than that present in living cells, which may tend to increase signs of N or P deficiency relative to C. With this factor in mind, the pond presented no evidence of N or P limitation according to cellular content despite consistently low DIN in other years. During 1994 N additions, all except one of the N:C ratios were confined to the zone of zero N deficiency. The spread of the N:C ratios were not significantly different from those of 1993 or 1995. However, the highest P:C ratios were measured in 1994 (Figure 6.5).

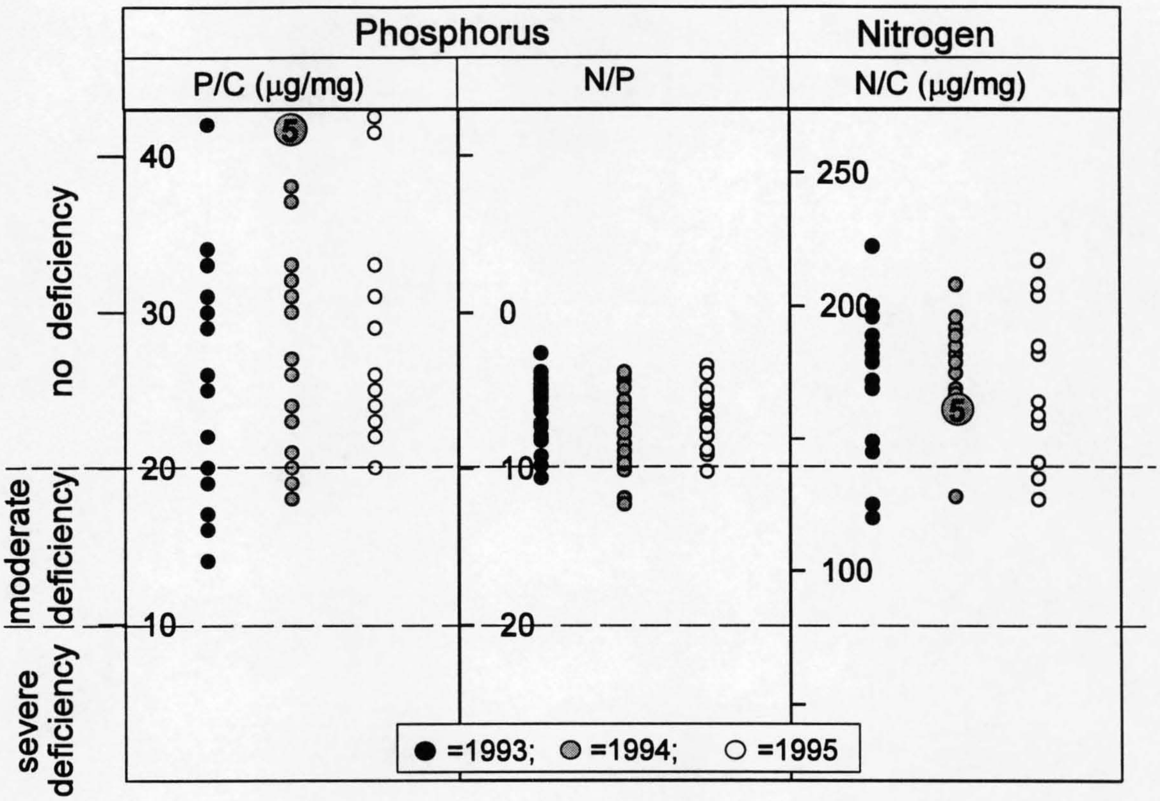

Figure 6.5 Particulate carbon/nitrogen/phosphorus ratios of Lakewood pond seston, 1993 to 1995. (according to the nutrient limitation scheme of Healey and Hendzel, 1978)

pH: The Lakewood pond typically experienced pH values >9.5 shortly after ice-melt and again later in the summers of 1992 to 1995. In 1994, pH tended to progressively increase within a band of oscillation from about 9.0, one week after additions began in May, to a phenomenal peak of 10.7 measured during the algal biomass peak at the beginning of August. By the end of the monitoring period, pH remained high at 10, despite a reduction in algal biomass. The mean pH for the season was 9.6.

6.2.4 Biological responses

Pigments: From the period April to June, chlorophyll fluctuated in parallel with the PN and PP changes described above. The first N addition stimulated an increase to 0.18 mg Chl *a*/L. Thereafter, Chl *a* increased gradually to 0.30 mg/L by the end of June. During this time, Chl *a* was always higher than Chl *b*, and both were slightly lower than in the control pond. By July, massive increases in Chl *b* accompanied the increasing biomass and water temperatures, showing that light was becoming increasingly limiting to the community. Secchi disk depths were reduced to 0.2 to 0.3 m (Figure 6.6).

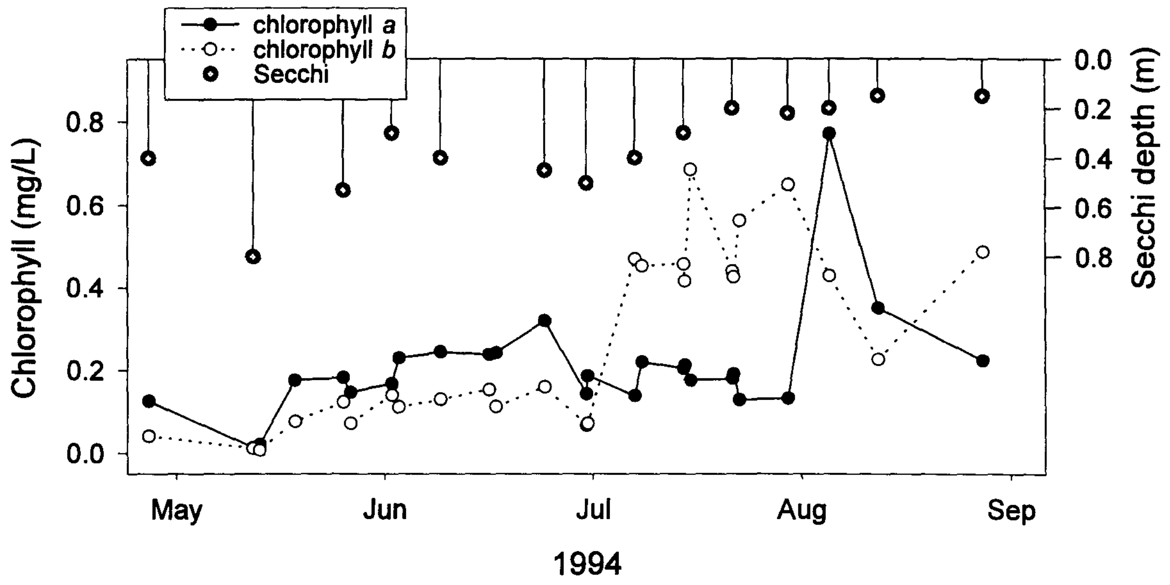

Figure 6.6 Chlorophyll and Secchi disk depth in Lakewood pond (Regina), 1994.

A temporary switch in the Chl *a:b* ratio in early August was associated with the peak of a *Microcystis* bloom (estimated at 90 to 100 mm^3/L) when greens were almost completely eliminated from the community. Peak Chl *a* levels in August reached 0.77 mg/L, and oxygen saturation of 200% was measured near mid-day (see Chapter 3, Figure 3.4, for Lakewood oxygen data).

<u>Phytoplankton species composition in the experimental pond:</u>

Comparison of the total cell volume data and the temporal pattern of blue-green dominance for 1993 to 1995 was presented for both the experimental and control ponds in Chapter 3 (Figures 3.12 c and 3.11 c, respectively). Despite N additions, the usual temporal increase in blue-green biomass (mid- to late summer) was developing prior to the last application date. However, notable deviations from the pattern observed in other years were that blue-greens were conspicuously absent from the phytoplankton until the beginning of July, and the usual development of N-fixing *Anabaena* sp. never took place (as it did in 1992, 1993, and 1995). Instead, increasing amounts of *Microcystis aeruginosa* ultimately formed a massive bloom towards the end of July, a species that was never observed in significant numbers in other years.

Figure 6.7 shows the 1994 algal composition according to the main groups. Chrysophytes and Dinophytes never contributed significantly to the biomass, although representatives were observed on some dates. Prior to July, the total biomass was below 25 mm^3/L, which was lower than the control pond, and comprised a mixture of diatoms, Cryptomonads, Euglenoids, and Chlorophytes. Immediately after the first N addition in May, Chlorophytes became dominant, and comprised mostly colonial *Pandorina* sp. and *Eudorina* sp. A large *Euglena* sp. also accounted for 25% of the total cell volume, along with numbers of various small coccales and flagellates (3 to 8 µm). In June, Chlorophytes gradually declined, and a peak in *Cryptomonas ovata* occurred in the middle of the month. This peak was stradelled on either side by a brief increase in pennate diatoms <30 µm in length. The second of these diatom peaks was accompanied by a resurgence of Chlorophytes. By July, the diatoms had all but disappeared and the Chlorophytes continued to increase, heavily dominated by both unicellular and colonial (<50 µm) *Pandorina* sp. (Figure 6.7).

Throughout July *Microcystis* and a lesser amount of *Aphanocapsca* colonies began to increase. *Microcystis* peaked in the first week of August when it dominated the biomass almost completely. The cell volume data reflected the Chl *a:b* fluctuation described above from July onwards, during the transition from shared blue-green and green algal dominance to complete blue-green dominance and back to shared dominance by the end of August. *Microcystis* were clearly able to optimize their light climate through buoyancy control under the quiescent conditions, forcing Chl *b* synthesis in the competing community of greens. The peak *Microcystis* bloom was preceded by and occurred during several weeks of hot, dry conditions, which had lower than average weekly wind speeds (Figure 6.2). Decline of the bloom coincided with moderate storm flows and unsettled weather conditions in the first week of August, which appeared to facilitate recovery of Chlorophytes and Cryptophytes, probably through mixing and external loading effects. The extent to which flushing losses were involved was probably minimal, as depth records from the control pond showed that most of the runoff compensated the cumulative evaporation losses from the preceding dry weeks (Figure 6.1). The later Chlorophyte community was dominated by *Scenedesmus* sp., *Golenkina radiata*, and various other small cells.

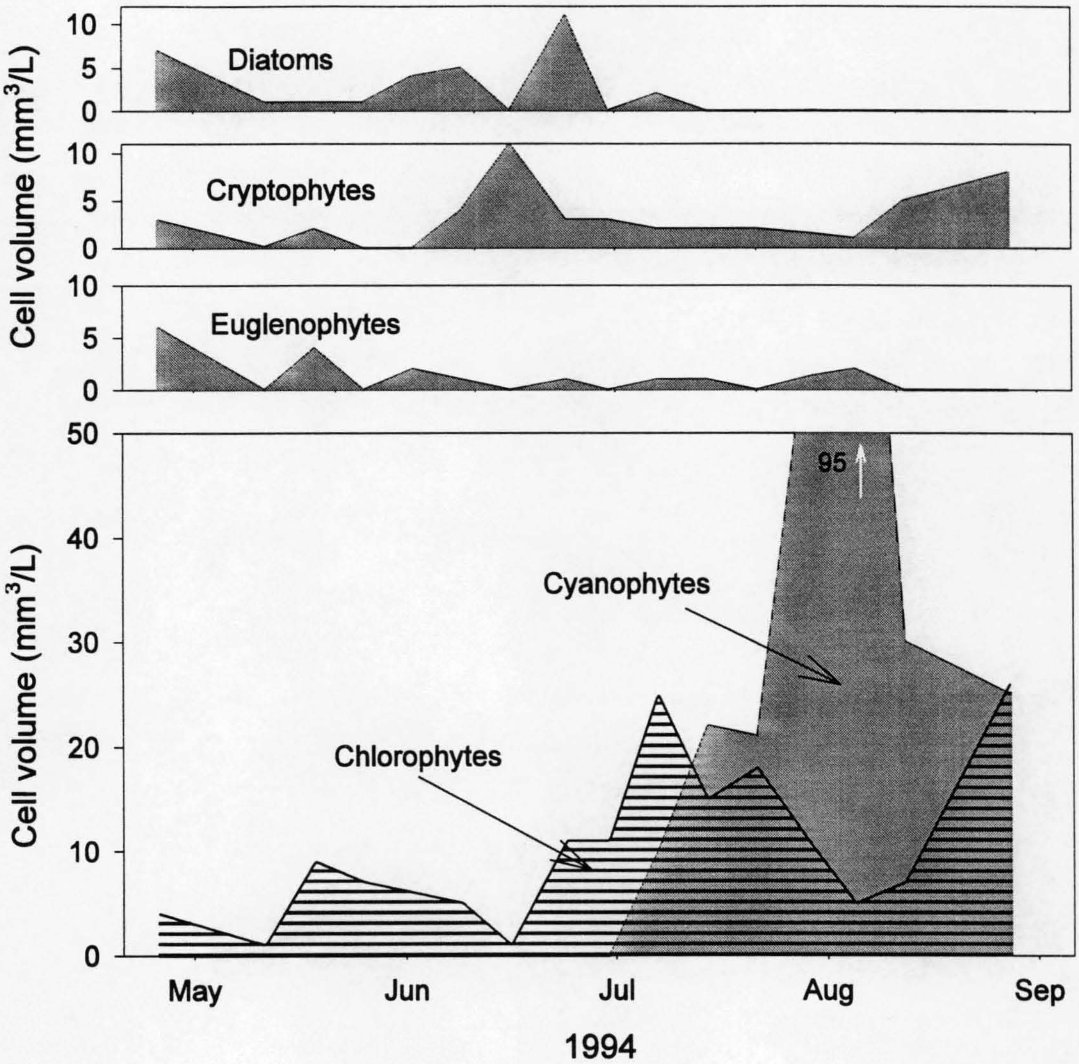

Figure 6.7 Phytoplankton composition during and after the period of nitrogen additions to Lakewood pond (Regina), 1994.
(algal divisions are graphed to relative scale)

Phytoplankton species composition in the control pond: Typical of the pattern observed in other years, the control pond was dominated by *Oscillatoria tenuis.* (≈ 4 µm diameter) from May through to mid-July at total biomass levels higher than that of the experimental pond (data not shown). Storm flows around the end of June flushed biomass from the pond and by the end of July a bloom of small lunate greens (<8 µm in length), tentatively identified as *Selanastrum*, had developed. This bloom peaked in the first week of August to about 60 mm^3/L (simultaneous with the *Microcystis* bloom in Lakewood pond), and continued to dominate the phytoplankton until the end of the monitoring period.

Zooplankton: Total zooplankton numbers for the seasons 1993 to 1995 in Lakewood were given in Chapter 3 (Figure 3.15). The zooplankton composition during 1994 in Lakewood pond is shown in Figure 6.8. As in previous years, the community continued to be dominated by cyclopoid copepods and their nauplii (*Ancanthocyclops vernalis* and *Mesocyclops edax*). In all years, calanoid copepods were present in only very low numbers. In 1994, cladocerans were absent until July, when the group peaked and declined over the course of the month. The species involved were *Daphnia parvula* and *Bosmina longirostris* (similar brief peaks occurred in other years). It was notable that this peak in herbivorous grazers occurred over the period when Chl *b* levels and Chlorophytes were highest. The decline in cladocerans preceded the blue-green peak, while a partial increase followed the recovery of Chlorophytes when the *Microcystis* bloom subsided. A reduction of the total zooplankton generally followed the same pattern and, since no precipitation was recorded between mid-July and the end of the first week of August, flushing cannot account for the simultaneous declines of all groups. However, adult cyclopoid copepods were on a rising limb of their cycle while *Microcystis* biomass was still high. Conceding that seasonal differences exist due to varied rainfall patterns, there was no evidence of significant deviation from the usual species composition and peak numbers compared to 1993 and 1995. However, since adult cyclopoid copepods were sustained in relatively high numbers during June and July 1994, it is not known to what extent a potential increase in herbivorous zooplankton may have been counteracted by cyclopoid predation.

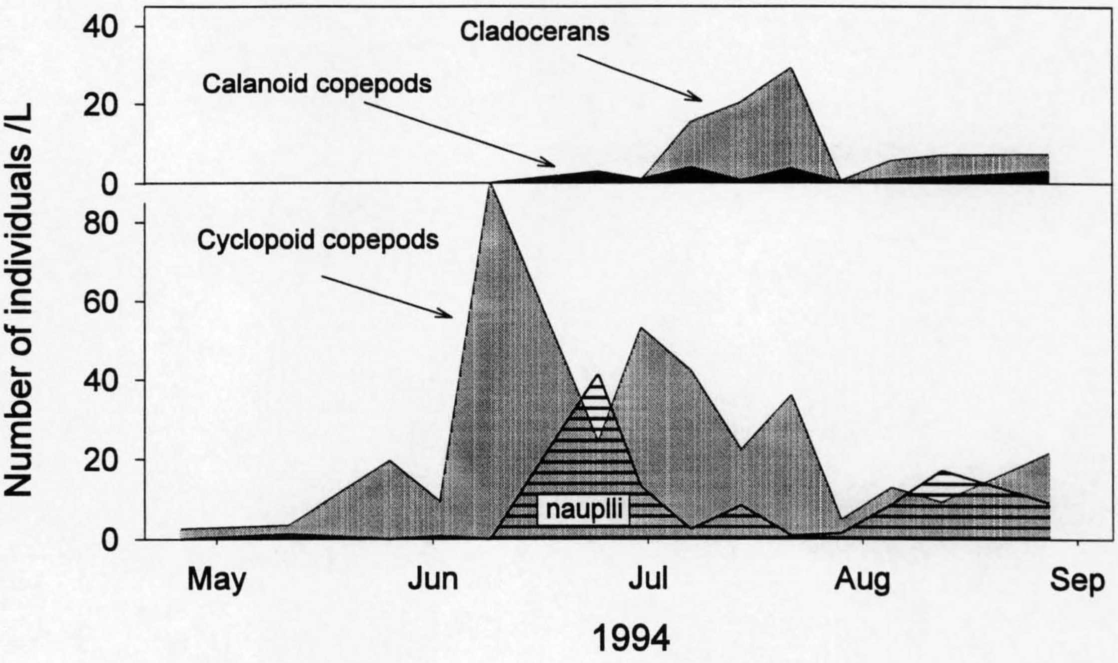

Figure 6.8 Zooplankton species composition by major groups in Lakewood pond (Regina), 1994. (zooplankton divisions are graphed to relative scale)

6.3 Discussion

A number of conditions must be satisfied prior to and during the development of a blue-green bloom, although the variables may differ from one genera to the next according to their light and nutrient optima, and potential vertical migration rate, *etc.* Reynolds (1984) summarized three basic requirements: (1) a pre-existing population; (2) a significant proportion of the population containing sufficient gas vesicles to render them buoyant; and, (3) stability of the water column. While detailed information on the stability of the water column is not available, the low average daily wind speeds that occurred before and during the *Microcystis* bloom development appeared to conform to stability requirements. Convective mixing in elongate "Langmuir cells" from wind-induced rotations of the surface water of lakes may occur above a threshold wind speed of about 11 to 12 km/hr (Evans and Taylor, 1980). In this case, the demarcation of 15 km/hr seems reasonable since the pond, although elongate, is small and partially sheltered by parkland topography and buildings. Competing non-buoyant species (also indicated by Chl *b* decline) were then lost by sinking and insufficient recruitment rate under increasingly light-limited conditions (Secchi depth 0.2 m).

Gas vacuolate *Microcystis* colonies are known to be capable of relatively rapid diel migrations according to photic, thermal, and possibly nutrient conditions (Reynolds, 1984). In this instance, the additional prerequisite of low N:P to bring about N-fixing *Anabaena* succession was prevented by N additions. After cessation of N additions, recycling of N within the system continued at a rate sufficient to prevent *Anabaena* appearance, even though DIN:DIP ratios returned to low levels (<1). Although a crop of *Oscillatoria* usually featured among the phytoplankton of Lakewood pond, it was notable that *Microcystis* was favoured following the N additions in conditions which may otherwise have suited N-fixing *Anabaena*. High UV tolerance and a capacity to maintain photosynthesis within the high pH range may also have been a significant feature in promoting this species (Reynolds, 1984; Shapiro, 1990). A sufficient inoculum existed within the pond, even though they did not feature significantly at any other time during the four years of monitoring. Studies have shown that *Microcystis* recruitment may derive from dormant stock in the sediment surface (*e.g.,* Preston, *et al.*, 1980), which may remain viable for many years (Stockner and Lund, 1980).

Seip and Reynolds (1995) gave an optimum N:P ratio (by wt) of 5 for *Microcystis*, while a high temperature optimum has also been reported (Reynolds, 1984). The control pond was most frequently dominated by *Oscillatoria* and tended to maintain DIN:DIP ratios of <2, which confirms a relatively low half-saturation constant for N for *Oscillatoria* growth (Tilman *et al.*, 1982, 1986; van Liere *et al.*, 1977; van Liere and Mur, 1980). *Oscillatoria* strains may also have a wider temperature adaptation range compared to *Anabaena* and *Microcystis*, thereby giving different ecological characteristics (Reynolds, 1984; 1987). The green bloom that occurred during late July in Rochdale pond was presumably initiated by altered nutrient conditions following the June/July storm flows, as well as the climatic conditions. This replacement occurred despite the quiescent conditions suggested to have predisposed the *Microcystis* bloom in Lakewood, although the Rochdale pond is somewhat less sheltered than Lakewood. However, Reynolds (1984) states that stability selects against *Oscillatoria*. In this case, a high growth rate of small chlorophytes was apparently sufficient to establish a high and dominant biomass under the altered nutrient conditions and high temperatures (20 to 25°C) that followed the flushing of *Oscillatoria* biomass.

In the Lakewood pond, the endurance of dominating *Eudorina* sp. and *Pandorina* sp. during the N addition period was also a departure from the composition seen in other years. Therefore, it is not surprising that these species have been reported to have a high nutrient requirement and to favour high N environments (Prescott, 1962). The brief increase in cladoceran zooplankton which coincided with the oscillation *of Pandorina* sp. (many of the cells were solitary) suggests that a grazing relationship may have been involved. The decline of all zooplankton during the peak of the *Microcystis* bloom may have been due to the quality of nutrition, exceptionally high pH, or possibly toxin production by the bloom (Fulton and Paerl, 1987).

In relation to the low seston C:N:P ratios, light limitation will reduce algal growth rate, so that respiratory C losses are increased relative to photosynthetic gain, and a reduced cellular C content may result (Fogg, 1975). The low C:N and C:P ratios may have resulted from the combination of both respiratory loss with surplus storage of N and P. Luxury storage of polyphosphate granules is a well recognized phenomenon, while the capacity for N storage is probably lower in magnitude. It stands to reason that more P may be stored if the cell has a N supply adequate to sustain a healthy condition. Rapid removal of N (particularly NH_4) suggests that the phytoplankton were substantially N-limited on occasion.

Volatilization of NH_4 is a complex function of pH, temperature, turbulence of the surface water, and the air velocity over the interface. Neglecting wind, mixing, algal uptake, and nitrification, a very rough estimate of NH_4 volatilization was made with the rate constant and pH/temperature coefficients of Stratton (1969). Assuming a complete dispersion concentration of each N addition, the volatilized loss calculated after 24 hours ranged from 1 to 23% of the added areal mass NH_4-N, with an average of 7.6% per day for the eight N applications. Only one value calculated to greater than 10%, on account of a pH of 10.4. Therefore, the almost complete NH_4 removal observed on some dates within 24 hours would be attributed to biological transformation, while theoretically significant losses may also occur at the very high pH values measured. Denitrification may occur within anaerobic sediment surface zones (Kaspar, 1985), but no estimate of this was made. It is likely to have been a significant loss mechanism at the elevated temperatures, and especially as the fertilization program enhanced the organic production rate. Nitrate addition alone would be preferable since metabolic expenditure is required for assimilation, while ammonium adds COD to the system.

Apart from its operation as a SWDP, the Lakewood pond presented many similar characteristics to hypereutrophic Indiana Lake, California, described by Lathrop (1988), in terms of depth, temperature, chlorophyll, and the frequent occurrence of N-fixing *Anabaena* blooms. Lathrop reported the results of ammonium nitrate additions carried out between April and July in 1981 and 1982. In that case, *Aphanocapsca* and *Microcystis aeruginosa* bloomed after N additions had ceased in July of the first year. After additions had ceased the second season, *Microcystis* again developed, but only after passing through a phase of *Anabaena* dominance. However, unlike this study, *Microcystis* and *Aphanocapsca* were significant among the phytoplankton reported by Lathrop for the pre-treatment summer and in the year following additions.

6.4 Conclusion

The results of inorganic N additions over three months in the hypereutrophic Lakewood pond prevented the usual bloom of N-fixing *Anabaena* under conditions which predisposed such blooms in other seasons, and was replaced by a bloom of non-fixing blue-greens. Nitrogen supplements during May and June increased productivity of green algal species, which lowered DIP *via* uptake and sedimentation of the particulate fraction. The average biomass level was similar to that of other years, but as a consequence of reducing N limitation of early to mid-season populations, the oxygen demand of the sediment surface would necessarily have increased. Increasing rates of recycling and favourable climatic conditions then produced N:P levels which allowed non-fixing blue-greens to thrive. Other than to specifically target a N-fixing bloom, there are no major benefits of N addition to these shallow systems as long as a high seasonal rate of P recycling and internal loading continues to occur. In systems with lower levels of P, potential N supplements in the form of nitrate only would be preferable to ammonium.

Chapter 7

USE OF ALUMINIUM SULPHATE TO REDUCE PHYTOPLANKTON BIOMASS IN A HYPEREUTROPHIC STORMWATER DETENTION POND

7.1 Introduction

Although nitrogen and phosphorus may singly or jointly limit primary productivity, control of phosphorus is the most effective strategy for management purposes. Unlike phosphorus, nitrogen imbalances may encourage the growth of N-fixing blue-green algae, when phosphorus remains available. Efforts to restore eutrophic lakes following reductions of external loads have frequently been delayed by the continued release of P from sediment reserves (Marsden, 1989). Internal P loading may be highly significant in warm, shallow, and well mixed water bodies since the ratio of sediment surface area/water column volume is high, and nutrient recycling takes place within, or, in close proximity to the photosynthetic zone (Cooke *et al.,* 1993*a*). This scenario is typical of the SWDPs in this study, particularly since they function as closed systems during dry weather.

Salts of aluminium, iron, and calcium are frequently used as agents to reduce particulate and dissolved P through pH controlled precipitation reactions. Phosphorus inactivation may involve the water column (subsequently forming a blanket of precipitate on the sediment surface), or in some cases, a direct application into the sediment structure (*e.g.,* Cooke *et al.,* 1993*a, b*; Quaak *et al.,* 1993; Ripl, 1985). The former method may be effective in deep stratifying lakes, but frequently less so in shallow systems due to the effect of floc displacement.

This chapter presents the results of a full-scale trial with aluminium sulphate in the City of Regina's Rochdale stormwater detention pond during 1995. Aluminium was chosen over iron salts partly because extended anoxia always develops under ice cover, so that reduced iron may become effectively lost from the P binding pool by precipitation with sulphide (Auer *et al., 1993;* Tessier *et al.,* 1993).

7.2 Results

7.2.1 Precipitation and pond volume exchanges

Pond elevations and the calculated percent pond volume exchanges following precipitation in 1995 are given in Figure 7.1. A very large storm event that occurred in May (45 mm) led to an exchange of over 60% of the pond permanent volume. The period June to mid-August was relatively dry, with only one event in the first week of July exceeding 10% exchange. In mid-August, two large exchange events were generated from thunderstorms, giving a total recorded precipitation of 67 mm.

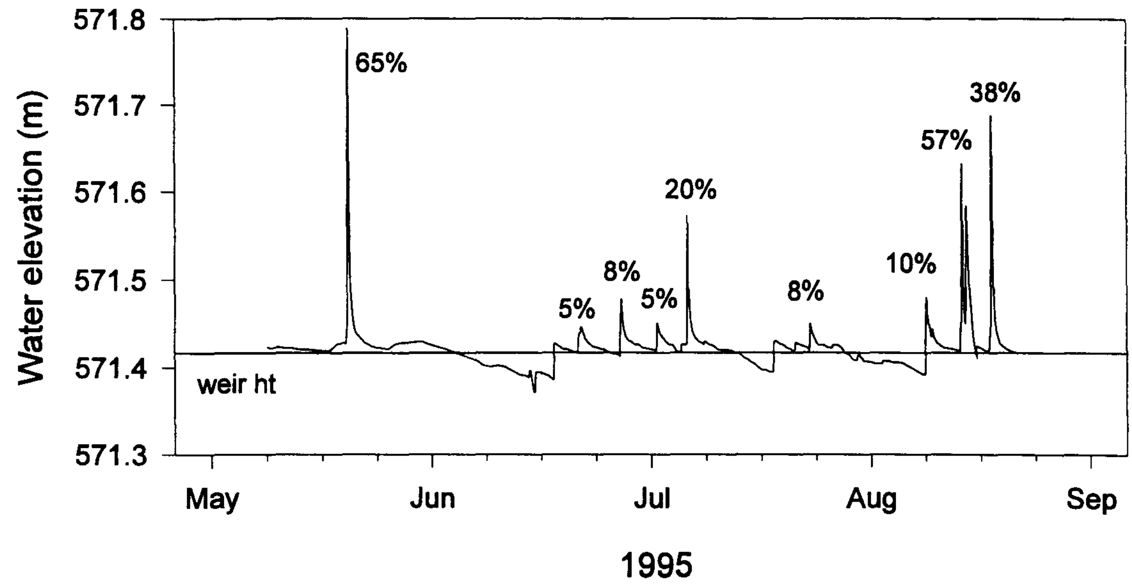

Figure 7.1 Elevation and percent volume exchange in Rochdale pond (Regina), 1995.

7.2.2 Responses to alum treatment

Chemical responses: On June 29, 1995, 6600 L (8.8 metric tonnes) of liquid aluminium sulphate (4.44% Al to give a final concentration of ~10 mg/L Al) was applied to the propwash of a traversing boat by pumping the chemical from a north and south shoreline location *via* a floating supply hose. A second boat was used to maintain gentle mixing behind the line of application. Immediately after the application, pH was reduced from 9.6 to a variation of 5.7 to 7.9. Within two hours, pH had stabilized between 6.7 and 7.0 throughout the pond. By the following morning, pH remained around 7.0. Alkalinity consumption was from 86 mg/L to 24 mg/L $CaCO_3$, which indicated optimum dose while remaining within environmental safety limits (*i.e.*, $\geq$25 mg/L alkalinity; pH$\geq$6.0, Cooke and Kennedy, 1989) (Figure 7.2). After the addition, DOC was reduced from 7.97 to 4.73 mg/L, indicating that a significant buffering capacity for soluble Al was still present. Organic ligands (*e.g.*, humics) may complex soluble Al, thereby providing a toxicity buffering mechanism (Driscoll *et al.*, 1980). No fish mortality was observed.

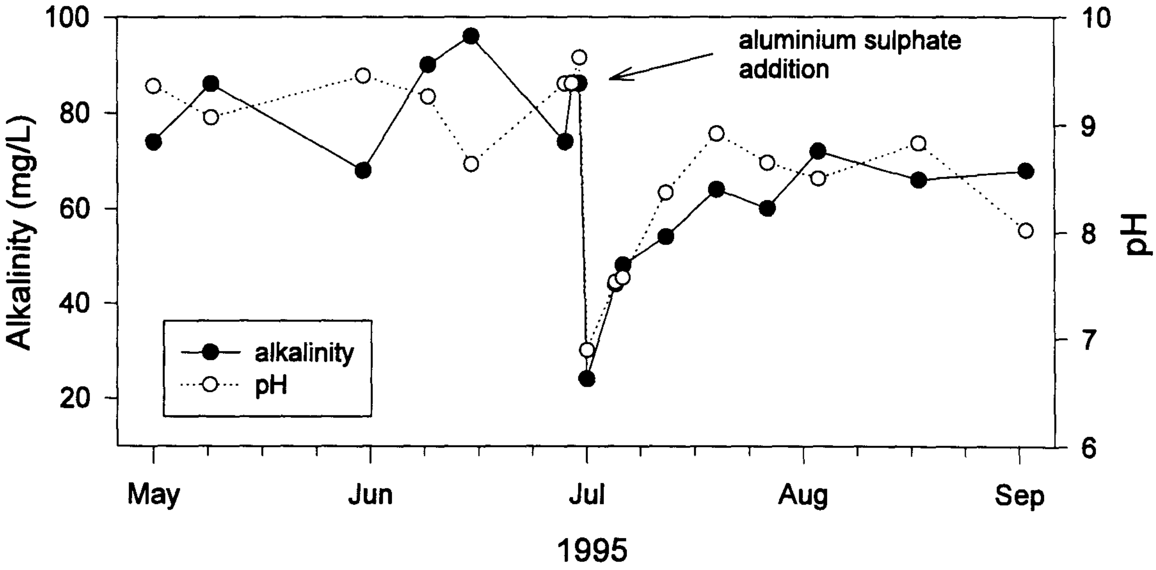

Figure 7.2 pH and alkalinity in the Rochdale pond (Regina), 1995.

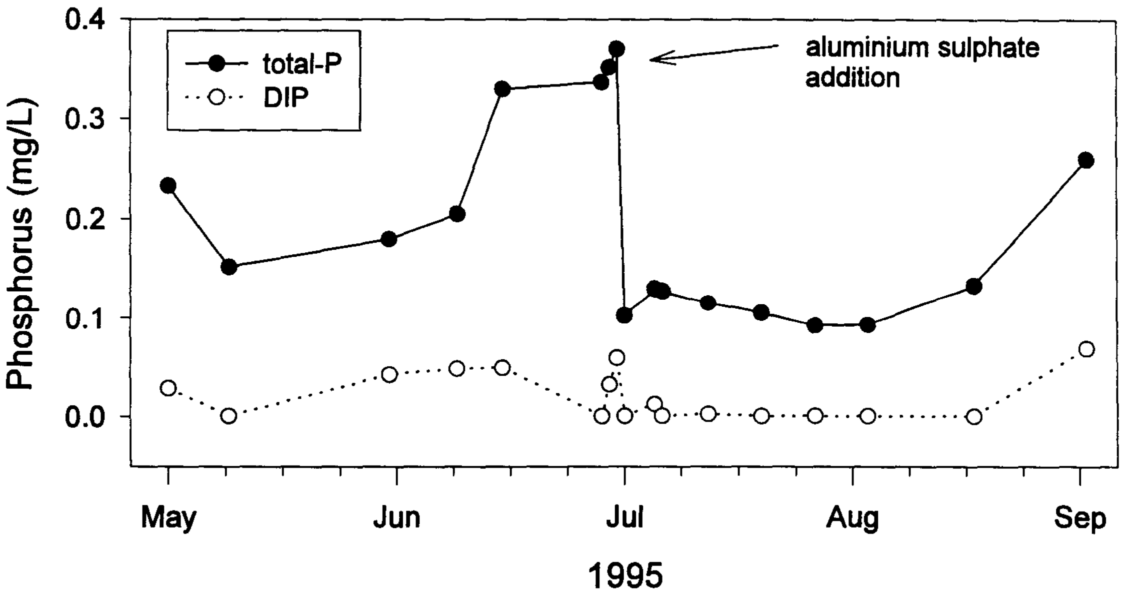

Figure 7.3 Water column phosphorus parameters in the Rochdale pond (Regina), 1995.

By the morning after the application, TP was reduced from 370 to 100 μg/L, while DIP was reduced from 60 to <10 μg/L (Figure 7.3). Chlorophyll *a* was reduced 10-fold from 320 to 34 μg/L, and Secchi disk depth increased from 0.2 to 0.6 m (see Chapter 3, Figure 3.11). Central axis sediment cores showed a floc depth between 0.5 and 3.5 cm throughout the pond.

Five days later, the pond was sampled again, following a small inflow on July 2. Chlorophyll *a* had recovered to about 50% of the pre-application value at 150 μg/L, but TP remained close to the post-application level of 130 μg/L. Dissolved inorganic phosphorus was still <10 μg/L, pH was still low at 7.5, and alkalinity had doubled from 24 to 48 mg/L (Figures 7.2 and 7.3). Following the 20% exchange event on July 6, floc resuspension and loss in outflows undoubtedly occurred. This may have acted to scavenge DIP before resettling, since DIP was still <10 μg/L on July 11. Total-P remained close to 100 μg/L.

Within two weeks of the application, sediment surface oxidation developed (~1 cm) as a result of reduced organic production and associated water column/sediment oxygen demand. This condition persisted until early August and would have contributed greatly to the reduced TP and DIP levels, which were sustained due to adsorption by sediment Fe-oxide complexes (Boström, 1982). In Chapter 4, data was presented concerning the absorption capacity of the surface sediment upon oxidation and the reduced concentrations of interstitial DIP that were measured at this time. The interstitial N:P ratio was considerably higher than that measured under anaerobic conditions at similar temperatures. Analysis of the surficial sediment at the end of July failed to show an increased Al content. It was notable that the period of improved water quality following treatment was sustained during peak water temperatures (20 to 24°C). In contrast, the neighbouring Lakewood pond developed high levels of TP, and phytoplankton biomass throughout July.

Sediment cores taken at the end of July showed no visual evidence of any remaining floc, which by this time may have been moved by wind action and inflow turbulence. By the end of July, alkalinity had increased to back to 72 mg/L. Chlorophyll *a* varied between 60 and 100 μg/L for the remainder of the sampling period, and pH was <9.0.

Phytoplankton responses: Total phytoplankton and blue-green biomass for the years 1993 to 1995 for both the experimental and control pond were presented in Chapter 3 (Figures 3.11 and 3.12). Prior to the aluminium sulphate application on June 29, there were several small precipitation events. In the Rochdale pond, *Oscillatoria tenuis* (~ 4 µm diameter) had dominated since late May, when the above events initiated change in the phytoplankton composition and a reduction of the total biomass. A shift to a small (<10 µm) lunate green, tentatively identified as *Selanastrum minutum*, developed. In the control pond a similar shift from a mixed biomasss of *Anabaena/Oscillatoria* sp. to a variety of greens also took place.

On the day of application, phytoplankton cell volume in Rochdale pond was 28 mm^3/L, comprising 20% blue-greens and 80% greens and others. One week after the addition, some shorter and thinner *Oscillatoria hamelli* filaments (~2.5 µm diameter) had replaced *O. tenuis*, but small greens were still dominant. Samples from the latter part of July showed that these blue-green filaments accounted for 30 to 40% of the biomass, but the total volume was relatively low at 9 to 13 mm^3/L. In contrast to the species composition and reduced biomass of the treated pond, the control pond developed a high N-fixing *Anabaena* bloom throughout July (total cell volume range 50 to 90 mm^3/L).

Zooplankton responses: Zooplankton appear to play a minor role in phytoplankton successions in these SWDPs. Both the Rochdale and Lakewood systems were dominated by cyclopoid copepods, with brief appearances of small-bodied cladocerans. The fluctuation of total zooplankton numbers in both ponds were given in Chapter 3 (Figures 3.14 and 3.15). One mid-summer and one late-summer peak in numbers was typical. Although a decline in total numbers took place after the aluminium sulphate treatment, a similar decline happened in the control pond. These rapid declines were also observed in other years. A closer examination of the adult copepods/nauplii fluctuations suggested that no insult to the cohort production sequence was apparent following treatment (Figure 7.4). Adult copepods (*Ancanthocyclops vernalis*) were on a falling limb of a cycle at the time of treatment. By mid-July, total numbers were again increasing, but it was notable that these were dominated by nauplii for the remainder of the monitoring period. Predation on adults by minnows (*Pimephales promelas*) may have been increased following treatment due to the increased water clarity.

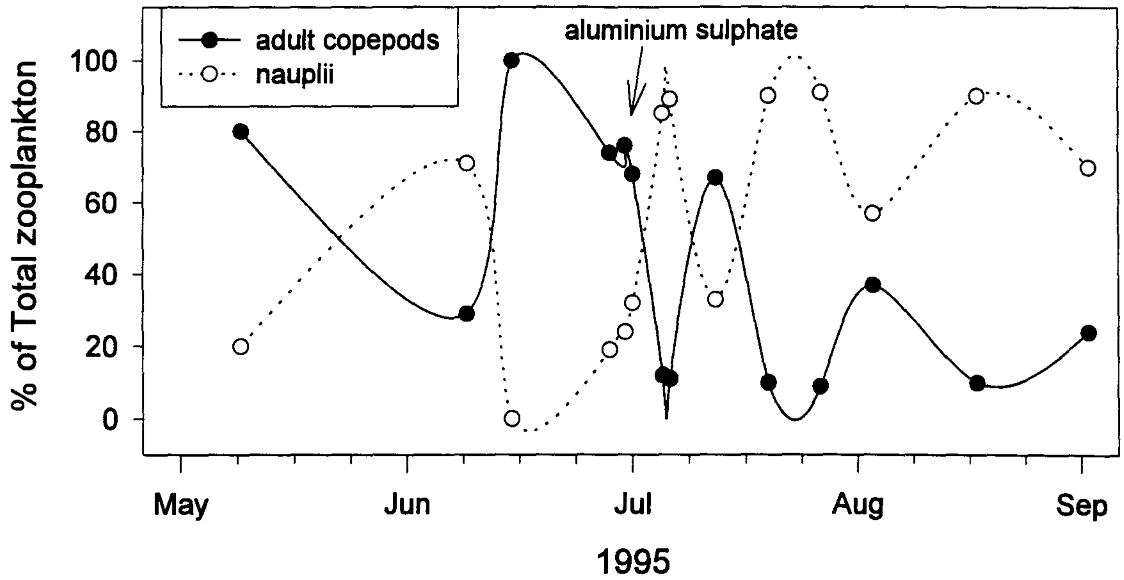

Figure 7.4 Production sequence of cyclopoid copepods in the Rochdale pond (Regina), 1995.

7.3 Discussion

For improved dry weather water quality, internal loading of P is the key process to be controlled. Aluminium sulphate treatment was successful in reducing water column TP to levels lower than those measured in other years, and considerably lower than concurrent concentrations in the control pond. The lack of visible floc one month after application suggested that a reduced oxygen demand, which was secondary to the precipitation of P, was the basis for additional P binding in the oxidized surface layer. In the surficial sediment, current Fe:TP ratios are >20 (by wt), which are in excess of those reported to be sufficient to suppress P release under oxidized conditions (Danen-Louwerse *et al.,* 1993). The areal dose of Al was 1.7 mg/cm^2, compared to a background concentration of 22 mg/g fresh weight in the top 1 cm of sediment. Therefore, incorporation would not be expected to produce a large change in Al content.

A number of data was previously given concerning the change in sediment surface TP content after oxidation, and the large reductions in the interstitial P concentrations after treatment of the Rochdale pond, despite peak water temperatures (Chapter 4, Figures 4.1 and 4.5). Based on the anaerobic and aerobic adsorption isotherms of the Rochdale pond sediments, an increase in the inorganic P binding from 324 to >737 mg P/m^2 was calculated following oxidation of the top 1 cm in July, 1995. The measured increase in the surface sediment TP of about 250 µg/g dry weight between June and July, 1995, conformed reasonably well to this prediction.

Water quality improvements lasted only about six weeks, and confirmed that the longevity of such treatments in these systems will be determined by the frequency and magnitude of stormwater loads. Treatments of SWDPs in the City of Edmonton (Alberta), using lime and/or alum were discussed by Babin *et al.* (1992). The authors concluded that alum alone was superior for controlling phytoplankton, but that routine applications are required due to continuous external loading throughout the open water season.

There are many more published examples of the use of Al over both Fe and Ca, and some of these were reviewed by Cooke *et al.* (1993*a, b*). Although Al salts may give improved P removal over Fe salts, the use of Al may present an environmental concern if pH excursions below 6.0 take place (Cooke *et al.*, 1993*a*). Boers (1991) stated that the use of Al in lake management/restoration efforts is currently disfavoured in the Netherlands, with Fe salts most often being used either alone *(e.g.,* Quaak *et al.*, 1993), or in conjunction with forced saprobel oxidation with nitrate and/or aeration (*e.g.,* Ripl, 1985). The dose of alum or iron is restricted by the available alkalinity in the systems, and the question of whether direct mixing of a large dose of buffered Al into the sediment surface would enhance treatment longevity is not known. However, as stated above, continued external loading will inevitably override any one single treatment of these systems.

Lowered pH may be an additional benefit favouring maintenance of green algae after treatment, whether *via* direct interaction, or by its effect on the HCO_3 buffering system and inorganic C availability (Shapiro, 1990). Proportionately less filamentous blue-greens were sustained following treatment than in other seasons during similar extended dry weather periods and zooplankton appeared to be unaffected, so that any potential grazing should not be reduced. Schumaker *et al.* (1992) monitored zooplankton populations for several years following alum treatment of a U.S. lake. They found that all zooplankton numbers were reduced following treatment, but that cladocerans were the last to recover (two months). They suggested that although direct toxicity and physical entrainment of the floc were possible mechanisms of decline, the responses were attributed to the ecological effects of altered food sources and competitive interactions.

7.4 Conclusion

Aesthetic management options for these hypereutrophic ponds are limited. The use of aluminium sulphate demonstrated that a large algal biomass may be reduced with this technique. Relative to management efforts with herbicides (*e.g.,* Reglone A and copper sulphate) which have been used in the past in these detention ponds, treatment effectiveness, and reduced environmental concerns associated with precipitation agents, make the technique useful for these small systems. Larger and deeper SWDPs should maintain the benefits of treatment longer than those with shorter residence times and a well mixed water column. In the long-term, sediment removal is required to reduce the internal P loading burden and maintain operating efficiency. However, more information is required with regard to incorporation of P inactivation agents directly into the sediment structure as a means of retaining P binding capacity (Scott *et al.,* 1996).

Chapter 8

SUMMARY AND CONCLUSION

8.1 Summary

The use of artificial ponds for the temporary storage of urban stormwater runoff is a commonly used environmental engineering practice in North America. By releasing runoff at a rate slower than the initial generation rate, both on-line flood control and "primary" treatment of suspended solids are achieved. Primary treatment during operational phase is a function of particulate settling velocity distribution, dilution in the permanent pond volume, and the surface overflow rate. More extensive chemical modification is achieved during the permanent storage phase, when biologically mediated transformations are important.

By nature of their design function, urban SWDPs may attain hypereutrophic status within several years of construction. Well designed ponds may be capable of removing 70 to 90% of the TP load. On the one hand, the condition represents an efficient environmental protection feature, while on the other, a significant aesthetic management problem for municipal authorities operating within restricted budgets. Excessive macrophyte and phytoplankton growth, and potentially toxic blooms are of primary concern.

The focus of the work was to quantify the nutrient dynamics and associated phytoplankton cycles within SWDPs in the Province of Saskatchewan, Canada. A further aim was to identify potential interventions that may be appropriate for aesthetic water quality management. The project was based around a four year monitoring program, and this thesis presents data from three hypereutrophic SWDPs, aged 15 to 17 years old (two ponds in the City of Regina, one in the City of Saskatoon).

Chapter 1 - Introduction, reviewed some of the operational characteristics of urban SWDPs and their relation to the eutrophication process. A further overview was given in regard to options that exist to alleviate the symptomatic problems of excess nutrient loading and algal growth.

Protocols, analytical methods, and experimental procedures are summarized in *Chapter 2 - Materials and Methods*.

Physical aspects of the ponds and the results of four years monitoring data from 1992 to 1995 are summarized in *Chapter 3 - Physical, Chemical, and Biological Characteristics*. Oxygen depletion under winter ice-cover was shown to be typical of all three systems under four to five months of permanent winter ice cover. During the open water season, hypereutrophic conditions were typical of all three SWDPs. The semi-arid prairie climate and the intermittent external loading into the ponds predispose conditions that are favourable for the development of algal blooms. Typically, the systems are low in DIN and high in DIP, (DIN:DIP <3) and internal loading of P contributes significantly to water column P concentrations during permanent storage periods. Internal loads are greatly increased at high water temperatures which may reach 26°C. At such times, nitrogen may become further reduced by loss processes. Nutrients and algal biomass were shown to fluctuate considerably within short time scales, according to rainfall and other climatic conditions. Nitrogen-fixing and non-fixing blue-green algae frequently dominated the phytoplankton at densities that resulted in severe light limitation to competing groups. Chlorophyll *b* levels were often higher than chlorophyll *a* during these periods. Open water Secchi disk depths were often as low as 0.2 m.

Two neighbouring shallow ponds, Rochdale and Lakewood in Regina, proved to be somewhat different in their phytoplankton characteristics. Rochdale was typically dominated by *Oscillatoria* spp. throughout much of the open water season, while summer blooms in Lakewood were dominated by N-fixing *Anabaena* spp. However, these N-fixing blooms were "short-term" compared to *Oscillatoria* dominance which generally occurred over a more extended temperature range in all ponds. A greater N load was implicated for the Rochdale pond catchment, and in this regard 20 ha of arable land presently draining to this pond were identified, in addition to its lower pond volume to catchment area ratio (V_{rel}).

Significant intermittent disturbances associated with stormflows and turbulence (flushing biomass, loading DIN, reducing pH) tended to be capitalized by fast growing green algae in the short-term. Subsequently, re-establishment of blue-green dominance occurred as the closed system equilibrium was restored. Compared to the shallow Rochdale and Lakewood ponds, the effect of such disturbances were generally less marked in the deeper and stratifying Lakeview pond in Saskatoon. Average phytoplankton levels in the Lakeview pond were lower than those of the shallow ponds, although several N-fixing blooms occurred under quiescent conditions. In this regard, a greater sedimentary loss of organic P to an anaerobic profundal environment is permitted within the P cycle of Lakeview pond. Stratification of this pond may be enhanced by cool groundwater inflows, which percolate into the storm sewer system and enter the hypolimnion. Variable disruption and resuspension is periodically induced by stormflows in all ponds.

The total monitoring data set of phytoplankton composition was delineated at 25% intervals of relative blue-green to green dominance. Statistically significant differences were found between the low and high blue-green dominance and the inorganic-C concentration, which agrees with arguments concerning competitive advantages of blue-greens at high pH and reduced CO_2 availability. Significant differences were also found among the seston C:N:P ratios at high and low blue-green dominance, indicating a lower C content of blue-greens, and a higher N:P ratio.

The zooplankton community in all three ponds were mostly dominated by cyclopoid copepods, with only brief appearances of small-bodied cladocerans and calanoid copepods. Therefore, grazing appears to be of minor importance in the phytoplankton successions. Factors reducing numbers of grazing zooplankton species are not clear. The dominance of hypereutrophic systems by cyclopoid copepods has been reported in other studies. It is possible that predation, food quality and quantity, non-selective feeding, inability to withstand blue-green toxins, *etc.* are factors involved in loss of large-bodied cladocerans. In the case of lower V_{rel} ponds, high exchange events may flush large numbers of zooplankton from the system.

The sediment composition among the three ponds are compared in *Chapter 4 - Detention Pond Sediments and Internal Loading.* The deeper stratified Lakeview pond has the highest annual net sedimentation rate (~1.4 cm/yr), highest organic matter content (~20%), highest TP content (~1.6 mg/g dry wt), and lowest Fe content (~26 mg/g dry wt). Inorganic bound P was primarily contained in the calcium fraction in all ponds. The higher Ca content of the Lakeview sediments (~82 mg/g dry wt) are assumed to derive from the groundwater baseflows that the pond receives, while Fe may be lost from this system by displacement of anaerobic water under the thermocline during flushing. Overall about 11 to 15% of the TP was obtained from the extractable inorganic fraction in the ponds. Sediment TP profiles indicated that about 50% of the gross P input was recycled during the burial process. Sediment TP and organic matter were significantly correlated (R^2 >0.9) in all ponds. The surficial sediment (~1cm) TP and organic matter content were shown to vary significantly throughout the season. An organic-C/organic-P ratio of about 65 was typical, indicating that P is more rapidly mineralized than C. An inverse relationship between the surficial sediment TP and the internal load of P was also apparent in the dry year of 1992. Data confirmed that with temperature, autochthonous organic P sedimentation is the key determinant of the gross P release rate. In 1992, net internal loading of P in the range of 30 to 45 mg/m^2/day was measured during peak summer temperatures for the shallow Rochdale and Lakewood ponds. Laboratory incubation of intact Rochdale cores at 20 °C gave an average release rate of 14.2 mg P/m^2/day over 14 days under anaerobic conditions, but initially, release rates of double this value were measured in some cores. As the surface sediment became oxidized, aerobic incubation of both Rochdale and Lakewood cores resulted in net uptake from the reservoir media in both cases. The results indicated that Fe is undersaturated with P in Rochdale and Lakewood ponds, and sufficient to suppress P release under the temperature, pH, and redox conditions of the experiment. Anaerobic incubations of Lakeview pond cores from 5 to 20 °C gave average release rates from 5 to 16 mg/m^2/day. Based on interstitial gradients and reservoir DIP, Ficks law predicted the release rates in these experiments well, using literature values for the whole sediment P diffusion coefficient.

Compiled regional rainfall statistics, pond design, and watershed characteristics are summarized in *Chapter 5 - Nutrient Loading and Theoretical Pond Operational Performance*. Theoretical P removal curves for the ponds were generated from this data using the model of Walker (1987). The theoretical long-term average P removal efficiencies were in the range 65 to 80% with Lakeview>Lakewood>Rochdale. The results of a more detailed stormwater runoff study for Lakeview pond in 1994 and 1995 were summarized in relation to groundwater baseflows and internal loading of N and P.

A two season average of about 0.25 to 0.3 kg TP/ha impervious/0.58 yr (~40-50% dissolved) from 250 mm of precipitation was calculated. External P loads may contain in excess of 1 mg/L TP, with a significant proportion as dissolved and bioavailable P. Most of the calibration storms measured in 1994 to 1995 were smaller events, and a seasonal mean concentration of about 0.2 mg TP/L was generated from the calibrated SWMM model by Raymond (1997).

Typically, the DIN/DIP ratios measured in stormwater runoff were below the stoichiometric Redfield average for algal composition. Lakeview receives a massive proportion of the total external DIN load from baseflows (160 mg/m^2/day NO_3-N), while total averaged external DIP load (2.02 mg P/m^2/day) was about 4 to 5 times less than the estimated release rate from the sediments at average hypolimnion temperatures (~10 mg P/m^2/day).

A light limitation equation for algal growth was input with 1994 Lakeview pond monitoring data for periods in which sediment traps were operated weekly below the thermocline. The set parameters were able to account for the equilibrium chlorophyll *a* concentration reasonably well, but underestimated some of the sediment trap C flux estimates. Although the system stratified, the reliability of trap data was questionable on account of the influence of resuspension by stormflows. The estimated average of net productivity during the period was 2.5 g C/m^2/day.

Chapter 6 - Addition of Inorganic Nitrogen to alter the N:P Ratio in a Nitrogen Limited System: Effects on Phytoplankton Biomass and Composition, summarized the results of ammonium nitrate additions to an N-limited SWDP, typically dominated by summer blooms of N-fixing blue-green algae. Weekly or two weekly additions were carried out at volumetric additions of 2.5 to 3.5 g/m^3 from May until July, 1994. During the N addition period, DIN:DIP ratios fell below 7 on only three occasions, while in other seasons DIN:DIP ratios were usually <0.5. Stimulation of primary productivity was indicated by movement of DIP into the particulate fraction following N additions. However, DIP was reduced <10 µg/L during only two periods. Preferential ammonium uptake was evident, and on some dates large removals occurred within 24 hours of additions. Warming water temperatures began to increase the internal P load in July, so that neither N or P limitation resulted. N additions ceased in mid-July, and although DIN became low, a dense bloom of non N-fixing blue-green algae developed (*Microcystis*). Quiescent dry weather conditions and recycling of accumulated organic-N sustained the bloom. Nutrient sufficiency, water column stability, and increasing light limitation were proposed as predisposing factors. Nitrogen supplements may prevent formation of N-fixing blooms, but will not prevent non-fixing blooms in shallow systems where P cannot be made limiting. There was no evidence of an increase in the numbers of herbivorous zooplankton during the period of complete green/diatom algal dominance from April to July, compared to the cycles measured in other seasons.

Chapter 7 - Use of Aluminium Sulphate to Reduce Phytoplankton Biomass in a Hypereutrophic Stormwater Detention Pond, describes a full-scale trial with aluminium sulphate in a shallow SWDP carried out in June 1995. Significant water quality improvements prevailed until several very large precipitation events occurred six weeks later. During the period of improved water quality, DIP was reduced from >60 µg/L to <10 µg/L, and a reduced algal biomass resulted. Pond volume exchanges of 20 to 30% took place, and water temperatures reached their seasonal peak during this period. Within two weeks of the addition, sediment surface oxidation was apparent. Reduced productivity and associated water column/sediment oxygen demand facilitated oxidation. No increase in sediment Al content was evident, and the floc had apparently been dispersed by wind and inflow turbulence. Therefore, sediment oxidation was implicated as a major factor in maintaining low P. Management options for these SWDPs are limited, but the use of P precipitation agents proved to be an efficient technique for improving water quality during dry weather periods.

8.2 Conclusion

The present hypereutrophic conditions represent an imbalance between the treatment and the aesthetic function of these urban SWDPs. External DIP loads from urban environments will always be higher than threshold concentrations for nuisance algal growth (*i.e.,* >25 µg/L). Initially, adsorption of P by the clay liners of newly constructed ponds is exhausted and physical burial proceeds. In each of the three SWDPs studied here, 3000 to 5500 metric tonnes of fresh sediment accumulation occurred after 14 to 16 years. Aside from complete sediment removal, which is costly, retrofit measures are limited. Biomanipulation measures, such as piscivore introduction to reduce grazing on herbivorous zooplankton, may produce only limited benefits. Such measures stand a better chance of success in higher V_{rel} ponds such as Lakeview. In the face of continued external and internal nutrient loading, and high flushing volumes, seasonal developments to both green and blue-green blooms will be unavoidable. Comprehensive nutrient budgets remain to be evaluated.

Repeated use of precipitation agents by manual means is inconvenient. However, the method is less expensive than flow controlled injector applications to inflows, and can be used selectively. More research is required in regard to direct injection of precipitation agents into the sediment structure. Current iron levels in the sediments of the stratified Lakeview pond are insufficient to bind P under aerobic conditions. Therefore, summer aeration could potentially increase productivity by enhancing both the mineralization and mixing rate.

Emerging designs of SWDPs now favour multiple stage systems. A primary sedimentation basin may be linked with a recirculating artificial wetland /hydroponic bed system, with a final basin designated as the main recreational area. Adequate maintenance access, and deeper basins with aeration facilities for overwintering increase the opportunity for a more ecologically balanced and useful stormwater detention facility. Such a system is presently (1997) under construction in the City of Saskatoon.

REFERENCES

Ahlgren, I. (1977). Growth of *Oscillatoria agardhii* in chemostat culture. 1. Nitrogen and phosphorus requirements. *Oikos* **29**:209-224.

Ahlgren, I., Frisk, T., and Kamp-Nielsen, L. (1988). Empirical and theoretical models of phosphorus loading, retention, and concentration *vs.* lake trophic state. *Hydrobiologia* **170**:285-303.

Ambrose, R.B., Wool, T.A., and Martin, J.L. (1993). *Water Quality Analysis Simulation Program, WASP5, Part A: Model Documentation Users Manual.* Environmental Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Athens, GA.

American Public Health Association [APHA] (1985). *Standard Methods for the Examination of Water and Wastewater.* 16th Edition, Clesceri, L.S., Greenberg, A.E., and Trussel, R.R. (eds). Water Pollution Control Federation, Washington, DC.

Andersen, F.Ø. and Jensen, H.S. (1992). Regeneration of inorganic phosphorus and nitrogen from decomposition of seston in a freshwater sediment. *Hydrobiologia* **228**:71-81.

Andersen, J.M. (1975). Influence of pH on release of phosphorus from lake sediments. *Arch. Hydrobiol.* **76**(4):411-419.

Andersen, J.M. (1982). Effect of nitrate concentration in lake water on phosphate release from lake sediment. *Water Research* **16**:1119-1126.

Andersson, G., Berggren, H., Cronberg, G., and Gelin, C. (1978). Effects of plantivorous and benthivorous fish on organisms and water chemistry in eutrophic lakes. *Hydrobiologia* **59**:9-15.

Athayde, D.N., Shelly, E.D., Driscoll, D., Gaboury, D., and Boyd, G. (1983). *Results of the Nationwide Urban Runoff Program - Volume 1 - Final Report.* Water Planning Division, U.S. Environmental Protection Agency, Washington, DC, 191 p.

Atmospheric Environment Service [AES] (--1996). Environment Canada Atmospheric Monitoring Division, AES Canada, Regina Airport, Regina, SK.

Auer, M.T., Johnson, N.A., Penn, M.R., and Effler, S.W. (1993). Measurement and verification of rates of sediment phosphorus release for a hypereutrophic urban lake. *Hydrobiologia* **253**:301-309.

Babin, J. and Prepas, E.E. (1985). Modelling winter oxygen depletion rates in ice-covered temperate zone lakes in Canada. *Can. J. Fish. Aquat. Sci.* **42**:239-249.

Babin, J., Prepas, E.E., and Zhang, Y. (1992). Application of lime and alum to stormwater retention lakes to improve water quality. *Wat. Pollut. Res. J. Can.* **27**(2):365-381.

Bailey-Watts, A.E., Kirika, A., May, L., and Jones, D.H. (1990). Changes in phytoplankton over various time scales in a shallow, eutrophic: The Loch Leven experience with special reference to the influence of flushing rate. *Freshwater Biol.* **23**:85-111.

Baird, D.J., Gates, T.E., and Davies, R.W. (1987). Oxygen conditions in two prairie pothole lakes during winter ice cover. *Can. J. Fish. Aquat. Sci.* **44**:1092-1095.

Bannister, T.T. (1974*a*). Production equations in terms of chlorophyll concentration, quantum yield, and upper limit to production. *Limnol Oceanogr.* **19**:1-12.

Bannister, T.T. (1974*b*). A general theory of steady state phytoplankton growth in a nutrient saturated mixed layer. *Limnol. Oceanogr.* **19**:13-30.

Barica, J. (1990). Seasonal variability of N:P ratio in eutrophic lakes. *Hydrobiologia* **191**:97-103.

Barica, J. (1992). Sustainable management of urban lakes: A new environmental challenge. *Wat. Pollut. Res. J. Can.* **27**(2).211-219.

Barica, J., Kling, H., and Gibson. J. (1980). Experimental manipulation of algal bloom composition by nitrogen addition. *Can. J. Fish. Aquat. Sci.* **37**:1175-1183.

Barica, J. and Mathias, J.A. (1979). Oxygen depletions and winterkill risk in small prairie lakes under extended ice cover. *J. Fish. Res. Bd. Can.* **36**:980-986.

Barica, J. and Mur, R.L.. (eds) (1980). *Hypertrophic Ecosystems*. Developments in Hydrobiology. Volume 2, W. Junk Publishers, The Hague.

Berner, R.A. (1980). *Early Diagenesis, a Theoretical Approach.* In: Holland, H.D. (ed), Princeton University Press, Princeton, NJ, USA.

Bloesch, J. and Burns, N.M. (1980). A critical review of sedimentation trap technique. *Schweiz. Z. Hydrol.* **42**:15-55.

Bloesch, J. and Sturm, M. (1984). Settling flux and sinking velocities of particulate phosphorus [PP] and particulate organic carbon [POC] in Lake Zug, Switzerland. In: Sly, P.G. (ed), *Sediments and Water Interactions.* Springer-Verlag, New York, ISBN 0-387-96293-X, pp. 481-490.

Boers, P.C.M. (1991). *The Release of Dissolved Phosphorus From Lake Sediments.* PhD Thesis, Limnological Institute and the Institute for Inland Water Management and Waste Treatment, Lelystad, The Netherlands.

Boers, P.C.M. and Van Hese, O. (1988). Phosphorus release from the peaty sediments of the Loosedrecht Lakes (The Netherlands). *Water Res.* **22**:355-363.

Boström, B., Andersen, J.M., Fleischer, S., and Jansson, M. (1988). Exchange of phosphorus across the sediment-water interface. *Hydrobiologia* **170**:229-244.

Boström, B., Jansson, M., and Forsberg, C. (1982). Phosphorus release from lake sediments. *Arch. Hydrobiol. Beih.* **18**:5-59.

Bouldin, D.R., Johnson., R.L., Burda, C., and Kao, C.W. (1974). Losses of inorganic nitrogen from aquatic systems. *J. envir. Qual.* **3**:107-114.

Brandl, Z., and Fernando, C.H. (1975). Food consumption and utilization in two freshwater cyclopoid copepods (*Mesocyclops edax* and *Cyclops vicinus*). *Int. Revue ges. Hydrobiol.* **60**(4):471-494.

Browman, M.G., Harris, R.F., Ryden, J.C., and Syers, J.K. (1979). Phosphorus loading from urban stormwater runoff as a factor in lake eutrophication: 1. Theoretical considerations and qualitative aspects. *J. envir. Qual.* **8**(4):561-565.

Brownlee, B.G. and Murphy, T.P. (1983). Nitrogen fixation and turnover in a hypereutrophic prairie lake. *Can. J. Fish. Aquat. Sci.* **40**:1853-1860.

Canfield, D.E. and Bachmann, R.W. (1981). Prediction of total phosphorus concentrations, chlorophyll *a*, and Secchi depths in natural and artificial lakes. *Can. J. Fish. Aquat. Sci.* **38**:414-423.

Cappenberg, Th.E. and Verdouw, H. (1982). Sedimentation and breakdown kinetics of organic matter in the anaerobic zone of Lake Vechten. *Hydrobiologia* **95**:165-179.

Carlson, R.E. (1977). A trophic state index for lakes. *Limnol. Oceanogr.* **22**:361-369.

Carr, N.G. and Whitton, B.A. (1982). *The Biology of Cyanobacteria.* Botanical Monographs, Volume 19, Blackwell Scientific Publications, ISBN 0-520-04717-6.

Chapra, S.C. (1975). Comment on "An empirical method of estimating the retention of phosphorus in lakes" by W.B. Kirchner and P.J. Dillon. *Wat. Resourc. Res.* **11**:1033-1034.

Christofi, N., Preston, T., and Stewart, W.D.P (1980). Zones of nitrification in eutrophic freshwater lakes. *Soc. gen. Microbiol. Q.* **7**:107.

Claesson, A. and Ryding, S. -O. (1977). Nitrogen - a growth limiting nutrient in eutrophic lakes. *Prog. Wat. Technol.* **8**:291-299.

Clasen, J. and Bernhardt, H. (1980). *OECD cooperative programme for Monitoring of Inland Waters - Regional Project - Shallow Lakes and Reservoirs.* Organization for Economic Cooperative Development [OECD], Paris, France.

Codd, G.A., Bell, S.G., and Brooks, W.P. (1989). Cyanobacterial toxins in water. *Wat. Sci. Technol.* **21**:1-13.

Cooke, G.D. and Kennedy, R.H. (1989). *Water Quality Management for Reservoirs and Tailwaters; In-Reservoir Water Quality Management Techniques, Report 1.* Technical Report E-89-1, U.S. Army Engineer Waterways Experimental Station, Vicksburg, MS, 183 p.

Cooke, G.D., Welch, E.B., Martin, A.B., Fulmer, D.G., Hyde, J.B., and Schrieve G.D. (1993*a*). Effectiveness of Al, Ca, and Fe salts for control of internal phosphorus loading in shallow and deep lakes. *Hydrobiologia* **253**:323-335.

Cooke, G.D., Welch, E.B., Peterson, S.A., and Newroth, P.R. (1993*b*). *Restoration and Management of Lakes and Reservoirs.* 2nd Edition, Lewis Publishers, ISBN 0-87371-397-4.

Crowther, J. and Evans, J. (1980). *Dual Channel for Determination of Dissolved Organic and Inorganic Carbon*. Ontario Ministry of Environment, Ottawa, ON. February, 1896. NAQUADAT No. 06107L.

Danen-Louwerse, H., Lijklema, L., and Coenraats, M. (1993). Iron content of sediment and phosphate adsorption properties. *Hydrobiologia* **253**:311-317.

De Bernardi, R. and Giussani, G. (1990). Are blue-green algae a suitable food source for zooplankton? An overview. *Hydrobiologia* **200/201**:29-41.

De Vitre, R.R., Sulzberger, B., and Buffle, J. (1994). Transformations of iron at redox boundaries. In: *Chemical and Biological Regulation of Aquatic Systems*. J. Buffle and R.R. De Vitre (eds), Lewis Publishers, ISBN 0-87371-448-2, pp. 89-135.

den Oude., P.J. and Gulati, R.D. (1988). Phosphorus and nitrogen excretion rates of zooplankton from the eutrophic Loosdrecht lakes, with notes on other P sources for phytoplankton requirements. *Hydrobiologia* **169**:379-390.

Dillon, P.J. and Rigler, F.H. (1974*a*). The phosphorus-chlorophyll relationship in lakes. *Limnol. Oceanogr.* **19**:767-773.

Dillon, P.J. and Rigler, F.H. (1974*b*). A test of a simple nutrient budget model predicting the phosphorus concentration in lake water. *J. Fish. Res. Bd. Can.* **32**:1519-1531.

Driscoll, E.D. (1983). Performance of runoff detention basins for control of urban runoff quality. In: *Proceedings of the International Symposium on Urban Hydrology, Hydraulics, and Sediment Control*, July 1993, University of Kentucky, Kentucky, U.S.A.

Driscoll, E.D. (1986). Review of papers on detention, retention, wetlands, and other treatment. In: *Urban Runoff Quality, its Impacts and Quality Enhancement Technology*. Engineering Foundation Conference, June 22-27, 1986, Henniker, NH, U.S.A.

Driscoll, C.T., Baker, J., Bisogni, J., and Schofield, C. (1980). Effects of aluminium speciation on fish in dilute, acidified waters. *Nature* **184**:161-164.

Driscoll, C.T. and Schecher, W.D. (1990). The chemistry of aluminium in the environment. *Env. Geochem. Health* **12**:28-49.

Duda, A.M. (1985). Environmental and economic damage caused by sediment from agricultural nonpoint sources. *Water Resour. Bull.* **21**:225-234.

Duda, A.M., Iwanski, M.L., Johnson, R.J., and Jaksch, J.A. (1987). Numerical standards for managing lake and reservoir water quality. *Lake and Reservoir Management* **3**:1-28.

Edmonson, W.T. (1991). *The uses of ecology: Lake Washington and beyond.* University of Washington Press, Washington, DC, ISBN 0-295-97024-3, 329 p.

Edmonson, W.T. (ed) (1966). *Freshwater Biology.* 2nd Edition, Wiley Publishers, New York, pp. 587-815.

Ellis, C.R. and Stefan, H.G. (1989). Oxygen demand in ice covered lakes as it pertains to winter aeration. *Wat. Resourc. Bull.* **25**(6):1169-1176.

Evans, G.Y. and Taylor, F.Z.R. (1980). Phytoplankton accumulation in Langmuir cells. *Limnol. Oceanogr.* **25**:840-845.

Field, R. and Pitt, R.E. (1990). Urban storm-induced discharge impacts: United States Environmental Protection Agency Research Program review. In: Urban Stormwater Quality and Ecological Effects Upon Receiving Waters. *Wat. Sci. Technol.* **22**(10/11):1-7.

Fogg, G.E. (1975). *Algal Cultures and Phytoplankton Ecology.* 2nd Edition. University of Wisconsin Press, London. ISBN 0-299-06760-2.

Forsberg, C. (1989). Importance of sediments in understanding the nutrient cyclings in lakes. *Hydrobiologia* **176/177**:263-277.

Forsberg, C. and Ryding, S.O. (1980). Eutrophication parameters and trophic state indices in 30 Swedish waste-receiving lakes. *Arch. Hydrobiol.* **89**:189-207.

Forsberg, C., Ryding, S.O., Claesson, A., and Forsberg, A. (1978). Water chemical analyses and /or algal assay - Sewage effluent and polluted lake water studies. *Mitt. Int. Ver. Limnol.* **21**:352-363.

Forsberg, C. and Shapiro, J. (1982). The effects of artificial circulation on algal populations. In: Shapiro, J., Forsberg, B., Lamarra, V., Lindmark, G., Lynch, M., Smeltzer, E., and Zoto, G. (1982). *Experiments and Experiences in Biomanipulation Studies of Biological Ways to Reduce Algal Abundance and Eliminate Blue-greens.* U.S. Environmental Protection Agency, Report No. 600/3-82-096, pp. 46-128.

Frevert, T. (1979). The pH-redox concept in natural sediment-water systems: Its role in controlling phosphorus release from lake sediments. *Arch. Hydrobiol.* **55**:278-297.

Frodge, J.D., Thomas, G.L., and Pauley, G.B. (1991). Sediment phosphorus loading beneath dense canopies of aquatic macrophytes. *Lake and Reservoir Management* **7**(1):61-71.

Fulton, R.S. (1988*a*). Grazing on filamentous algae by herbivorous zooplankton. *Freshwater Biology* **20**:263-271.

Fulton, R.S. (1988*b*). Resistance to blue green algal toxins by *Bosmina longirostris*. *J. Plankton. Res.* **10**(4):771-778.

Fulton R.S. and Paerl, H.W. (1987). Toxic and inhibitory effects of the blue-green alga *Microcystis aeruginosa* to herbivorous zooplankton. *J. Plankton Res.* **9**:837-855.

Furumai, H. and Ohgaki, S. (1989). Adsorption/desorption of phosphorus by lake sediments under anaerobic conditions. *Water Research* **23**(6):677-683.

Furnass, T.I. (1979). Laboratory experiments on prey selection by perch fry (*Perca fluviatilis*). *Freshwat. Biol* **9**:33-43.

Grimm, M.P. and Backx, J.J.G.M. (1990). Restoration of shallow eutrophic lakes and the role of northern pike, aquatic vegetation, and nutrient concentration. *Hydrobiologia* **200/201**:557-566.

Golterman, H.L., Clymno, R.S., and Ohnstad, M.A.M. (1978). *Methods for the Chemical Analysis of Fresh Water.* International Biological Programme Handbook, Blackwell Scientific Publications, Oxford, U.K. 213 p.

Gophen, M. (1990). Biomanipulation: retrospective and future development. *Hydrobiologia* **200/201**:1-11.

Gunnison, D. and Alexander, M. (1975). Resistance and susceptibility of algae to decomposition by natural microbial communities. *Limnol. Oceanogr.* **20**(1):64-70.

Håkanson, L. and Jansson, M. (1983). *Principles of Lake Sedimentology.* Springer-Verlag, Berlin. ISBN 0-387-12645-7.

Haney, J.F. (1987). Field studies on zooplankton-cyanobacteria interactions. *N.Z.J. Marine and Freshwater Res.* **21**:467-475.

Hanson, M.A. and Butler, M.G. (1994). Responses of plankton, turbidity, and macrophytes to biomanipulation in a shallow prairie lake. *Can. J. Fish. Aquat. Sci.* **51**(5):1180-1188.

Harper, D. (1992). *Eutrophication of Freshwaters: Principles, Problems and Restoration.* Chapman and Hall Publishers, London, 327 p.

Harrison, R.M. and Wilson, S.J. (1985). The chemical composition of highway drainage waters, II: Chemical associations of metals in suspended sediments. *Sci. Total Envir.* **43**(1):79-88.

Hayes, C.R., Clark, R.G., Syent, R.F., and Redshaw, C.J. (1984). The control of algae by chemical treatment in a eutrophic water supply reservoir. *J. Inst. Water Eng. Scientists* **38**(2):149-162.

Healey, F.P. (1973). Inorganic nutrient uptake and deficiency in algae. *CRC Crit. Rev. Microbiol.* **3**:69-113.

Healey, F.P. and Hendzel, L.L. (1978). Physiological Indicators of nutrient deficiency in lake phytoplankton. *Can. J. Fish. Aquat. Sci.* **37**:442-453.

Hegemann, D.A., Johnson, A.H., and Keenan, J.D. (1983). Determination of algal-available phosphorus on soil and sediment: A review and analysis. *J. envir. Qual.* **12**(1):12-16.

Helfrich, L.A. (1976). *Effects of Predation by Fathead Minnows (Pimephales promelas) on Planktonic Communities in Small, Eutrophic Ponds.* PhD Thesis, Michigan State University, E. Lansing, MI, U.S.A.

Hessen, D.O., Nilssen, J.P., and Erikson, T.O. (1986). Food size spectra and species replacement within herbivorous zooplankton. *Int. Rev. ges. Hydrobiol.* **71**:1-10.

Hieltjes, A.H.M., and Lijklema, L. (1980). Fractionation of inorganic phosphates in calcareous Sediments. *J. envir. Qual.* **9**(3):405-407.

Holtan, H., Kamp-Nielsen, L., and Stuanes, A.O. (1988). Phosphorus in soil, water, and sediment: an overview. *Hydrobiologia* **170**:19-34.

Huber, W.C. and Dickinson, R.E. (1988). *Stormwater Management Model: Users manual Version IV.* U.S. Environmental Protection Agency [USEPA], Athens, GA., pp.109, 111-116, 143-160.

Humphries, S.E. and Lyne, V.D. (1988). Cyanophyte blooms: the role of cell buoyancy. *Limnol. Oceanogr.* **33**:79-91.

Jansson, M. (1986). Nitrate as a catalyst for phosphorus mobilization in sediments. In: P.G. Sly (ed), *Sediments and Water Interactions*, Springer-Verlag, New York, pp. 387-391.

Janus, L.L. and Vollenweider, R.A. (1981). *The OECD cooperative program on eutrophication. Canadian contribution summary report*. Scientific series No. 131, Canada Centre for Inland Waters, Burlington, ON.

Jensen, H.S. and Andersen, F. Ø. (1992). Importance of temperature, nitrate, and pH for phosphate release from aerobic sediments of four shallow, eutrophic lakes. *Limnol. Oceanogr.* **37**(3):577-589.

Jensen, H.S., and Andersen, F. Ø. (1990). Impact of nitrate and blue-green algal abundance on phosphorus cycling between sediment and water in two shallow, eutrophic lakes. *Verh. int. Verein. Limnol.* **24**:224-230.

Jewell, W.J. and McCarty, P.L. (1971). Aerobic decomposition of algae. *Envir. Sci. Technol.* **5**:1023-1031.

Jones, J.R. and Bachman, R.W. (1976). Prediction of phosphorus and chlorophyll in lakes. *J. Wat. Pollut. Control. Fed.* **48**:2176-2182.

Jones, C.R. and Redfield, G.W. (1984). Effects of urban stormwater runoff on reservoir phytoplankton. *Verh. int. Verein. Limnol.* **22**:1486-1492.

Jørgensen, S.E. (1980). *Lake Management.* Water Development, Supply, and Management, Volume 14, Pergamon Press, ISBN 0-08-022432-6, 167 p.

Jørgensen, S.E., Kamp-Nielsen, L., and Jørgensen, L.A. (1986). Examination of the generality of eutrophication models. *Ecol. Modelling* **32**:251-266.

Jørgensen, S.E. and Vollenweider, R.A. (eds) (1989). *Guidelines of Lake Management.* International Lake Environment Committee, United Nations Environment Program, 3-4-22 Kyomachi, Otsu, Shiga 520, Japan, 199 p.

Kaspar, H.F. (1985). The denitrification capacity of sediment from a hypereutrophic prairie lake. *Freshwat. Biol.* **15**:449-453.

Kelts, K., and Hsü, K.J. (1978): Freshwater carbonate precipitation. In: Lerman, A. (ed), *Lakes, Chemistry, Geology and Physics*. Springer-Verlag, New York, 363 p.

Klapwijk S.P., and Bruning, C, (1984). Available phosphorus in the sediments of eight lakes in the Netherlands. In: *Sediments and Water Interactions*, P.G. Sly (ed), Springer-Verlag, New York, ISBN 0-387-96293-X, pp. 391-398.

Knauer, D.R. (1975). The effect of urban runoff on phytoplankton ecology. *Verh. int. Verein. Limnol.* **19**:893-903.

Koroleff, F. (1972). Determination of total nitrogen in natural waters by means of persulphate oxidation. In: *New Baltic Manual with Methods for Sampling and Analyses of Physical, Chemical and Biological Parameters*, S.R. Carlberg (ed), Charlottenlund, International Council for the Exploration of the Sea, pp. 197-173.

Köschel, R., Benndorf, J., Proft, G., and Recknagel, F. (1983). Calcite precipitation as a natural control mechanism in eutrophication. *Arch. Hydrobiol.* **98**:380-408.

Krom, M.D. and Berner, R.A. (1980). The diffusion coefficients of sulphate, ammonium, and phosphate ions in anoxic marine sediments. *Limnol. Oceanogr.* **25**:327-337.

Larsen, D.P. and Mercier, H.T. (1976). Phosphorus retention capacity of lakes. *J. Fish. Res. Bd. Can.* **33**:1742-1750.

Lathrop, R.C. (1988). Evaluation of whole lake nitrogen fertilization for controlling blue-green algal blooms in a hypereutrophic lake. *Can. J. Fish. Aquat. Sci.* **45**:2061-2027.

Leonardson, L. and Bengtsson, L. (1978). Effects of sewage diversion in Lake Södra Bergundasjön. II. Phytoplankton changes and the role of nitrogen fixation. *Verh. int. Verein. Limnol.* **20**:2701-2707.

Leonardson, L. and Ripl, W. (1980). Control of undesirable algae and induction of algal successions in hypereutrophic lake ecosystems. In: J. Barica and L.R. Muir (eds), *Hypereutrophic Ecosystems. Developments in Hydrobiology* 2, W. Junk Publishers, Netherlands, pp. 57-65.

Li, Y. and Gregory, S. (1974). Diffusion of ions in sea water and deep sea sediments. *Geochim. Cosmochim. Acta* **38**:703-714.

Liao, C. F.H. and Lean, D.R.S. (1978). Nitrogen transformations within the trophogenic zone of lakes. *J. Fish. Res. Bd. Can.* **35**:1102-1108.

Lijklema, L. (1980). Interaction of orthophosphate with iron (III) and aluminium hydroxides. *Env. Sci. Technol.* **14**(5):537-539.

Lijklema, L. and Peeters, J.H.C. (1979). Comparative studies on eutrophication in large scale model reservoirs. *State of the art in Ecological Modelling* **7**:603-615.

Lindh, G. (1987). Urban catchment comprehensive system planning. In: *Proceedings of the IV International Conference in Urban Storm Drainage.* École Polytechnique Fédérale, Lausanne, France, pp. 399-402.

Löfgren, S. (1987). *Phosphorus Retention in Sediments - Implications for Aerobic Phosphorus Release in Shallow Lakes.* Acta Universitatis Upsaliensis, Comprehensive Summaries of Uppsala Dissertations from the Faculty of Science,100. ISBN 91-554-2079-6.

Long, E.B. (1975). The interaction of phytoplankton and the bicarbonate system. PhD Thesis, Kent State University. Cited in: Shapiro, J. (1990). Current beliefs regarding dominance by blue-greens: The case for the importance of CO_2 and pH. *Verh. int. Verein. Limnol.* **24**:38-54.

Lopez, P. and Morgui, J.A. (1992): Phosphate and calcium carbonate saturation in a stratified coastal lagoon. *Hydrobiologia* **228**:55-63.

Lorenzen, M. (1980). Use of chlorophyll-Secchi disk relationships. *Limnol. Oceanogr.* **25**(2):371-372.

Mackareth, F.J.H., Heron, J., and Talling, J.F. (1978). *Water Analysis: Some Revised Methods for Limnologists.* Freshwater Biological Association Scientific Publication, No. 36, 121 p.

MacKinney, G. (1941). Absorbtion of light by chlorophyll solutions. *J. Biol. Chem.* **140**:315-322.

Makepeace, D.K., Smith, D.W., and Stanley, S.J. (1995). Urban stormwater quality: summary of contaminant data. *Crit. Rev. Envir. Sci. and Technol.* **25**(2):93-139.

Marsden, W.W. (1989). Lake restoration by reducing external phosphorus loading: The influence of sediment phosphorus release. *Freshwat. Biol.* **21**:139-162.

Matisoff, G., Lindsay, A.H.S., Matis, S., and Soster, F.M. (1880). Trace mineral equilibria in Lake Eire sediments. *J. Great Lakes Res.* **6**:353-366

Megard, R.O., Settles, J.C., Boyer, H.A., and Combs, W.S. Jr. (1980). Light, Secchi disks, and trophic states. *Limnol. Oceanogr.* **25**(2):373-377.

McNeil, D.D. and Weber, B.T. (1994). Stormwater basin review study. In: *Proceedings of the Western Canadian Water and Wastewater Association [WCWWA] Conference*, October 4-7, 1994, Winnipeg, MB, pp. 71-84.

Miura, T. (1990). The effects of planktivorous fishes on the plankton community in a eutrophic lake. *Hydrobiologia* **200/201**:567-579.

Molongowski, J.J. and Klug, M.J. (1980). Anaerobic metabolism of particulate organic matter in the sediments of a hypereutrophic lake. *Freshwat. Biol.* **10**:507-518.

Morin, A., Hambright, K.D., Hairson, G., Sherman, D.M., and Howarth, R.W. (1991). Consumer control of gross primary production in replicate freshwater ponds. *Verh. int. Verein. Limnol.* **24**:1512-1516.

Moss, B., Balls, H., Irvine, K., and Stansfield, J. (1986). Restoration of two lowland lakes by isolation from nutrient rich water sources with and without removal of sediment. *J. Appl. Ecol.* **23**:391-414.

Murphy, T.P. and Brownlee, B.G. (1981). Ammonium volatilization in a hypereutrophic prairie lake. *Can. J. Fish. Aquat. Sci.* **38**:1035-1039.

Murphy,T.P., Hall, K.G., and Northcote, T.G. (1988). Treatment of a hardwater lake to reduce eutrophication. *Lake and Reservoir Management* **4**:51-62.

Nicholls, K.H. and Dillon, P.J. (1980). An evaluation of phosphorus-chlorophyll-phytoplankton relationships for lakes. *Int. Revue ges. Hydrobiol.* **63**(2):141-154.

Niemczynowicz, J. (1991). Urban growth and pollution control - call for a new realism. *Stockholm Water Prize and Symposium Newsletter* **2**:3-4.

Nix, S. (1990). Non-point source pollution models: An overview. In: *Urban Non-Point Source Pollution and Stormwater Management Symposium*, University of Kentucky, Lexington, Ky, USA. Cited in: Wanielista, M.P., and Yousef, Y.A. (1992). *Stormwater Management.* John Wiley and Sons Inc., N.Y. ISBN 0-471-57135-0.

Nurnberg, G.K. (1984). The prediction of internal phosphorus loads in lakes with anoxic hypolimnia. *Limnol. Oceanogr.* **29**:111-124.

Nurnberg, G.K. (1988). Prediction of phosphorus release rates from total and reductant-soluble phosphorus in anoxic lake sediments. *Can. J. Fish. Aquat. Sci.* **45**:453-462.

Overbeck, J. (1989). Ecosystem Concepts. In: Jørgensen, S.E. and Vollenweider, R.A. (eds), *Guidelines of Lake Management*, Volume 1. International Lake Environment Committee and United Nations Environment Programme, 3-4-22 Kyomachi, Otsu, Shiga 520, Japan, pp. 19-36.

Parr, W. and Clarke, S. (1992). *Methods for Controlling Phytoplankton, with Particular Reference to Cyanobacteria, and Sampling Guidelines for the Water Industry.* Report No. FR 0248, Foundation for Water Research, Allen House, The Listons, Marlow, Bucks, U.K. 33 p.

Pettersson, K. (1984). The fractional composition of phosphorus in lake sediments of different characteristics. In: Sly, P.G. (ed), *Sediments and Water Interactions*, Springer-Verlag, New York, NY, ISBN 0-387-96293-X, pp. 149-155.

Porter, K.G. (1977). The plant-water interface in freshwater ecosystems. *Am. Sci.* **65**:159-160.

Prairie, Y.T., Duarte, C.M., and Kalff, J. (1989). Unifying nutrient-chlorophyll relationships in lakes. *Can. J. Fish. Aquat. Sci.* **46**:1176-1182.

Prescott, G.W. (1962). *Algae of the Western Great Lakes Area.* WM. C. Brown Publishers, Iowa, U.S.A. ISBN 0-697-04552-8, 977 p.

Preston, T., Stewart, W.D.P., and Reynolds, C.S. (1980). Bloom forming cyanobacterium *Microcystis aeruginosa* overwinters on a sediment surface. *Nature* **288**:365-367.

Prych, E.A. and Ebbert, J.C. (1987). Effects of Street Sweeping on Runoff Quality. In: *Proceedings of the IV International Conference in Urban Storm Drainage.* École Polytechnique Fédérale, Lausanne, France, pp. 53-54.

Quaak , M., van der Does, J., Boers, P., and van der Vlugt, J. (1993). A new technique to reduce internal phosphorus loading by in-lake phosphate fixation in shallow lakes. *Hydrobiologia* **253**:337-344.

Rast, W. and Lee, G.F. (1983). Nutrient loading estimates for Lakes. *J. envir. Engng* **109**(2):502-517.

Raymond, B.E. (1997). *SWMM Stormwater Modelling in Saskatoon.* MSc Thesis, Department of Civil Engineering, University of Saskatchewan, Saskatoon, SK. (pending)

Raymond, B.E., Kells, J.A., and Putz, G. (1995). PC-SWMM modelling in the City of Saskatoon. In: *Proceedings of Canadian Society for Civil Engineering 1995 Hydrotechnical Conference*, June 1-3, 1995, Ottawa, ON.

Reckhow, K.H., Beaulac, M.N., and Simpson, J.T. (1980). *Modelling phosphorus loading and lake response under uncertainty: A manual and compilation of export coefficients.* United States Environmental Protection Agency, Clean Lakes Section, Washington, DC.

Redfield, A.C. (1958). The biological control of chemical factors in the environment. *American Scientist* **46**:205-221.

Redfield, G.W. (1991). Phosphorus, chlorophyll and the comparative limnology of three suburban lakes in Northern Virginia, U.S.A. *Verh. int. Verein. Limnol.* **24**:1294 - 1297.

Redfield, G.W. and Jones, R.C. (1982). Effects of urbanization on lake ecosystems. In: Jones, R.C., Redfield, G.W., and Kelso, D.P. (eds), Urbanization, Stormwater Runoff, and the Aquatic Environment. George Mason University, Fairfax, VA, USA. pp. 42-60.

Reid Crowther (1995). *Technology transfer seminar on stormwater systems, design, operation and maintenance.* Western Canadian Water and Wastewater Association [WCWWA], May 29, 1995, Red Deer, AB.

Reynolds, C.S. (1984). *The Ecology of Freshwater Phytoplankton.* Cambridge University Press, ISBN 0-521-28222-5, 384 p.

Reynolds, C.S. (1987). Cyanobacterial water blooms. *Adv. Bot. Res.* **3**:67-143.

Rhee, G.Y. (1978). Effects of N:P ratios and nitrate limitation on algal growth, cell composition, and nitrate uptake. *Limnol. Oceanogr.* **23**:10-25.

Rhee, G.Y. and Gotham, I.J. (1981). The effect of environmental factors on phytoplankton growth: Light and the interactions of light with nitrate limitation. *Limnol. Oceanogr.* **26**(4):649-659.

Ripl, W. (1976). Biochemical oxidation of polluted lake sediments with nitrate - a new lake restoration method. *Ambio* **5**:132-135.

Ripl, W. (1985). Oxidation of saprobelic sediments by nitrified effluents by treatment plants. In: *Lake and Watershed Management: Practical Applications.* Proceedings of the International Symposium of the North American Lake Management Society, October 16-19, 1984.

Ripl, W. (1992). Management of the water cycle: An approach to urban ecology. *Wat. Pollut. Res. J. Can.* **27**(2):221-237.

Ryding, S.-O. (1978). *Research on Recovery of Polluted Lakes. Loading, Water Quality and Responses to Nutrient Reduction.* Acta Universitatis Upsaliensis. ISBN 91-554-0752-8.

Sartor, J.D., G.B. Boyd, and F.J. Agardy (1974). Water pollution aspects of street surface contaminants. *J. Wat. Pollut. Control Fed.* **46**(3):458-467.

Schindler, D.W. (1977). Evolution of phosphorus limitation in lakes. *Science* **195**:260-262.

Schindler, D.W. and Fee, E.J. (1973). Diurnal variation of dissolved inorganic carbon and its use in estimating primary production and CO_2 invasion in Lake 227. *J. Fish. Res. Bd. Can.* **30**(10):1501-1510.

Schueler, T.R., Kumble, P.A., and Heraty, M.A. (1992). *A Current Assessment of Urban Best Management Practices - Techniques for Reducing Non-Point Source Pollution in the Coastal Zone.* Anacostia Restoration Team, Department of Environmental Programs, Metropolitan Washington Council of Governments, Washington, DC. Prepared for United States Environmental Protection Agency [USEPA], Office of Wetlands, Oceans, and Watersheds, 127 p.

Schumaker, R.J., Funk, W.H., and Moore, B.C. (1992). Zooplankton fluctuations in response to aluminium sulphate treatment of Newman Lake. *Proceedings of the North American Lake Management Society, 12th Annual International Symposium*, November 2-7, 1992, Cincinnati, OH.

Scott, K.A., Peterson, H., Christofi, N, Nieminen, G., and Sexsmith, W.C. (1996). Use of alum to control algal growth in a stormwater detention pond, Results of a full scale trial in the City of Regina. In: *Proceedings of the Western Canadian Water and Wastewater Association [WCWWA] Conference*, September 24-27, 1996, Regina, SK, pp. 71-80.

Seip, K.L. and Reynolds, C.S. (1995). Phytoplankton functional attributes along trophic gradient and season. *Limnol. Oceanogr.* **40**(3):589-597.

Shapiro, J. (1990). Current beliefs regarding dominance by blue-greens: The case for the importance of CO_2 and pH. *Verh. int. Verein. Limnol.* **24**:38-54.

Shapiro, J., Forsberg, B., Lamarra, V., Lindmark, G., Lynch, M., Smeltzer, E., and Zoto, G. (1982). *Experiments and Experiences in Biomanipulation Studies of Biological Ways to Reduce Algal Abundance and Eliminate Blue-greens.* United States Environmental Protection Agency, Report No. 600/3-82-096, 259 p.

Shapiro, J. and Wright. D.I. (1984). Lake restoration by biomanipulation. Round Lake, Minnesota, the first two years. *Freshwat. Biol.* **14**:371-383.

Smith, C.D. (1985). *Hydraulic Structures.* University of Saskatchewan Publication, Saskatoon, SK.

Smith, V.H. (1982). The nitrogen and phosphorus dependence of algal biomass in lakes: an empirical and theoretical analysis. *Limnol. Oceanogr.* **27**:1101-1112.

Smith, V.H. (1983). Low nitrogen to phosphorus ratios favour dominance by blue-green algae in lake phytoplankton. *Science* **221**:669-671.

Smith, V.H. (1986). Predicting the proportion of blue-green algae in lake phytoplankton. *Can. J. Fish. Aquat. Sci.* **43**:148-153.

Søndergaard, M., Jeppesen, E., Kristensen, P., and Sortkjær, O. (1990). Interactions between sediment and water in a shallow and hypereutrophic lake: a study on phytoplankton collapses in Lake Søbygård, Denmark. *Hydrobiologia*, **191**:139-148.

Sonzogni, W.C., Chapra, S.C., Armstrong, D.E., and Logan, T.J. (1982). Bioavailability of phosphorus inputs to lakes. *J. envir. Qual.* **11**:555-563.

Stainton, M.P., Capel, M.J., and Armstrong, F.A.J. (1977). The Chemical Analysis of Fresh Water. Canadian Fisheries and Marine Services Miscellaneous Special Publications 25 (2nd Edition). Freshwater Institute, Winnipeg, MB. 180 p.

Stewart, W.P.D., Preston, T., Peterson, H.G., and Christofi, N. (1982). Nitrogen cycling in eutrophic freshwaters. *Phil. Trans. R. Soc. Lond.* **B 296**:491-509.

Stockner, J.G. and Lund, J.W.G. (1970). Live algae in postglacial lake deposits. *Limnol. Oceanogr.* **15**:41-58.

Stratton, F.E. (1969). Nitrogen losses from alkaline impoundments. *J. Sanit. Engng. Div. Am. Soc. Civ. Engrs.* **95**(SA2):223-231.

Striegel, R.G. (1987). Suspended sediment and metals removal from urban runoff by a small lake. *Wat. Res. Bull.* **23**:985-996.

Talling, J.F. (1957). The phytoplankton population as a compound photosynthetic system. *New Phycol.* **56**:133-149.

Talling, J.F. (1976). The depletion of CO_2 from lake waters by phytoplankton. *J. Ecol.* **64**:79-121.

Tessier, A., Carnignan, R., and Belzile, N. (1994). Processes occurring at the sediment-water interface: Emphasis on trace elements. In: Buffle, J. and De Vitre, R.R. (eds), *Chemical and Biological Regulation of Aquatic Systems*. Lewis Publishers, ISBN 0-87371-448-2, pp. 137-173.

Tilman, D., Kiesling, R., Sterner, R., Kilham, S., and Johnson, F.A. (1986). Green, blue-green and diatom algae: Taxanomic differences in competitive ability for phosphorus, silica, and nitrogen. *Arch. Hydrobiol.* **106**:473-485.

Tilman, D., Kilham, S.S., and Kilham, P. (1982). Phytoplankton community ecology: The role of limiting nutrients. *Annu. Rev. Ecol. Syst.* **13**:349-372.

Tilzer, M.M. (1987). Light-dependence of photosynthesis and growth in cyanobacteria: implications for their dominance in eutrophic lakes. *N.Z.J. Mar. Freshwat. Res.* **21**:401-412.

Tourbier, J.T. and Westmacott, R. (nd). *Water Resources Protection Technology: A Handbook of Measures to Protect Water Resources in Land Development*, pp. 3.

Trimbee, A.M. and Prepas, E.E. (1987). Evaluation of total phosphorus as a predictor of the relative biomass of blue-green algae with emphasis on Alberta lakes. *Can. J. Fish. Aquat. Sci.* **44**:1337-1342.

Uehlinger, U. and Bloesch, J. (1987). Variation in the C:P ratio of suspended and settling seston and its significance for P uptake calculations. *Freshwat. Biol.* **17**:99-108.

Uhlmann, D. (1982). Strategies for controlling eutrophication of lakes and reservoirs. *Int. Revue ges. Hydrobiol.* **67**(6):821-835.

Ulén, B. (1978). Seston and sediment in Lake Norrviken II. Aerobic decomposition of algae. *Schweiz. Z. Hydrol.* **40**(1):104-118.

United Nations (1987). United Nations, World Commission on Environment and Development. *Our Common Future*. (The Bruntland Report). Oxford University Press, New York.

United States Environmental Protection Agency (1983). Athayde, D.N., Shelly, E.D., Driscoll, D., Gaboury, D., and Boyd, G. (1983). *Results of the Nationwide Urban Runoff Program - Volume 1 - Final Report*. Water Planning Division, U.S. Environmental Protection Agency, Washington, DC, 191 p.

United States Environmental Protection Agency. (1989). *Water Quality of the Nations Lakes*. Report to Congress, EPA 440/5-89-003.

United States Environmental Protection Agency. (1990*a*). *National Water Quality Inventory*. Report to Congress, 1988, EPA 440/4-90-003.

United States Environmental Protection Agency. (1990*b*). *The Lake and Reservoir Restoration Guidance Manual*. 2nd Edition, EPA 440/4-90-006.

Urbonas, B. (1986). Standardization of detention pond design for phosphorus removal. In: *Urban Runoff Pollution*. Torno, H.C., Marsalek, J., Desbordes, M. (eds), Springer-Verlag, Berlin, pp. 739-760.

Uttormark, P.D., and Hutchins, M.L. (1985). Cellular phosphorus concentration as a criterion for trophic status in rapidly flushed lakes. *Can. J. Fish. Aquat. Sci.* **42**:300-306.

van Eck, G.Th.M. and Smits, J.G.C. (1986). Calculation of nutrient fluxes across the sediment-water interface in shallow lakes. In: Sly, P.G. (ed), *Sediments and Water Interactions*. Springer Verlag, NY. ISBN 0-387-96293-X, pp. 289-301.

van Liere, L. and Mur, L.R. (1980). Occurrence of *Oscillatoria agardhii* and some related species, a survey. In: Barica, J. and Mur. L.R. (eds), *Hypertrophic Ecosystems*, W. Junk Publishers, The Hague, pp. 67-77.

van Liere, L., Zevenboom, W., and Mur, L.R. (1977). Nitrogen as a growth limiting factor for the growth of the blue-green algae *Oscillatoria agardhii*. *Prog. Wat. Technol.* **8**:301-312.

van Raaphorst, W. and Brinkman, A.G. (1984). The calculation of transport coefficients of phosphate and calcium across the sediment-water interface from experiments with undiscores. *Wat. Sci. Technol.* **17**:941-951.

van Straten, G. (1986). Identification, uncertainty assessment and prediction in lake eutrophication. PhD Thesis, Technical University of Twente, The Netherlands. Cited in: Boers, P.C.M. (1991). *The Release of Dissolved Phosphorus From Lake Sediments*. PhD Thesis, Limnological Institute and the Institute for Inland Water Management and Waste Treatment, Lelystad, The Netherlands.

Verduin, J. (1982). Components contributing to light extinction in natural waters: Method of isolation. *Arch. Hydrobiol.* **93**(3):303-312.

Vincent, W.F. (1989). Cyanobacterial growth and dominance in two eutrophic lakes: review and synthesis. *Archiv. f. Hydrobiologie Beih.* **32**:239-254.

Vollenweider, R.A. (1968). The Scientific Basis of Lake and Stream Eutrophication, with Particular Reference to Phosphorus and Nitrogen as Eutrophication Factors. OECD (Organization for Economic Cooperative Development) Paris, Technical Report DAS/DSI/68. 182 p.

Vollenweider, R.A. (1975). Input-output models with special reference to the phosphorus loading concept in limnology. *Schweitz. Z. Hydrol.* **37**:53-84.

Vollenweider, R.A. (1976). Advances in defining critical loading levels for phosphorus in lake eutrophication. *Mem. Inst. Ital. Idrobiol.* **53**:55-83.

Vollenweider, R.A. (1985). Elemental and biochemical composition of plankton biomass; some comments and explorations. *Arch. Hydrobiol.* **105**:11-29.

Wanielista, M.P. and Yousef, Y.A. (1992). *Stormwater Management.* John Wiley and Sons Inc., New York, ISBN 0-471-57135-0.

Walker, W.W. (1987). Phosphorus removal by urban runoff detention basins. *Lake and Reservoir Management* (**III**):314-326.

Wetzel, R.G. (1975). *Limnology.* W.B. Saunders Co., Philadelphia.

Whyte, A. (1985). Ecological Approaches to Urban Systems: Retrospect and Prospect. Nature and Resources Volume 21(1), Unesco, Paris, ISBN 0028-0844.

Williams, J.D.H., Jaquet, J.M., and Thomas. R.L.I. (1976). Forms of phosphorus in the surficial sediments of Lake Erie. *J. Fish. Res. Bd. Can.* **33**:413-429.

Williams, J.D.H., Shear, H., Thomas, R.L. (1980). Availability to *Scenedesmus qaudricauda* of different forms of phosphorus in sediment materials from the great lakes. *Limnol. Oceanogr.* **25**:1-11.

Willén, E. (1976). A simplified method of phytoplankton counting. *Br. Phycol. J.* **11**:265-278.

Yousef, Y.A. and Wanielista, M.P. (1990). *Efficiency Optimization of Wet Detention Ponds for Urban Stormwater Management.* Final Report submitted to Florida Department of Environmental Regulation, DER Contract WM159 (A1 and A2). Cited in: Wanielista, M.P. and Yousef, Y.A. (1992). *Stormwater Management.* John Wiley and Sons Inc., New York, ISBN 0-471-57135-0.

Zar, J.H. (1984). Biostatistical analysis. 2nd ed. Prentice-Hall, Englewood Cliffs, NJ. 718 p.

Zevenboom, W. and Mur, L.R. (1980). N_2-fixing cyanobacteria: why they do not become dominant in Dutch hypereutrophic lakes. In: Barica, J., and Mur, L.R. (eds), *Hypertrophic Ecosystems*. W. Junk Publishers, The Hague.