1990

Effects of Lake Management on Chemical, Physical, and Phycological Characteristics of a Hypereutrophic Reservoir

Kaye J. Surratt
Eastern Illinois University

THESIS REPRODUCTION CERTIFICATE

TO: Graduate Degree Candidates who have written formal theses.

SUBJECT: Permission to reproduce theses.

The University Library is receiving a number of requests from other institutions asking permission to reproduce dissertations for inclusion in their library holdings. Although no copyright laws are involved, we feel that professional courtesy demands that permission be obtained from the author before we allow theses to be copied.

Please sign one of the following statements:

Booth Library of Eastern Illinois University has my permission to lend my thesis to a reputable college or university for the purpose of copying it for inclusion in that institution's library or research holdings.

[redacted] [redacted]

Date Author

I respectfully request Booth Library of Eastern Illinois University not allow my thesis be reproduced because _____

_____

_____

_____ _____

Date Author

m

Effects of Lake Management on Chemical, Physical, and

Phycological Characteristics of a Hypereutrophic Reservoir
(TITLE)

BY

Kaye J. Surratt

THESIS

SUBMITTED IN PARTIAL FULFILLMENT OF THE REQUIREMENTS
FOR THE DEGREE OF

Master of Science

IN THE GRADUATE SCHOOL, EASTERN ILLINOIS UNIVERSITY
CHARLESTON, ILLINOIS

1990
YEAR

I HEREBY RECOMMEND THIS THESIS BE ACCEPTED AS FULFILLING
THIS PART OF THE GRADUATE DEGREE CITED ABOVE

[redacted]
DATE

[redacted]
ADVISER

[redacted]
DATE

[redacted]
DEPARTMENT HEAD

TABLE OF CONTENTS

ABSTRACT

Surratt, Kaye J. M.S., Eastern Illinois University. June 1990. Effects of Lake Management on Chemical, Physical, and Phycological Characteristics of a Hypereutrophic Reservoir.

Lake Charleston is a 1.1 billion gallon capacity side channel reservoir which was constructed in 1982 as a public drinking water supply for the City of Charleston in Coles County, Illinois. Past studies of the lake indicated an increase in eutrophication and the capacity to support large algal populations. Recent lake management practices include copper sulfate applications as an algicide and aeration to reduce flavor and odor.

In order to assess present lake status and to evaluate the effectiveness of lake management practices, two lake sites were sampled on a weekly basis from May to October, 1989. Chemical, physical, and phycological analyses followed standard methods (APHA 1985).

Carlson's Trophic State Indices calculated from data on secchi depth (m), chlorophyll a (ug/L), and total phosphorus (mg/L) indicated that Lake Charleston is hypereutrophic. Phosphorus probably limited algal standing crop at site 3 while light was most likely the limiting factor at site 1. Abundance of phosphorus at site 1 was attributed to the circulation of hypolimnetic phosphorus by the aeration unit.

Phytoplankton density has increased by a factor greater than five since 1982. During a bloom in July, bluegreen algae dominated

at site 3, however bluegreens never comprised more than fifty percent of total algae at site 1. The inability of bluegreens to dominate at site 1 could have resulted from low pH/high carbon dioxide water circulated from the bottom which would favor dominance by green algae. After copper sulfate treatment in late July, phytoplankton density decreased dramatically and green algae resumed dominance at site 3.

Several chemical, physical, and phycological parameters were significantly correlated. Site specific differences were observed and attributed to the effects of aeration.

Aeration may be a good management option for Lake Charleston with higher carbon dioxide and lower pH levels at site 1 presumably preventing bluegreen algal dominance. Total algal biomass should decrease with continued and/or extended aeration as nutrient concentrations level off and algae become light limited. Copper sulfate, which is toxic to zooplankton, could then be eliminated as a management technique and increased zooplankton populations could possibly provide biological control for phytoplankton. Dissolved oxygen has not yet increased, possibly because of an increase in oxygen demanding substances and a decrease in photosynthesis, both of which are attributable to circulation. With constant aeration, dissolved oxygen should eventually increase and overall lake conditions should improve.

ACKNOWLEDGEMENTS

I am deeply grateful to my advisors, Dr. Charles Pederson for his extensive assistance and suggestions in developing this study, and Dr. William Weiler for his guidance and encouragement throughout my educational experience at Eastern.

Thanks are extended to Dr. Kandy Baumgardner and Mr. Hank Nilsen as members of my graduate committee for their skill, time, and energy exerted in preparation of this thesis.

I would also like to acknowledge Mark Christ and the personnel of the Charleston Water Treatment Plant for sample collections and some of the analyses.

LIST OF FIGURES

LIST OF TABLES

INTRODUCTION

Most of the time, the loss of attractiveness and usefulness of a body of water is due to the process of eutrophication. Symptoms of eutrophication include bluegreen algal blooms, loss of volume, noxious odor, tainted fish flesh, degradation of domestic water supplies, dissolved oxygen depletion, fish kills, and nuisance animal populations (such as carp). Eutrophication is caused by excessive additions of plant nutrients, organic matter, and silt (Cooke et al. 1986). Management and protection of clean water systems and restoration of eutrophic water bodies are greatly needed due to increased recreational, industrial, and domestic use (Wetzel 1983). Lake management techniques include: i) physical and chemical control of nutrients using methods such as aeration, bank stabilization or presettling basins for contributing waters, ii) decreasing human contributions (runoff of lawn fertilizers and discharge of sewer wastes), and iii) controlling plant biomass by physical removal of aquatic macrophytes or use of algicides (Cooke et al. 1986).

Assessment of trophic state is essential for planning lake protection and/or restoration (Ravera 1983). Various indices have been developed by placing a numerical range on different trophic states based on specific parameters. These indices allow comparisons of a lake's present and past trophic conditions, and can help predict possible future states. Phosphorus is probably the most important nutrient associated with eutrophication while chlorophyll *a* is considered one of the more useful parameters for measuring

manifestations of nutrient enrichment (Taylor et al. 1979). The major benefit of chlorophyll analysis is increased speed and ease of algal biomass determinations in comparison to cell counts and species identification.

Carlson's Trophic State Index (TSI), calculated from secchi depth, total phosphorus, and/or chlorophyll a, is useful for lakes with few macrophytes and little nonalgal turbidity (Cooke et al. 1986). This index assumes that turbidity is due mostly to algal cells and that total phosphorus is the limiting nutrient. Given that these assumptions are true, phosphorus concentrations should directly affect algal biomass (chlorophyll a), and algal biomass will correlate negatively with transparency (secchi depth). Comparisons of TSI's calculated from secchi depth, chlorophyll a and total phosphorus may be used to identify possible nutrient or light limitation and presence of non-algal turbidity (Carlson 1977).

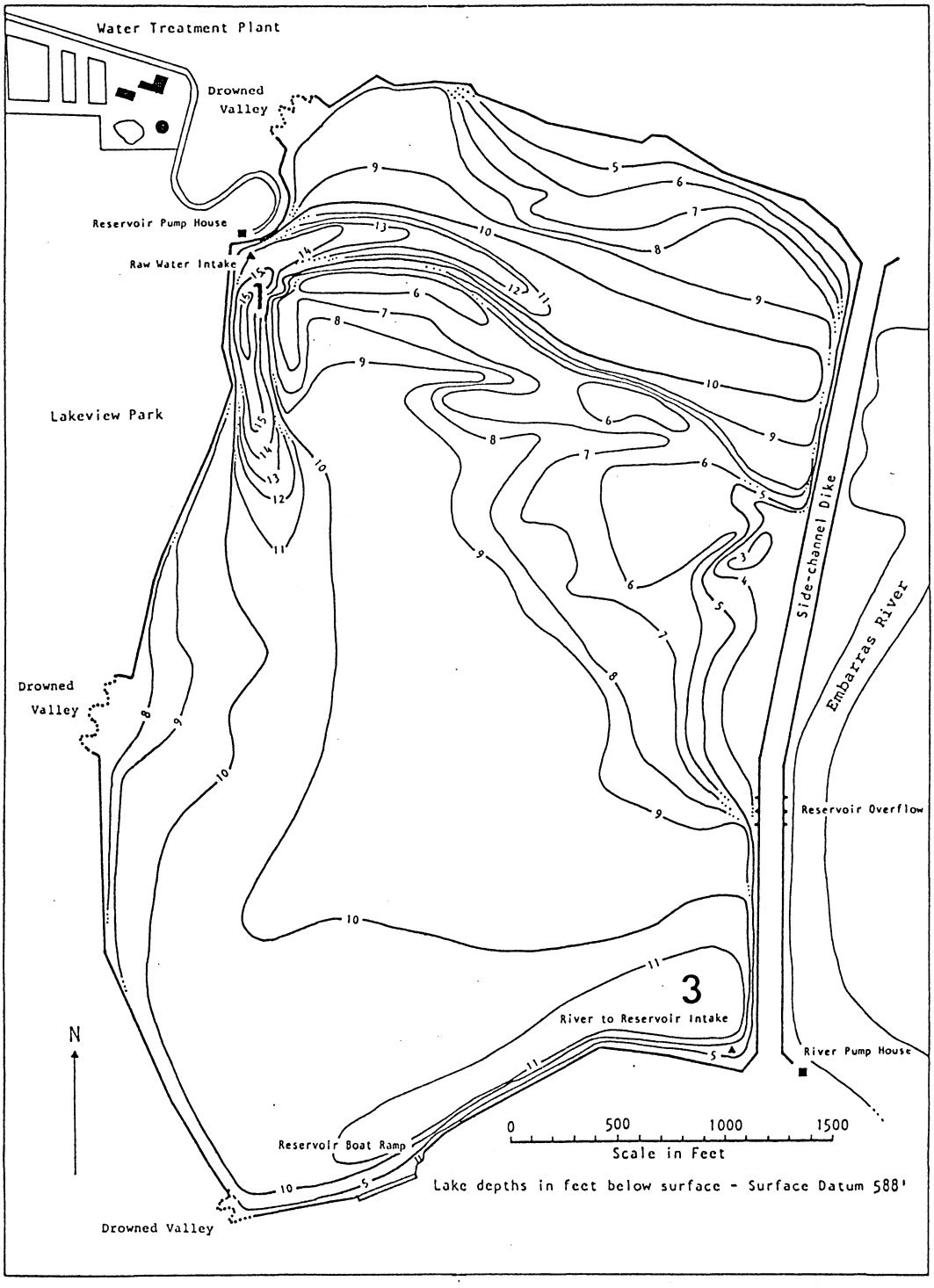
Phytoplankton structure and biomass may be the most useful indicator of trophic state since algae depend upon the quality and quantity of nutrients available (Ravera 1983). Rott (1984) found that each trophic state has its own general seasonal changes in phytoplankton community structure which can be used for determining specific trends of development for a body of water. Several algal divisions have been shown to be collectively associated with eutrophication and used as indicators. Nygaard's Trophic State Indices (Nygaard 1949) classify a body of water as oligotrophic or eutrophic based on taxonomic ratios. In more recent studies, Mantere and Heinonen (1982) and Somashekar (1984) found that populations of

desmids (Zygnematales, Chlorophyceae) and Chlorococcales (Chlorophyceae) increased with eutrophication. Species distributions may also reflect water quality changes because some species are typically found in eutrophic lakes while other species apparently prefer oligotrophic conditions. Palmer (1969) assigned pollution index numbers to certain genera and species known to be sensitive to or tolerant of organic pollution. Summation of the index numbers for the species present in a body of water provides a guideline for assessing the extent of organic pollution.

Lake Charleston (Figure 1), located in Coles County of east-central Illinois, is a 1.1 billion gallon capacity reservoir with 346 acres of surface area which was constructed in 1982. Historical data on the formation of the lake was reviewed by Lookis (1983). In 1880, the Embarras River became the public drinking water supply for the City of Charleston. Construction of a dam in the early 1930's and enlargement of this impoundment in 1947 by the Riverview Dam ensured a constant water supply for the growing population. Flow of the Embarras River through Lake Charleston resulted in sedimentation problems and by 1974 the lake was losing 2.3 percent of its storage capacity annually (Yang 1974). In an attempt to alleviate the sedimentation problem, Lake Charleston was converted to a side channel pump storage reservoir in 1982 by constructing a dike to separate it from the flow of the Embarras River.

Previous studies of Lake Charleston (Morris et al. 1978, Lookis 1983, Hawes 1988) indicated increases in algal densities and eutrophication. Present management of the lake as a public water

Figure 1. Contour map of Lake Charleston Side Channel Reservoir showing the locations of sites 1 and 3. This map was produced from data collected in 1988 by Dr. Vincent Gutowski of the Geology/ Geography Department, and Mr. Mark Christ and Mr. Robert Young of the Environmental Biology program at Eastern Illinois University.


Water Treatment Plant
Drowned Valley
Reservoir Pump House
Raw Water Intake
Lakeview Park
Drowned Valley
Side-channel Dike
Embarras River
Reservoir Overflow
3
River to Reservoir Intake
River Pump House
Reservoir Boat Ramp
Drowned Valley
N
0 500 1000 1500
Scale in Feet
Lake depths in feet below surface - Surface Datum 588'

supply involves treatments which include aeration near the raw water intake to reduce flavor and odor, and up to three additions per year of copper sulfate (at a rate of 2000 lb. per treatment) as a phycological control. Citric acid is added as a chelating agent at the rate of 1000 lb. per application of copper sulfate. The purposes of this study were to: i) assess water quality changes in Lake Charleston by comparing present algal abundance, structure and diversity data to past studies, ii) identify correlations between algal community and various chemical and physical parameters and, iii) assess effects of management practices on phytoplankton and trophic state.

METHODS

Lake sites (Figure 1) were sampled initially on 5 May 1989 and on a weekly basis with few exceptions through 24 October 1989. Grab samples were collected from the upper meter of water in one liter acid rinsed plastic screw cap bottles. Secchi depth, temperature, dissolved oxygen, and pH were measured on site. A 100 mL portion of each sample was preserved for phycological examination by the addition of 3 ml of Lugol's solution, with the remainder being used for chemical analysis.

All of the chemical tests except for phosphorus were performed within six hours after samples were collected. Portions of the samples were frozen for 1-3 days for determinations of total phosphorus. Standard methods (APHA 1985) were used for all physical, chemical, and biological analyses (Table 1). Quality control included testing of known samples from the USEPA for each parameter except phytoplankton, and all results were within the 95% confidence intervals reported. Statistical analyses were performed using the Statpak version 4.1 (Northwest Analytical). Phytoplankton samples were concentrated by filtration, and enumeration followed the methods of McNabb (1960). Identification was accomplished using a phase contrast microscope with a total magnification of 1000X. Algae were identified at least to genus, and to species when possible using the keys of Smith (1950), Taft and Taft (1971), and Prescott (1978). Species richness was defined as the number of algal taxa present in a

Table 1. APHA methods (1985) utilized for chemical, physical, and biological analyses by the author (KS) or the Charleston Water Treatment Plant (CWTP).

Parameter	Reported as	Standard method	Section number	Analyzer
Chlorophyll *a*	ug/L	Spectrophotometric	1002G I	CWTP
Secchi depth	m			CWTP
pH		pH	423	CWTP
Total alkalinity	mg/L $CaCO_3$	Alkalinity	403	CWTP
Hardness	mg/L $CaCo_3$	EDTA titrimetric	314B	CWTP
Turbidity	NTU	Nephelometric	214A	CWTP
Temperature	°C	Temperature	212	CWTP
Dissolved oxygen	mg/L	Membrane electrode	421F	CWTP
Total solids	mg/L	Dried at 103-105 °C	209A	KS
Suspended solids	mg/L	Dried at 103-105 °C	209C	KS
Total phosphorus	mg/L	Persulfate digestion	424C III	KS
		Ascorbic acid colorimetry	424F	KS

sample. Phytoplankton diversity was calculated using the Shannon-Weaver Index (Shannon and Weaver 1963):

$$H' = -\text{sum } p_i \log_{10} p_i$$

where p_i = number of individuals of species i divided by the total number of individuals.

Evenness (J') was calculated as a ratio of observed diversity to maximum diversity for a given number of taxa (Pielou 1969), using the formula:

$$J' = H'/H'max$$

where $H'max = \log s$

s = number of taxa present in a given sample.

Carlson's Trophic State Indices were calculated from data on total phosphorus, chlorophyll a, and secchi depth (Carlson 1977).

RESULTS AND DISCUSSION

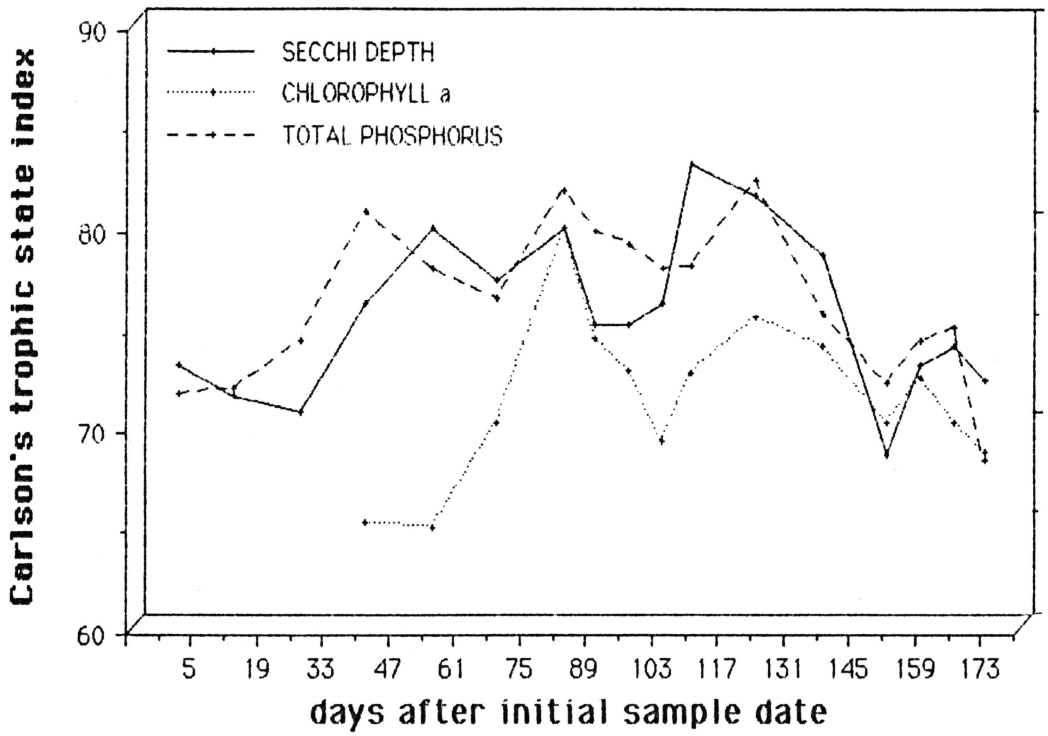
Lake Trophic State

Carlson's (1977) Trophic State Indices (TSI) were calculated from data on secchi depth, chlorophyll *a*, and total phosphorus (Appendix A, Appendix B). TSI values (Figure 2) ranged from 65-85 indicating that Lake Charleston is hypereutrophic. At both sites, trophic states calculated from data on total phosphorus and secchi depth were comparable. TSI values at site 1 based on chlorophyll *a* were lower than those calculated from the other two parameters from early June until the time of the bloom in mid-July, and again shortly after the bloom until mid-September. At site 3, trophic state as determined by chlorophyll *a* was lower than trophic states based on total phosphorus and secchi depth only from early June until the onset of the algal bloom.

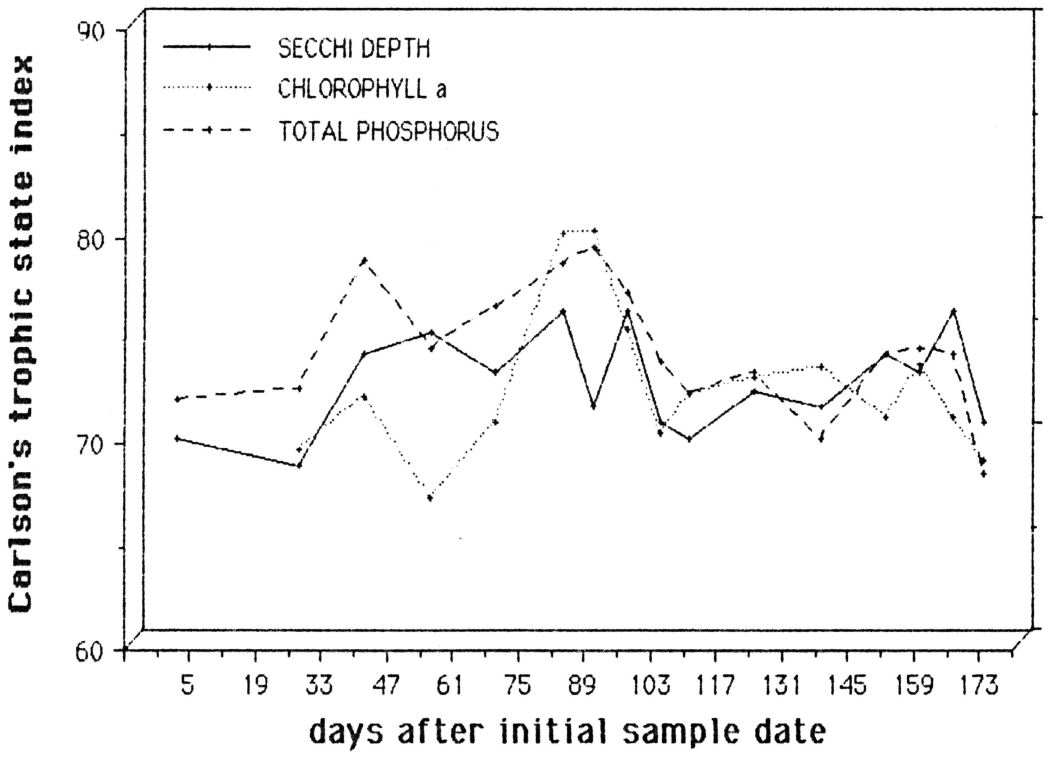
Carlson's TSI assumes that phytoplankton growth is limited by available phosphorus, and that secchi depth is determined primarily by algal density. Given that these assumptions are true, similar TSI values should be calculated whether based on chlorophyll *a*, total phosphorus, or secchi depth (Carlson 1977). In Lake Charleston, (Figure 2) algal standing crop at site 1 was probably not limited by available phosphorus from June to late July, but may have been limited by phosphorus during the bloom in late July and from late September to the end of October. At site 3, phosphorus probably limited algal standing crop at all times except during June and July. Through August and early September, the reduced standing crop at both

Figure 2. A) Carlson's Trophic State Index as determined from secchi depth (m), chlorophyll a (ug/L), and total phosphorus (mg/L) at site 1. B) Carlson's Trophic State Index as determined from secchi depth (m), chlorophyll a (ug/L), and total phosphorus (mg/L) at site 3.

A


SECCHI DEPTH
CHLOROPHYLL a
TOTAL PHOSPHORUS
Carlson's trophic state index
90
80
70
60
5
19
33
47
61
75
89
103
117
131
145
159
173
days after initial sample date

B


SECCHI DEPTH
CHLOROPHYLL a
TOTAL PHOSPHORUS
Carlson's trophic state index
90
80
70
60
5
19
33
47
61
75
89
103
117
131
145
159
173
days after initial sample date

sites was probably caused by the addition of copper sulfate. Abundance of phosphorus at site 1 is most likely attributable to increased internal phosphorus loading resulting from circulation of water from the hypolimnion.

Phytoplankton density, species diversity, and community dominance

Lake Charleston was classified as eutrophic in 1973 and supported 702-8,980 algal units mL^{-1} from May through October (Morris et al. 1978). Following conversion to a side channel pump storage reservoir in 1982, algal density ranged from 10-20,000 units mL^{-1} during the summer months, but no bluegreen algal blooms were observed (Lookis 1983). By 1987, Lake Charleston was classified as hypereutrophic by the U.S. EPA Volunteer Lake Monitoring Program with nutrients and suspended solids the major causes of impairment (Hawes et al. 1988). Phytoplankton density during 1989 at site 3 normally ranged from 55,022 to 140,994 units mL^{-1} with numbers reaching as high as 312,939 on July 26 (Figure 3). At site 1, algal density ranged from 61,126 to 174,528 units mL^{-1} with the highest numbers being recorded in late August (Figure 4).

In 1972, twenty-four algal taxa were identified from Lake Charleston and algal diversity ranged from 0.608 to 0.716 (Morris et al. 1978). Fifty-seven taxa were identified from the lake during 1989 (Appendix C) and increased diversity, ranging from 0.695 to 1.077, was observed. Algal diversity at site 1 was typically lower than diversity at site 3 except during the bloom (Figure 5). Fluctuations in diversity generally followed patterns of species

Figure 3. Phytoplankton density and community structure fluctuations with time at site 1.

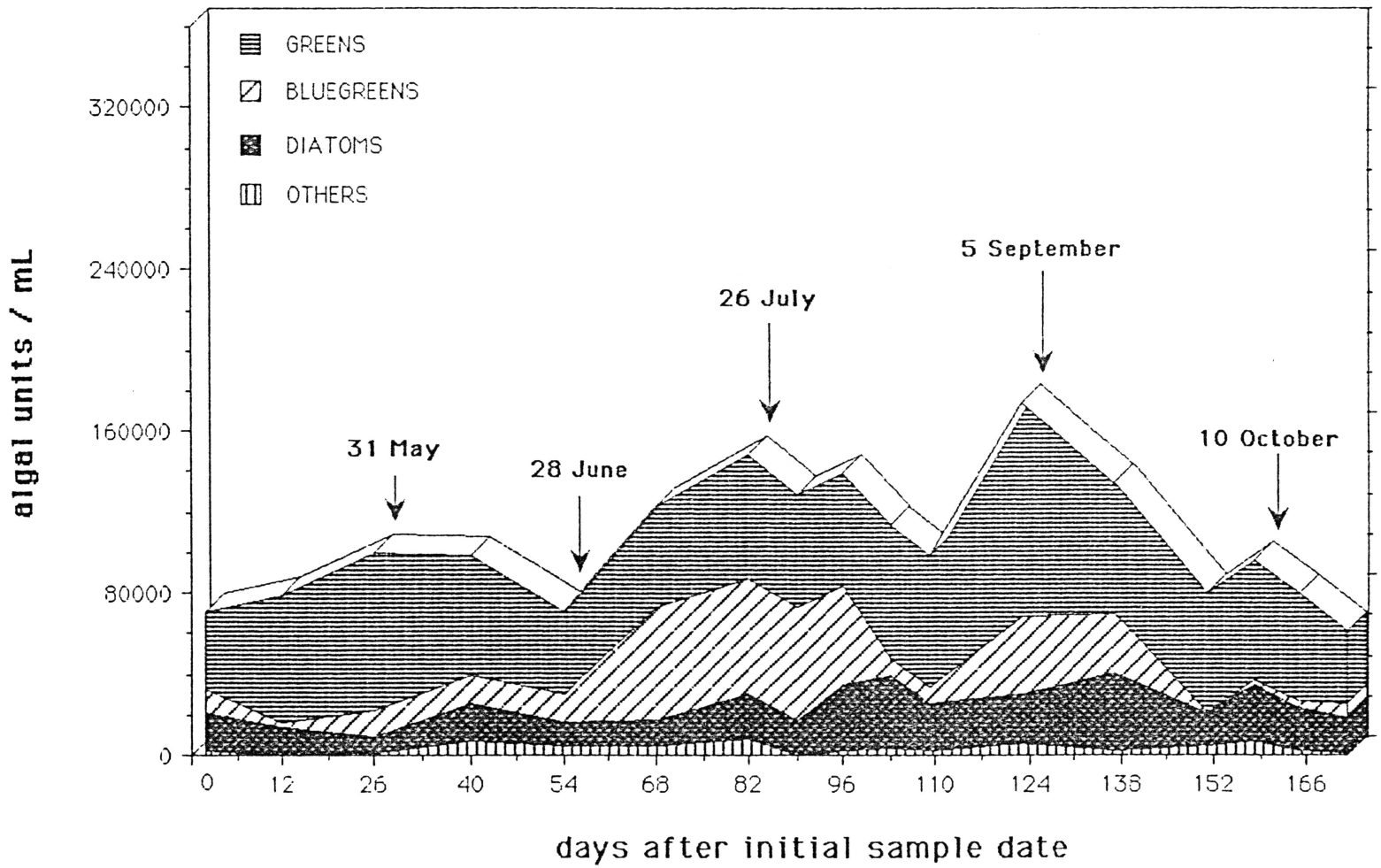

GREENS
BLUEGREENS
DIATOMS
OTHERS
320000
240000
160000
80000
0
algal units / mL
31 May
28 June
26 July
5 September
10 October
0
12
26
40
54
68
82
96
110
124
138
152
166
days after initial sample date

Figure 4. Phytoplankton density and community structure fluctuations with time at site 3.

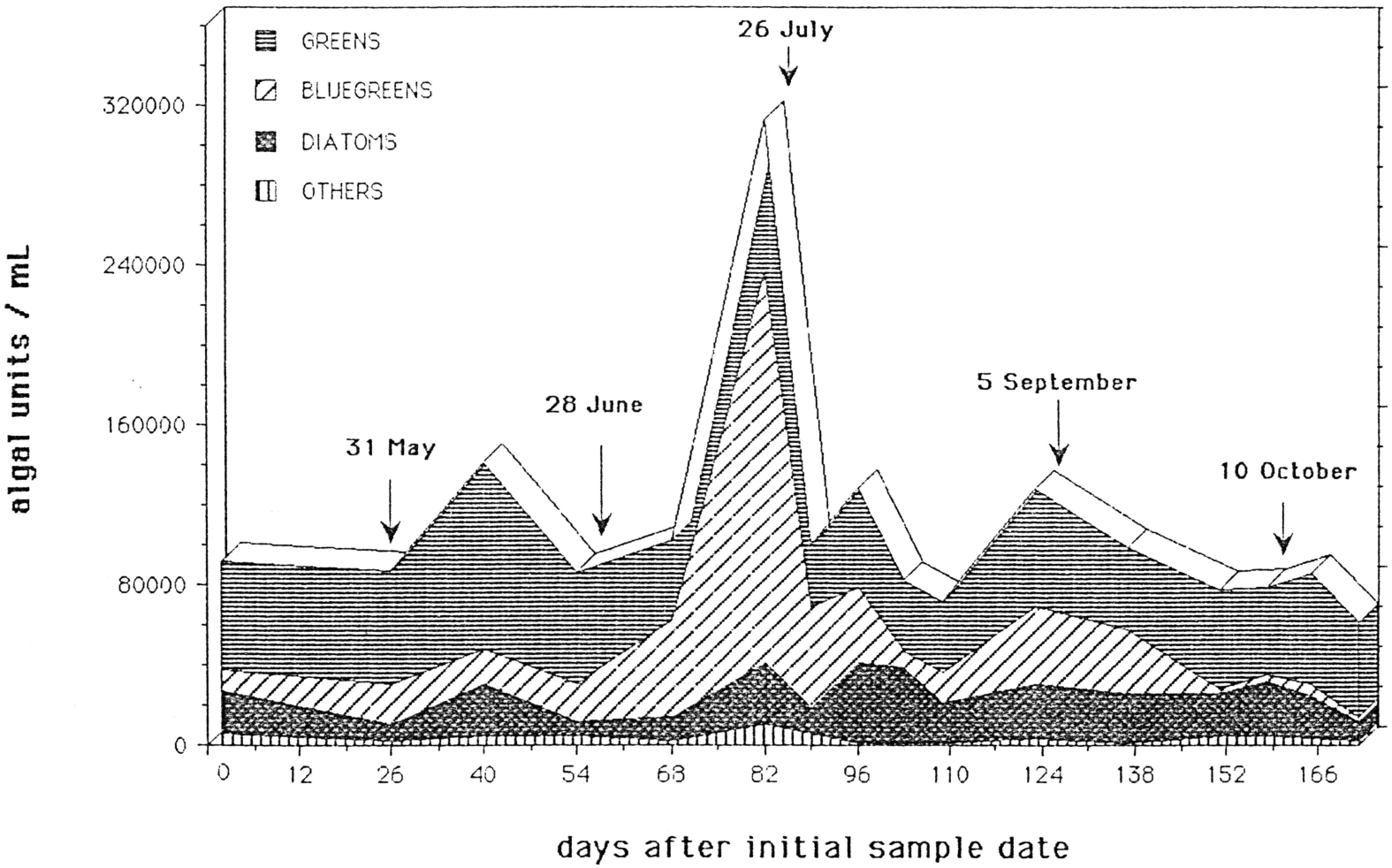
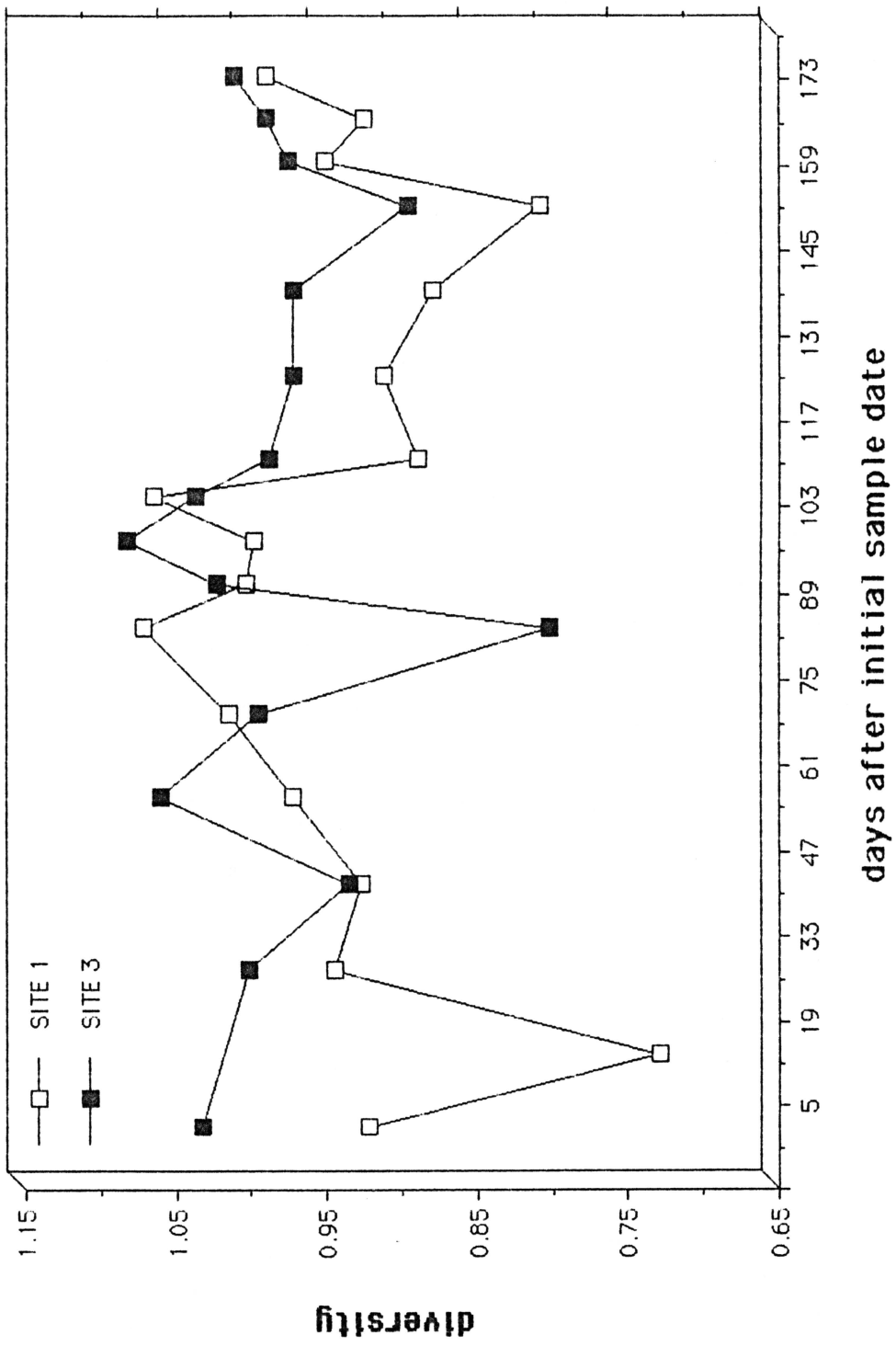

GREENS
BLUEGREENS
DIATOMS
OTHERS
26 July
28 June
31 May
5 September
10 October
320000
240000
160000
80000
0
algal units / mL
0
12
26
40
54
68
82
96
110
124
138
152
166
days after initial sample date

Figure 5. Algal diversity differences between sites 1 and 3.


SITE 1
SITE 3
1.15
1.05
0.95
0.85
0.75
0.65
diversity
5
19
33
47
61
75
89
103
117
131
145
159
173
days after initial sample date

richness (Figure 6A). Bluegreen algae became dominant during the bloom at site 3 (Figure 7A), thereby producing a decline in both species richness and evenness (Figures 6A, 6B). Higher species richness and evenness values which were observed at site 1 (Figures 6A, 6B) resulted from the codominance of bluegreen and green algae (Figure 7B).

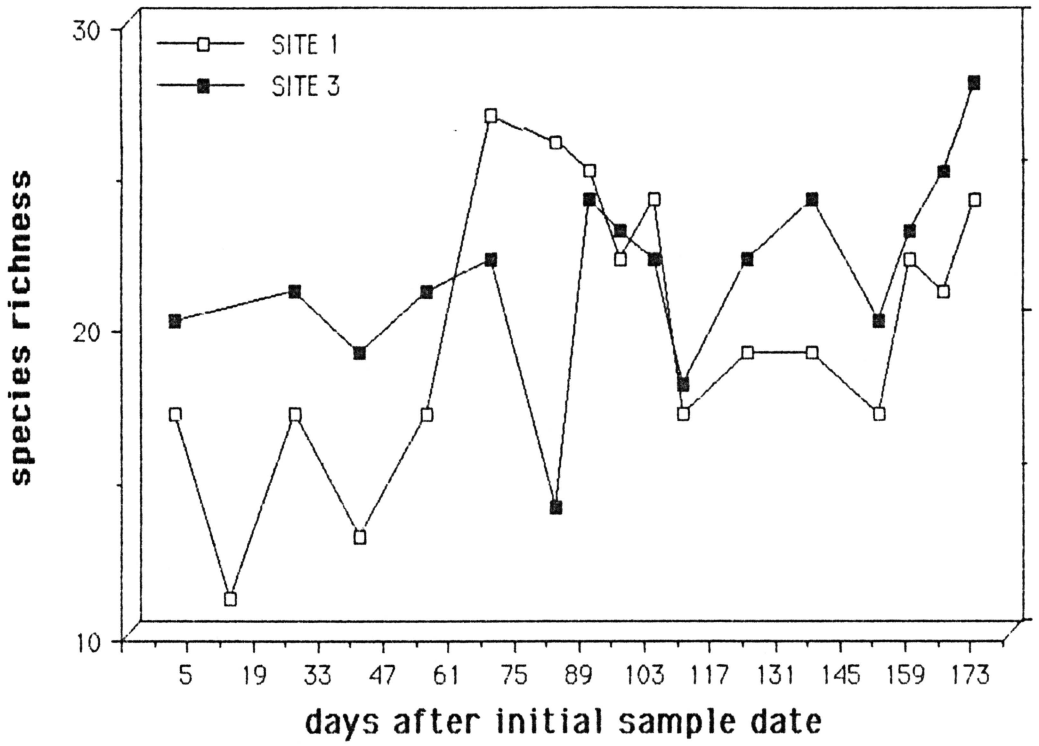
Aeration at site 1 is presumed to be responsible for differences in the patterns of algal diversity and community dominance observed at sites 1 and 3. The inability of the bluegreen algae to become dominant at site 1 may be attributable to increased circulation at that site brought about by the aerator. Increased circulation can result in elevated carbon dioxide levels and decreased pH in the euphotic zone by verticle transport of bottom water which has a high carbon dioxide content due to bacterial respiration and lack of photosynthetic activity. Bluegreen algal dominated cultures shift to ones dominated by green algae in response to decreased pH and associated increases in free carbon dioxide concentrations (Shapiro 1984). This phenomenon is due to the possible lysing of the bluegreen algae by viruses favored at low pH and the lack of the advantage bluegreen algae have over green algae in their capacity to absorb carbon dioxide when present at low concentrations (Shapiro 1982).

Chemical, Physical, and Phycological Correlations

Correlation coefficients between selected physical, chemical, and phycological data are presented in Table 2. Correlation

Figure 6. A) Species richness fluctuations with time at sites 1 and 3. B) Evenness fluctuations with time at sites 1 and 3.

A


SITE 1
SITE 3
species richness
30
20
10
5
19
33
47
61
75
89
103
117
131
145
159
173
days after initial sample date

B

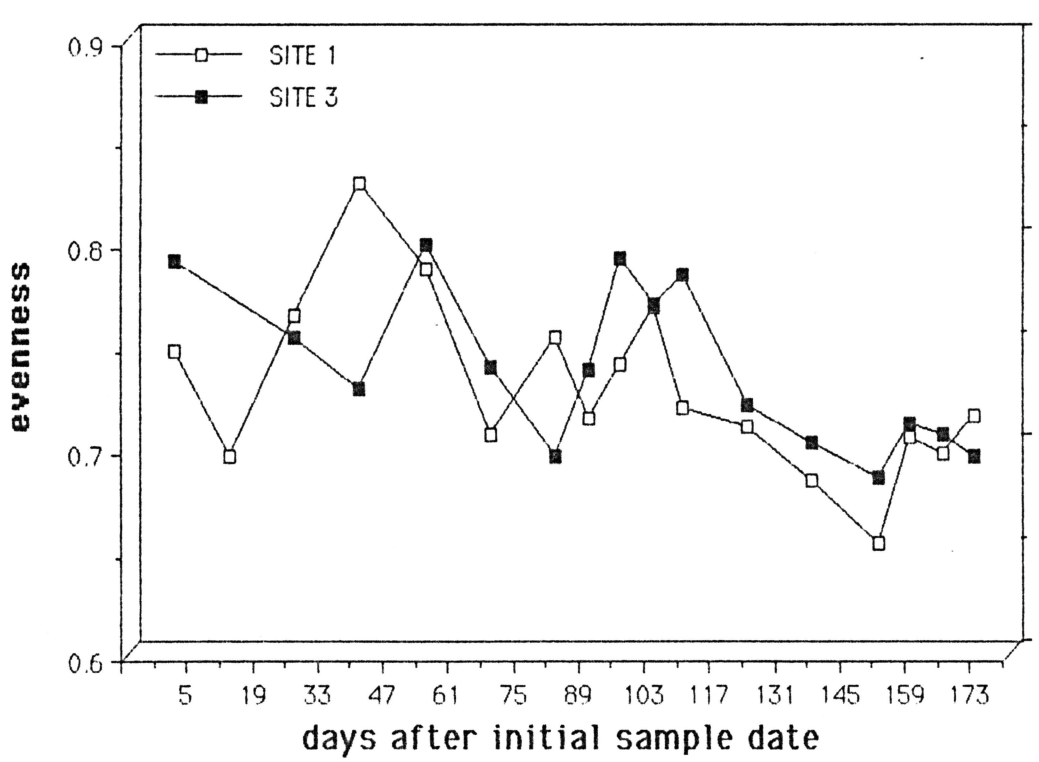
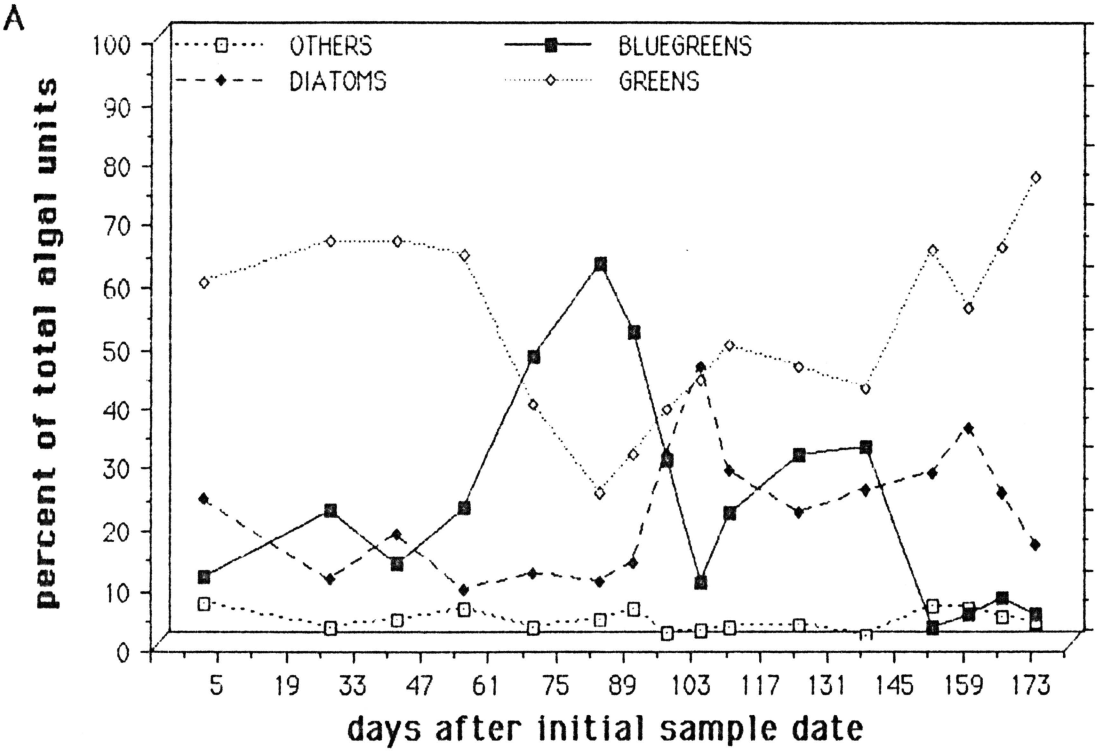

SITE 1
SITE 3
evenness
0.9
0.8
0.7
0.6
5
19
33
47
61
75
89
103
117
131
145
159
173
days after initial sample date

Figure 7. A) Percentage composition of algal groups at site 3.

B) Percentage composition of algal groups at site 1.

A


OTHERS
BLUEGREENS
DIATOMS
GREENS
percent of total algal units
0
10
20
30
40
50
60
70
80
90
100
5
19
33
47
61
75
89
103
117
131
145
159
173
days after initial sample date

B

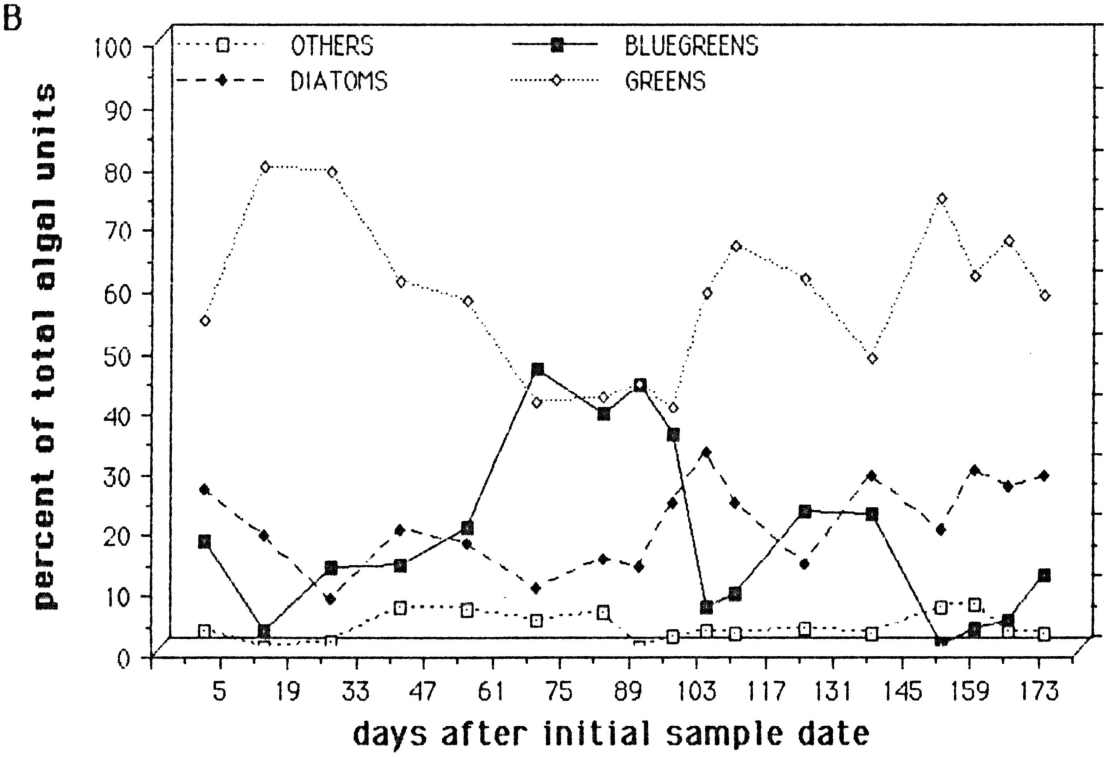

OTHERS
BLUEGREENS
DIATOMS
GREENS
percent of total algal units
0
10
20
30
40
50
60
70
80
90
100
5
19
33
47
61
75
89
103
117
131
145
159
173
days after initial sample date

Table 2. Significant correlations ($p < 0.05$) between selected physical, chemical, and biological parameters at site 1 (r_{s1}), and at site 3 with (r_{s3}) and without ($r_{s3}*$) an outlier.

			r_{s1}	r_{s3}	$r_{s3}*$
SECCHI DEPTH	vs	pH	0.70	-----	-----
		TURBIDITY	-0.73	-0.65	-0.81
		TSS	-0.56	-0.61	-0.63
		TPO4	-0.72	-0.47	-0.54
		TOTALG	-0.54	-0.46	-----
		BLGRNS	-0.43	-----	-----
pH	vs	TURBIDITY	-0.76	-----	-----
		TSS	-0.62	-----	-----
		TPO4	-0.78	-----	-----
		TOTALG	-0.71	0.52	-----
		TOTGRNS	-0.51	-----	-----
		BLGRNS	-0.41	0.56	-----
TOTALK	vs	CHL-*a*	-0.56	-0.66	-0.47
		BLGRNS	-0.70	-0.77	-0.74
		GRNS	-----	-----	0.52
HARDNESS	vs	TOTPO4	-0.60	-----	-----
		CHL-*a*	-0.66	-0.60	-0.55
		TOTALG	-0.68	-----	-----
		GRNS	-----	-----	0.49
		BLGRNS	-0.57	-0.44	-0.49
		DIATOMS	-0.48	-----	-----
TURBIDITY	vs	TSS	0.88	0.47	0.50
		TPO4	0.67	-----	-----
		TOTALG	0.61	-----	-----
		GRNS	0.69	-----	-----
		DIATOMS	0.47	-----	-----
TS	vs	TPO4	0.44	-----	0.47
		TOTALG	-----	-----	0.71
		GRNS	-----	0.60	0.75
		BLGRNS	0.42	-----	-----
TSS	vs	TPO4	0.69	0.70	0.74
		TOTALG	0.51	-----	0.62
		GRNS	0.72	0.53	0.56

Table 2. (cont.)

TPO4	vs	CHL-a	0.48	0.66	0.58
		TOTALG	0.78	0.57	0.61
		GRNS	0.51	-----	-----
		BLGRNS	0.61	0.53	0.51
		OTHERS	0.46	0.56	0.44
CHL-a	vs	TOTALG	0.71	0.66	-----
		BLGRNS	0.65	0.74	0.59
TOTALG	vs	GRNS	0.67	0.58	0.56
		BLGRNS	0.77	0.95	0.60
		DIATOMS	0.43	-----	-----

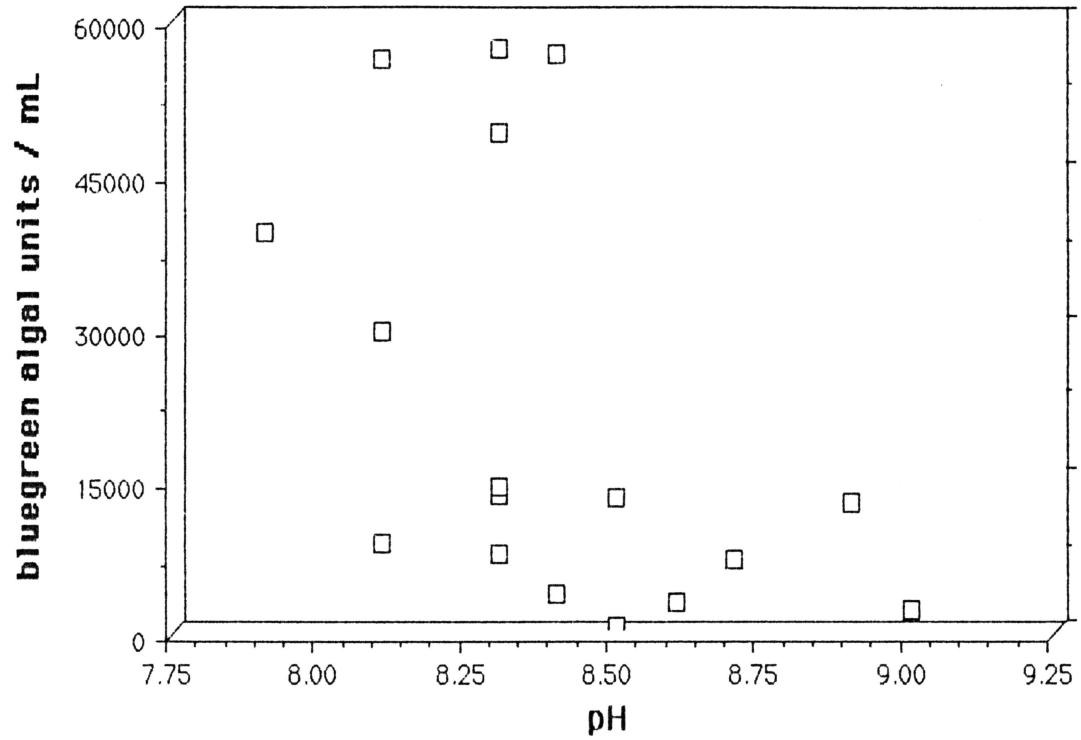
TOTALK = Total alkalinity
TS = Total solids
TSS = Total suspended solids
TPO4 = Total phosphorus
CHL-a = Chlorophyll a
TOTALG = Total algae
GRNS = Total green
BLGRNS = Total blue-green algae
DIATOMS = Total diatoms
OTHERS = Total others

coefficients of pH versus total algae and total bluegreens were negative at site 1 (r_{*1} = -0.71, r_{*1} = -0.41) but positive at site 3 (r_{*3} = 0.52, r_{*3} = 0.56). Scattergrams of pH versus total bluegreens and total greens at site 1 (Figure 8) and site 3 (Figure 9) suggest that this magnitude of difference is due to one extreme data point at site 3. When this outlier is omitted (Figure 10), the correlations are not significant for total algae (r_{*3}* = -0.06) or bluegreens (r_{*3}* = 0.06). The marked increase in pH observed at site 3 during the bluegreen bloom is to be expected if carbon dioxide is being taken out of solution by the algae.

The strong negative relationship between pH and algae at site 1 may exist because of the positive correlations of total phosphorus with total algae (r_{*1} = 0.78), green algae (r_{*1} = 0.51), bluegreen algae (r_{*1} = 0.61), and other algae (r_{*1} = 0.46). In addition to being negatively correlated with algal parameters, pH was also negatively correlated with total phosphorus (r_{*1} = -0.78) at site 1. Circulation of bottom waters theoretically could decrease pH and increase phosphorus loading from sediments as stated in previous sections. Although normally these trends would be expected to level off with constant circulation, the aeration system was not operating on a constant basis due to mechanical problems. Intermittent operation of the aeration system resulted in periodically elevated total phosphorus concentrations which increased algal growth potential at the same time pH levels were declining. Thus, there was probably no cause and effect relationship between pH and the algal community.

Figure 8. A) Scatter diagram of pH versus bluegreen algal units per milliliter at site 1 ($r_{s1} = -0.41$, $p < 0.05$). B) Scatter diagram of pH versus total algal units per milliliter at site 1 ($r_{s1} = -0.71$, $p < 0.05$).

A


bluegreen algal units / mL
60000
45000
30000
15000
0
7.75
8.00
8.25
8.50
8.75
9.00
9.25
pH

B

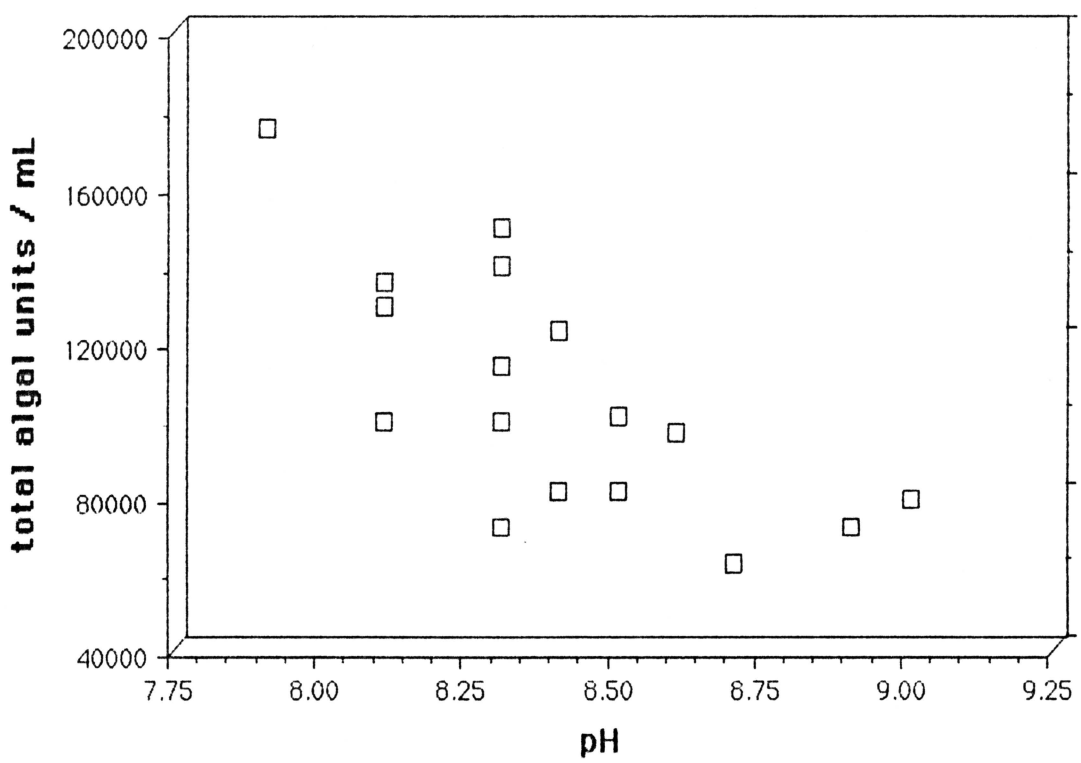
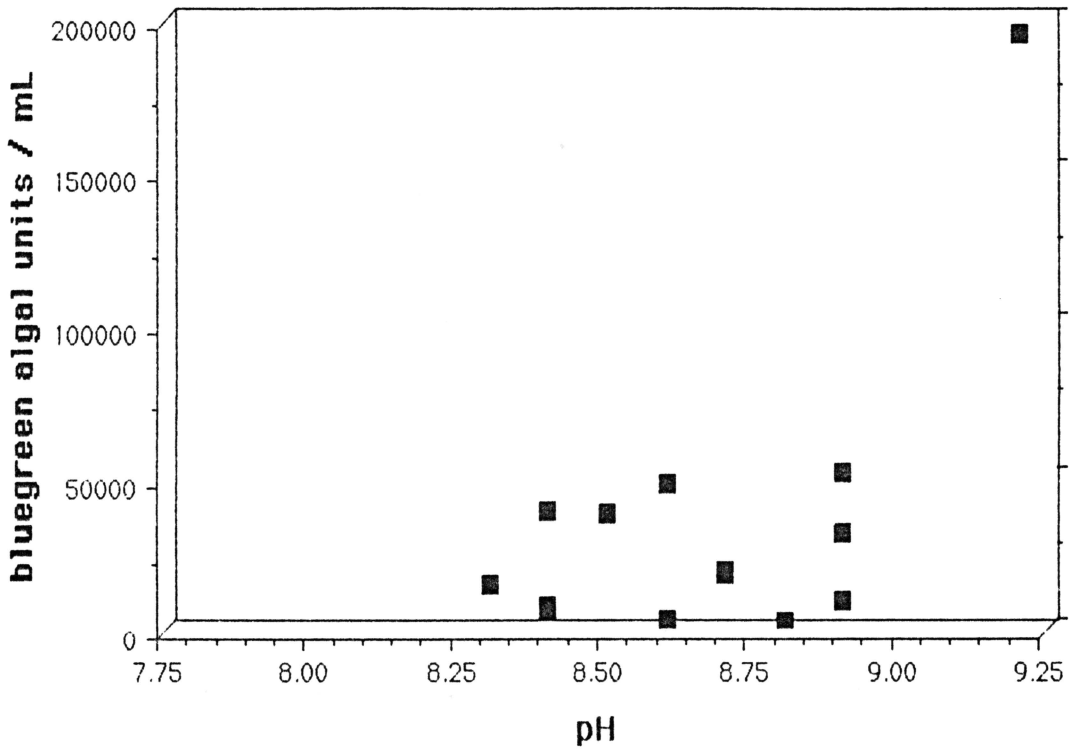

total algal units / mL
200000
160000
120000
80000
40000
7.75
8.00
8.25
8.50
8.75
9.00
9.25
pH

Figure 9. A) Scatter diagram of pH versus bluegreen algal units per milliliter at site 3 (r_{*3} = 0.56, $p < 0.05$). B) Scatter diagram of pH versus total algal units per milliliter at site 3 (r_{*3} = 0.52, $p < 0.05$).

A


bluegreen algal units / mL
200000
150000
100000
50000
0
7.75
8.00
8.25
8.50
8.75
9.00
9.25
pH

B

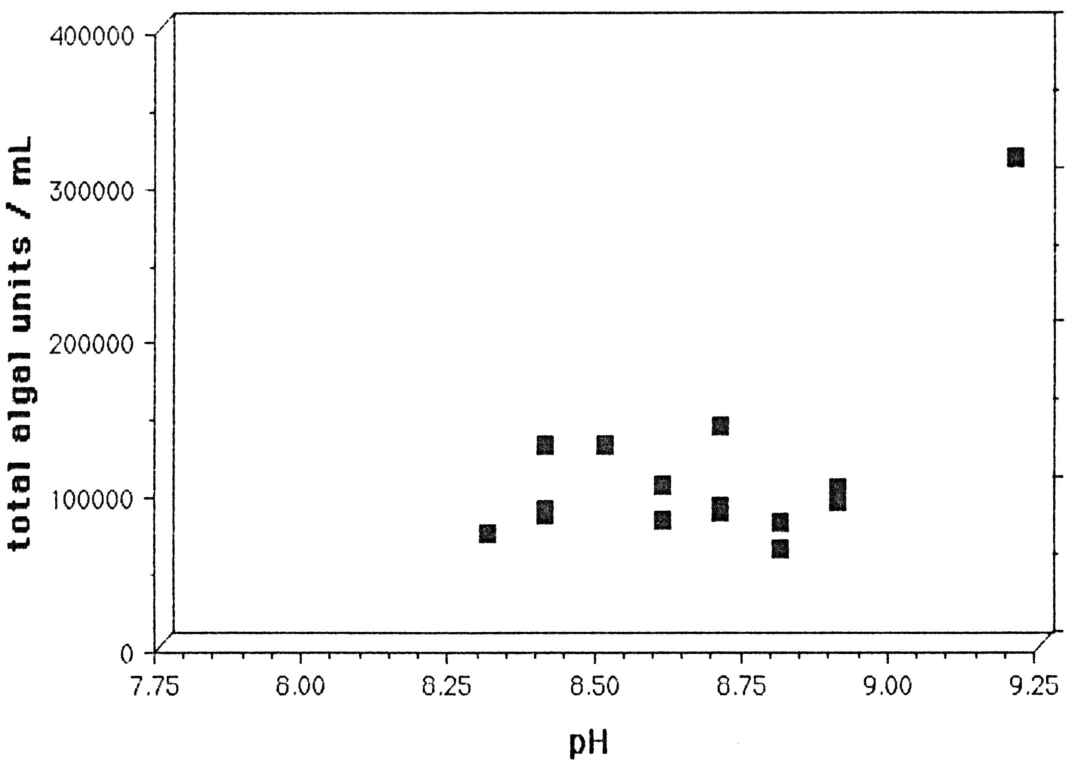
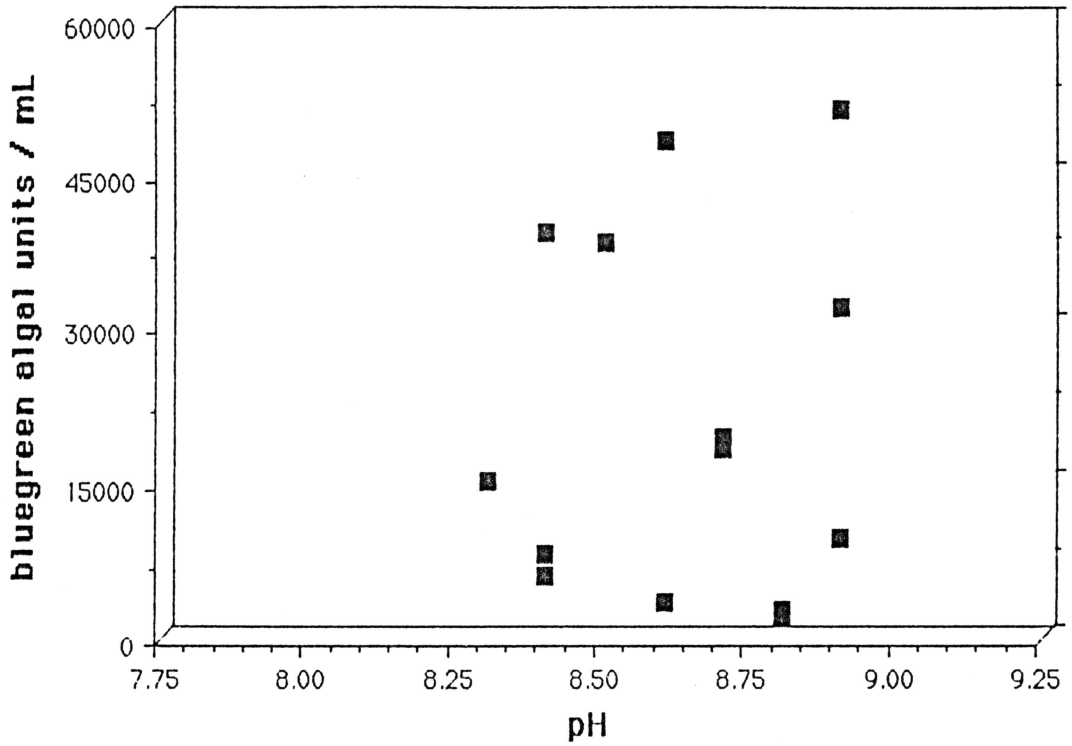
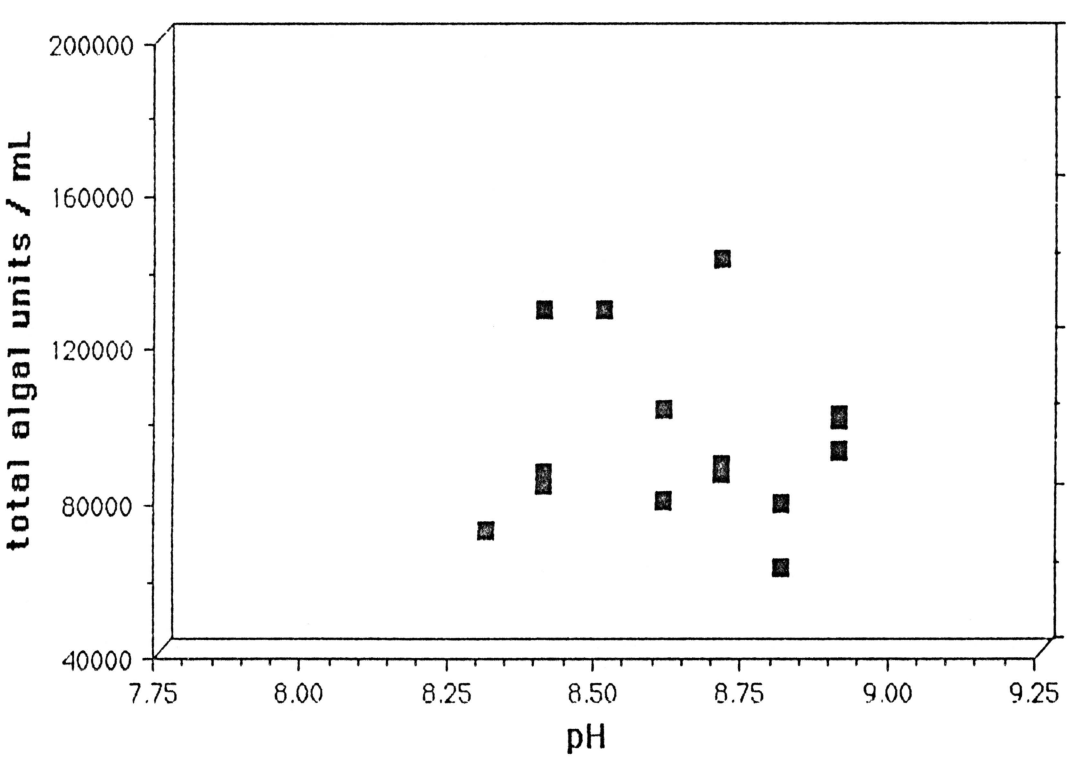

total algal units / mL
400000
300000
200000
100000
0
7.75
8.00
8.25
8.50
8.75
9.00
9.25
pH

Figure 10. A) Scatter diagram of pH versus bluegreen algal units per milliliter at site 3 without the outlier. B) Scatter diagram of pH versus total algal units per milliliter at site 3 without the outlier.

A


bluegreen algal units / mL
60000
45000
30000
15000
0
7.75
8.00
8.25
8.50
8.75
9.00
9.25
pH

B


total algal units / mL
200000
160000
120000
80000
40000
7.75
8.00
8.25
8.50
8.75
9.00
9.25
pH

Secchi depth was correlated negatively with turbidity (r_{*1} = -0.73, r_{*3}* = -0.81), total suspended solids (r_{*1} = -0.56, r_{*3}* = -0.63), and total algae (r_{*1} = -0.54). A similar correlation would be expected between secchi depth and total algae at site 3, but this was not significant (r_{*3}* = -0.37). Total suspended solids including algal cells and non-organic particulate matter directly affect the turbidity of water, and as turbidity increases the depth of light penetration (secchi depth) decreases. The fact that total phosphorus was highly correlated with total algae (r_{*1} = 0.78, r_{*3}* = 0.61) and chlorophyll *a* (r_{*1} = 0.48, r_{*3}* = 0.58) may explain why total phosphorus and secchi depth were inversely related (r_{*1} = -0.72, r_{*3}* = -0.47).

When comparing the percentage composition of major algal groups, a high correlation of bluegreen algae (r_{*1} = 0.77, r_{*3}* = 0.60) and green algae (r_{*1} = 0.67, r_{*3}* = 0.56) with total algae is anticipated. Chlorophyll *a* was highly correlated with bluegreen algae (r_{*1} = 0.65, r_{*3}* = 0.59) and with total algae at site 1 (r_{*1} = 0.71). Chlorophyll *a* is the major photosynthetic pigment of algae and is estimated to account for 0.5-2.0 percent of a cell's dry weight (Reynolds 1984). Because of this relationship, chlorophyll *a* is often used to estimate algal density instead of direct cell counts and therefore should be highly correlated with algal parameters.

In Lake Charleston, the negative correlations of total alkalinity with chlorophyll *a* (r_{*1} = -0.56, r_{*3}* = -0.47) and bluegreen algae (r_{*1} = -0.70, r_{*3}* = -0.74) may be attributable to increased phytoplankton densities accompanied by increased

photosynthetic activity. Total alkalinity refers to bicarbonates, carbonates, and hydroxides present in water which, when in equilibrium as shown below, constitute the major buffering mechanism in fresh waters (Wetzel 1983).

$$H_2CO_3 \longleftrightarrow HCO_3^- \longleftrightarrow CO_3^{-2}$$

Photosynthetic consumption of inorganic carbon (as carbon dioxide or as bicarbonate ions) could shift the above equilibrium to the left thereby decreasing alkalinity. Similarly negative correlations of blue-green algae and chlorophyll a with hardness probably existed because hardness is a measure of calcium and magnesium salts which are largely combined with carbonates and bicarbonates. At site 3 hardness was significantly correlated with total alkalinity (r_{s3}* = 0.54) while at site 1 the correlation was fairly high (r_{s1} = 0.49) but not significant.

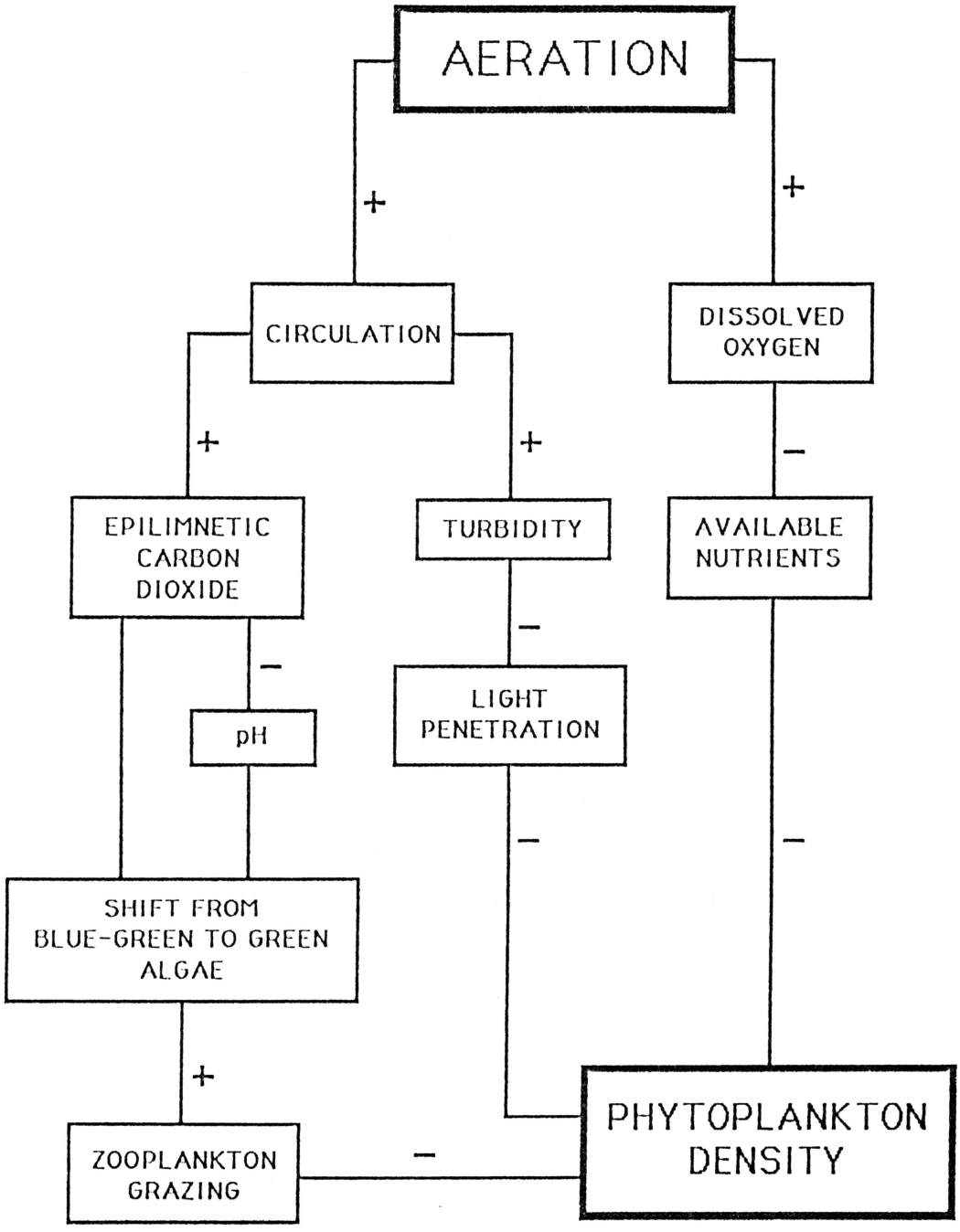
CONCLUSIONS

Lake Charleston is a hypereutrophic reservoir which continues to increase its capacity to support large populations of phytoplankton with 55,022-312,939 algal units mL^{-1} . From Carlson's Trophic State Index, it appears that phosphorus is the limiting nutrient at unaerated site 3. Algal biomass is probably limited mostly by light at site 1 where nutrients may be supplied by the circulation of water from the hypolimnion and algal cells may be dispersed throughout the water column.

Application of copper sulfate as an algicide was effective at reducing algal numbers, but treatment is costly and the benefits are temporary. Algal biomass increased again after application especially at site 1 but was probably controlled by cooler temperatures in September. It has been suggested that rebound of algal populations after copper sulfate treatment, as seen in this study, may be due to toxic effects of copper on zooplankton (Cooke et al. 1986). One way in which aeration can decrease phytoplankton densities depends on higher zooplankton grazing rates attainable when algal community dominance shifts from bluegreen algae to green algae thus, with simultaneous use of copper sulfate and aeration as lake management techniques, aeration may not decrease phytoplankton biomass as expected.

Between site comparisons can provide insight into the usefulness of aeration in this lake as a management scheme. Figure 11 shows the ideal effects of aeration. Although aeration did not decrease

Figure 11. Possible beneficial effects of aeration (modified from Shapiro, 1979).


AERATION
+
+
CIRCULATION
DISSOLVED OXYGEN
+
+
−
EPILIMNETIC CARBON DIOXIDE
TURBIDITY
AVAILABLE NUTRIENTS
−
−
pH
LIGHT PENETRATION
−
−
SHIFT FROM BLUE-GREEN TO GREEN ALGAE
+
PHYTOPLANKTON DENSITY
ZOOPLANKTON GRAZING
−

phytoplankton biomass at site 1, it may have served to increase carbon dioxide concentration in the euphotic zone which could allow green algae to be co-dominant with bluegreen algae. Algal diversity was lower at the aerated site except during the bloom.

While circulation has served to improve dissolved oxygen concentrations in most studies (Cooke et al. 1986), Site 1 had a lower dissolved oxygen concentration than unaerated site 3 (Figure 12). Possible reasons for this are increases in oxygen demanding substances and a decrease in photosynthesis in the epilimnion with both conditions conceivably resulting from circulation. Dissolved oxygen concentration should increase at site 1 with continued aeration (Fast et al. 1975, Garrell et al. 1977, LaBaugh 1980).

Except during the bloom in July and during early October, secchi depth was lower at site 1 as expected (Figure 13). The decrease in secchi depth could be a result of increased algal cells below the surface and/or increased sediments in the water column. There were significant correlations between turbidity and total suspended solids at both sites, but the algae were also significantly correlated with turbidity at site 1. Decreased secchi depth values signify a decrease in light penetration. As light intensities decrease and as mixing depth increases, net photosynthesis should approach zero and in some cases might even be negative due to light limitation. As long as phytoplankton are circulated out of the euphotic zone, light limitation should eventually cause a decrease in biomass.

Site 1 had pH values lower than site 3 as expected with one exception (Figure 14). This probably resulted from circulation of

Figure 12. Dissolved oxygen concentration (mg/L) differences between sites 1 and 3.

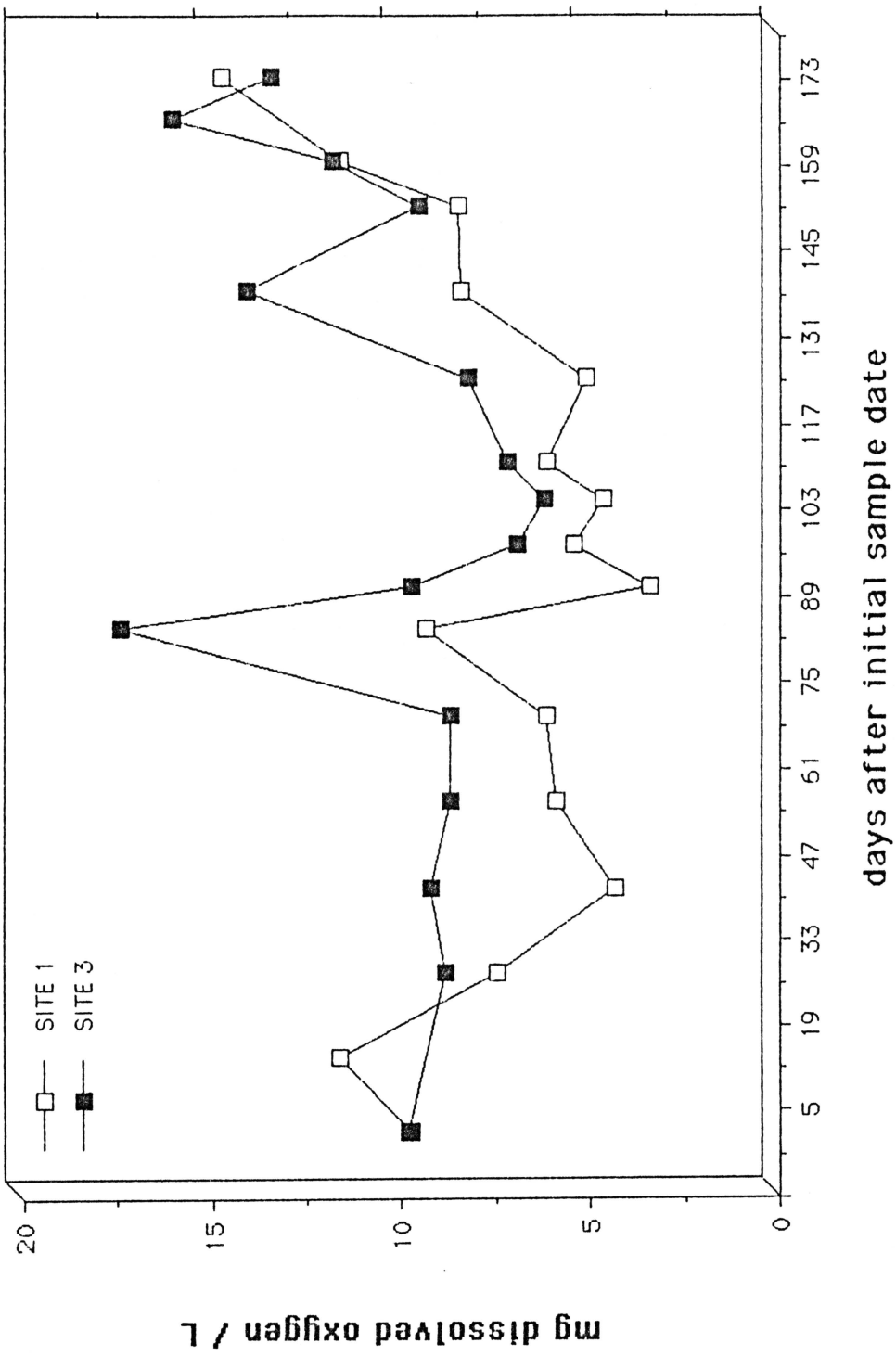

mg dissolved oxygen / L
20
15
10
5
0
SITE 1
SITE 3
5
19
33
47
61
75
89
103
117
131
145
159
173
days after initial sample date

Figure 13. Secchi depth (m) differences between sites 1 and 3.

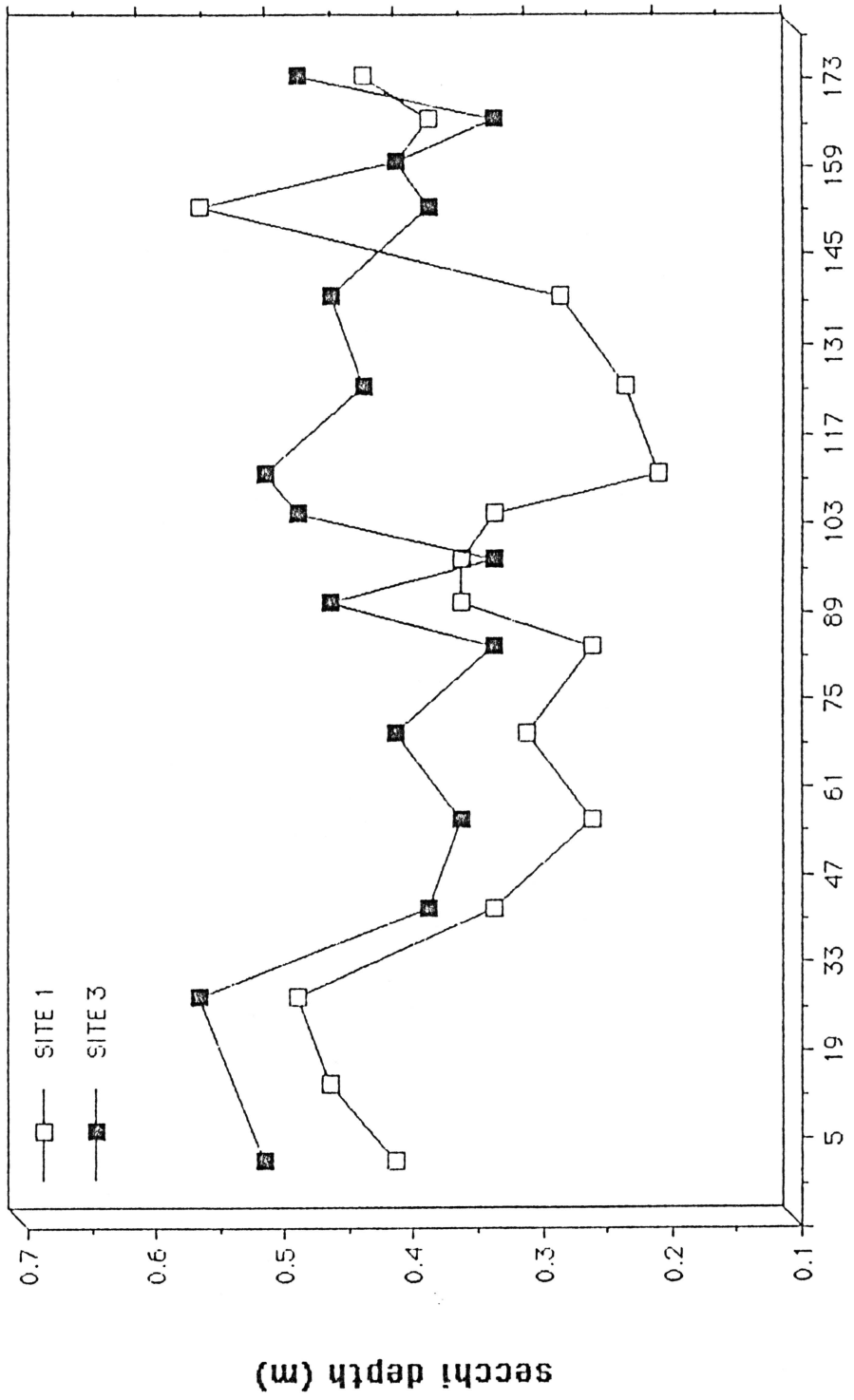
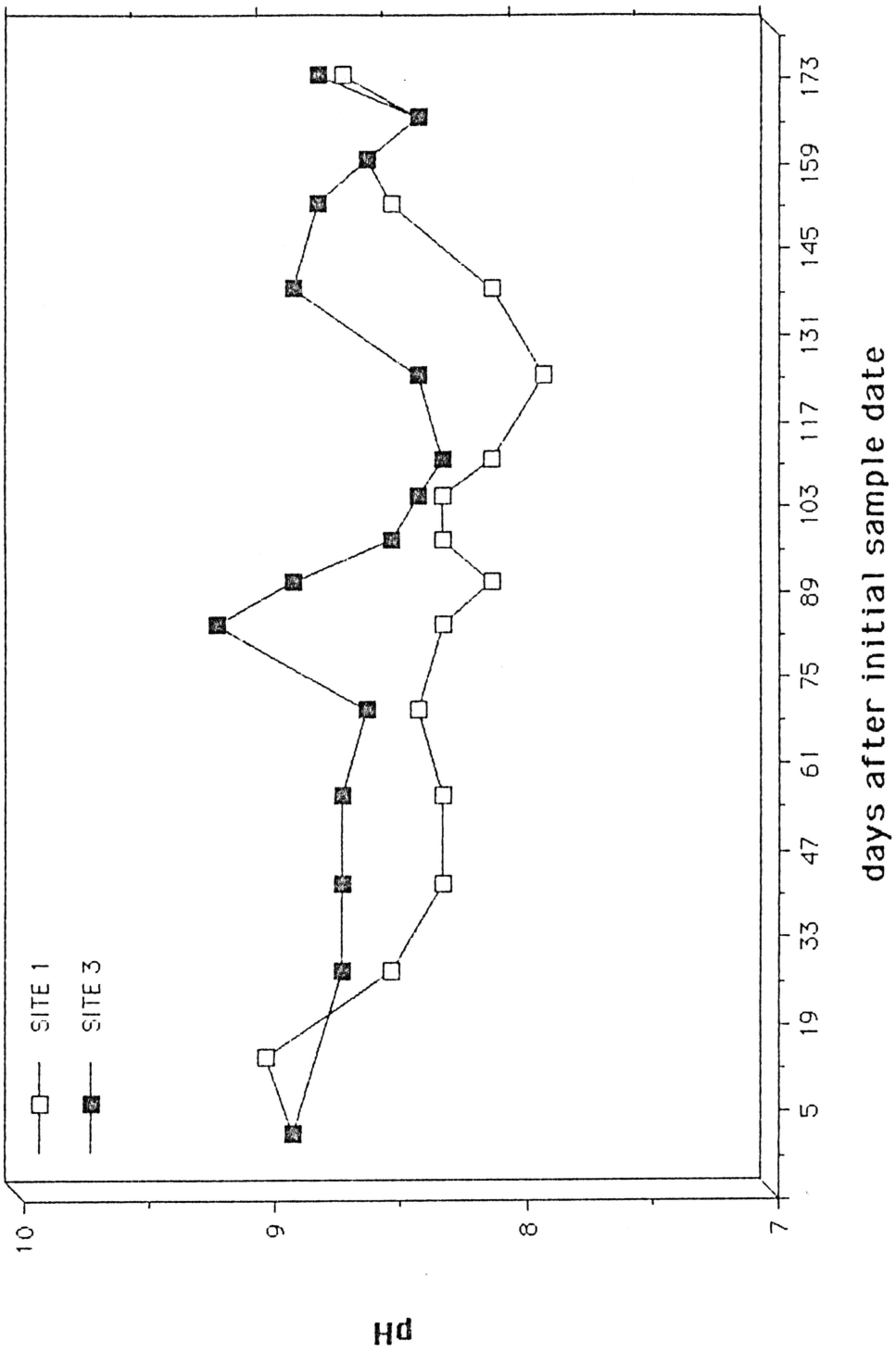

SITE 1
SITE 3
0.7
0.6
0.5
0.4
0.3
0.2
0.1
5
19
33
47
61
75
89
103
117
131
145
159
173
secchi depth (m)
days after initial sample date

Figure 14. Differences in pH between sites 1 and 3.


SITE 1
SITE 3
10
9
8
7
pH
5
19
33
47
61
75
89
103
117
131
145
159
173
days after initial sample date

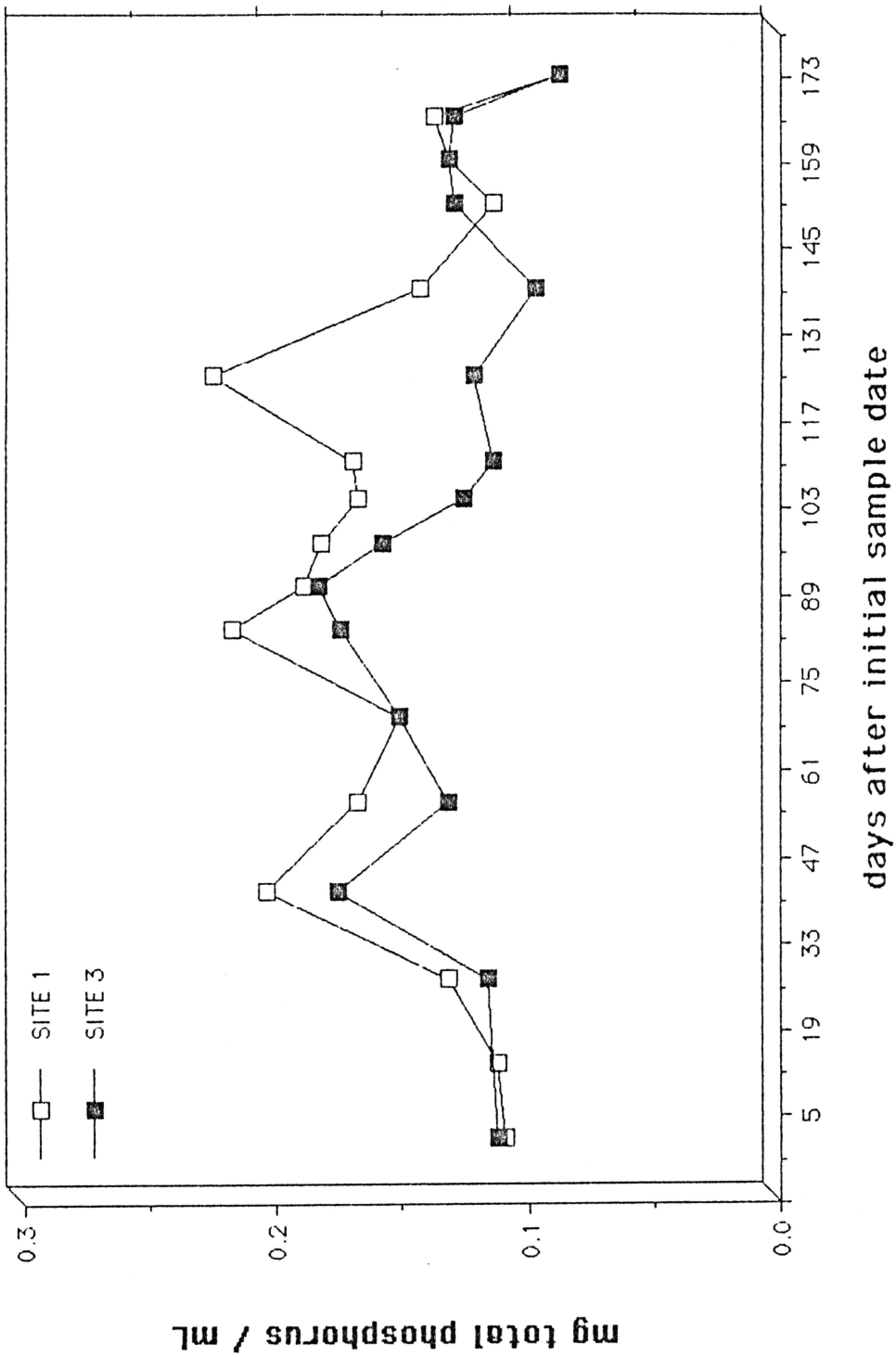
hypolimnetic water with low pH and high carbon dioxide content to the surface waters. These two factors may have combined to prevent bluegreen algae from becoming dominant at site 1. Since circulated water contained higher concentrations of suspended solids and total phosphorus, it is logical that pH at site 1 also correlated with turbidity, total suspended solids, secchi depth, total algae and bluegreen algae.

Total phosphorus is also apparently supplied to the photic zone via circulation (Figure 15). As long as nutrients are not limiting production, algal biomass should eventually decrease due to light limitation (Cooke et al. 1986). If nutrients are limiting, algal densities could potentially increase and result in lower transparencies and carbon dioxide levels as well as higher pH values. These in turn would support dominance of a community by 'nuisance' bluegreen algae.

With these results, it appears that aeration may be a good management option for Lake Charleston although further monitoring is essential as the system becomes fully operational. If total biomass is not decreased by light limitation in the future, the number of aeration units could be increased in an effort to prevent blue green algal blooms throughout the entire lake. This then could possibly eliminate the need for copper sulfate treatments and increase zooplankton populations for biological control of phytoplankton densities.

Eventually controls will have to be directed to decrease nutrient loading. Internal loading may be decreased through nutrient

Figure 15. Total phosphorus (mg/L) differences between sites 1 and 3.


SITE 1
SITE 3
0.3
0.2
0.1
0.0
5
19
33
47
61
75
89
103
117
131
145
159
173
days after initial sample date
mg total phosphorus / mL

oxidation by the aeration system. Other options include phosphorus inactivation by the addition of aluminum salts and sediment removal if cost and disposal are not problems. The most important source of external loading for Lake Charleston is probably erosion, but bankline stabilization measures are expensive.

LITERATURE CITED

American Public Health Association. 1985. Standard methods for the examination of water and wastewater, 16th edition. American Public Health Association, Washington, D.C.

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

Cooke, G.D., E.B. Welch, S.A. Peterson, and P.R. Newroth. 1986. Lake and Reservoir Restoration. Butterworth Publishers, Stoneham, MA.

Fast, A.W., V.A. Dorr, and R.J. Rosen. 1975. A submerged hypolimnion aerator. Water Resour. Res. 11:287-293.

Forsberg, B.R., and J. Shapiro. 1980. Predicting the algal response to destratification. Pages 134-139 in Restoration of Lakes and Inland Waters, USEPA, EPA 440/5-81-010.

Garrell, M.H., J.C. Confer, D. Kirchner, and A.W. Fast. 1977. Effects of hypolimnetic aeration on nitrogen and phosphorus in a eutrophic lake. Water Resour. Res. 13:343-347.

Hawes, J., N. Thomas, and V. Wood. 1988. Volunteer Lake Monitoring Program, 1987, Volume 5:East-Central Illinois Region, IEPA/WPC/88-016e.

LaBaugh, J.W. 1980. Water chemistry changes during artificial aeration of Spruce Knob Lake, West Virginia. Hydrobiologia 20:201-216.

Lookis, K.G. 1983. Raw water quality in the new Charleston, Illinois pump storage reservoir. M.S. thesis, Eastern Illinois University, Charleston, Illinois.

Mantere, R., and P. Heinonen. 1982. The quantity and composition of phytoplankton, particularly Chlorophyta, in lakes of different trophic levels. Publications of the Water Research Institute, National Board of Waters, Finland No.49.

McNabb, C.D. 1960. Enumeration of freshwater phytoplankton concentrated on the membrane filter. Limnol. Oceanogr. 5:57-61.

Morris, M.K., L.R. Williams, W.D. Taylor, F.A. Hiatt, S.C. Hern, J.W. Hilgert, V.W. Lambou, and F.A. Morris. 1978. Distribution of phytoplankton in Illinois lakes. USEPA, EPA 600/3-78-050.

Nygaard, G. 1949. Hydrobiological studies of some Danish ponds and lakes. II. (K danske Vidensk. Selsk.) Biol. Sci. 7:131-144.

Palmer, C.M. 1969. A composite rating of algae tolerating organic pollution. J. Phycol. 5:78-82.

Pielou, E.C. 1969. An introduction to mathematical ecology. J. Wiley, New York.

Prescott, G.W. 1978. How to know the freshwater algae, third edition. Wm. C. Brown Company, Dubuque, IA.

Ravera, O. 1983. Assessment of the trophic state of a water body. Annls. Limnol. 19:229-234.

Reynolds, C.S. 1984. The ecology of freshwater phytoplankton. Cambridge University Press, New York.

Rott, E. 1984. Phytoplankton as biological parameter for the trophic characterization of lakes. Verh. Internat. Verein. Limnol. 22:1078-1085.

Shannon, C.E., and W. Weaver. 1963. The mathematical theory of communication. University of Illinois Press, Urbana.

Shapiro, J. 1984. Blue green dominance in lakes: The role and management significance of pH and CO_2. Int. Rev. ges. Hydrobiol. 69:765-780.

Shapiro, J., B. Forsberg, V. Lamarra, G. Lindmark, M. Lynch, E. Smeltzer, and G. Zoto. 1982. Experiments and experiences in biomanipulation - Studies of biological ways to reduce algal abundance and eliminate blue greens, USEPA, EPA 600/3-82-096.

Smith G.M. 1950. The freshwater algae of the United States. McGraw-Hill Book Company Inc., New York.

Somashekar, R.K. 1984. Phytoplankton constituents as indicator of water quality: A study of the River Cauvery. Environ. Studies 23:209-215.

Taft, C.E., and C.W. Taft. 1971. The algae of western Lake Erie, Volume 4. Ohio State University Press. Columbus, OH.

Taylor, W.D., L.R. Williams, S.C. Hern, and V.W. Lambou. 1979. Phytoplankton water quality relationships in U.S. lakes, Part VII: Comparison of some new and old indices and measurements of trophic state. USEPA, EPA 600/3-79-079.

Wetzel, R.G. 1983. Limnology, 2nd ed. Philadelphia: Saunders.

Yang, C.T. 1974. Sedimentation of Lake Charleston. Illinois State Water Survey, Champaign, Illinois.

APPENIDIX A: BIOLOGICAL PARAMETERS IN LAKE CHARLESTON

SITE	DATE	TOTAL GREENS	TOTAL DIATOMS	TOTAL BLUE-GREENS	TOTAL OTHERS	TOTAL ALGAE	CHLOROPHYLL
1	890505	38430	18398	12552	1977	71357	
1	890517	62043	14472	2006	0	78521	
1	890531	78407	7737	13068	1032	100244	
1	890614	59536	19129	13541	6663	98868	33.80
1	890628	40622	12036	14185	4299	71142	32.70
1	890712	49348	11692	56570	5158	122768	55.40
1	890726	61556	21837	57086	8253	148732	149.50
1	890802	55882	16851	56054	0	128786	85.45
1	890809	55022	32669	49004	2235	138931	72.75
1	890816	66414	36538	7523	3009	113483	50.70
1	890822	65052	23212	8597	2006	98868	72.10
1	890905	105746	24072	39117	5588	174524	96.10
1	890919	64479	37828	29517	2866	134690	83.00
1	891003	59493	15475	516	5158	80642	55.70
1	891010	58461	27855	2923	6706	95945	70.00
1	891017	53647	21149	3611	2235	80642	55.40
1	891024	35335	17194	7136	1462	61126	48.10
3	890505	54248	21751	9715	5846	91560	
3	890531	57515	8941	13172	1891	87520	51.60
3	890614	92678	25276	18226	4814	140994	68.75
3	890628	54334	7394	19086	4642	85456	40.75
3	890712	40063	11348	48144	2407	101963	58.70
3	890726	76515	30090	195157	11176	312939	150.20
3	890802	30663	13039	51153	5302	100158	152.85
3	890809	49176	39203	37828	1719	127927	92.75
3	890816	35535	37541	8167	1433	82677	55.75
3	890822	34675	19917	15188	1433	71214	67.90
3	890905	58119	27340	39031	3439	127929	73.00
3	890919	41410	24789	31666	860	98725	77.50
3	891003	49864	21321	1719	4642	77547	60.10
3	891010	43502	27855	3439	4127	78922	78.20
3	891017	55538	20805	6018	3439	85800	60.10
3	891024	46521	9839	2675	1910	60945	49.00

APPENDIX B: CHEMICAL AND PHYSICAL PARAMETERS IN LAKE CHARLESTON

SITE	DATE	SECCHI DEPTH	pH	TOTAL ALKALINITY	HARDNESS	TURBIDITY	TEMPERATURE	DISSOLVED OXYGEN	TOTAL SOLIDS	SUSPENDED SOLIDS	TOTAL PHOSPHORUS
1	890505	16	8.9	121	181	12.0	15.9	9.5	345	30	0.106
1	890517	18	9.0	113	174	11.0	19.0	11.4	335	37	0.108
1	890531	19	8.5	120	182	14.0	22.5	7.2	313	35	0.128
1	890614	13	8.3	125		24.0	23.1	4.1	385	71	0.200
1	890628	10	8.3	124	179	22.0	26.5	5.7	388	39	0.184
1	890712	12	8.4	113	172	17.0	28.2	5.9	388	26	0.148
1	890726	10	8.3	114	166	18.0	26.0	9.1	295	36	0.214
1	890802	14	8.1	116	162	13.0	26.0	3.2	368	33	0.186
1	890809	14	8.3	119	172	18.0	24.9	5.2	315	46	0.179
1	890816	13	8.3	123	170	24.0	24.9	4.4	308	46	0.164
1	890822	8	8.1	121	168	28.0	24.9	5.9	268	67	0.166
1	890905	9	7.9	122	169	40.0	23.2	4.9	395	89	0.222
1	890919	11	8.1	121	164	27.0	19.0	8.2	278	44	0.140
1	891003	22	8.5	121	171	13.0	16.2	8.3	263	32	0.110
1	891010	16	8.6	127	175	19.0	13.0	11.4	340	37	0.128
1	891017	15	8.4	127	176	15.0			268	36	0.134
1	891024	17	8.7	127	179	7.7	10.8	14.5	270	17	0.084
	AVERAGE	**14**	**8.4**	**121**	**173**	**19.2**	**21.5**	**7.4**	**321**	**42**	**0.152**
	STD	**3.7**	**0.3**	**4.5**	**5.8**	**7.6**	**5.1**	**3.0**	**43.3**	**17.2**	**0.039**
3	890505	20	8.9	121	182	8.2	15.9	9.6	298	19	0.108
3	890531	22	8.7	118	178	7.5	23.0	8.6	300	16	0.122
3	890614	15	8.7	128		12.0	24.5	9.0	395	55	0.172
3	890628	14	8.7	124	178	16.0	27.5	8.9	270	31	0.128
3	890712	16	8.6	116	173	12.0	28.0	8.5	300	26	0.148
3	890726	13	9.2	109	166	9.0	29.3	17.2	263	31	0.171
3	890802	18	8.9	115	166	10.0	27.5	9.5	270	31	0.180
3	890809	13	8.5	119	172	13.0	25.1	6.7	295	38	0.154
3	890816	19	8.4	122	172	11.0	26.2	6.0	290	30	0.122
3	890822	20	8.3	120	170	10.0	24.7	7.0	250	23	0.110
3	890905	17	8.4	121	168	15.0	23.0	8.0	340	33	0.118
3	890919	18	8.9	116	160	11.0	19.6	13.9	235	15	0.094
3	891003	15	8.8	122	169	18.0	16.2	9.3	280	40	0.126
3	891010	16	8.6	127	175	16.0	13.0	11.8	295	27	0.128
3	891017	13	8.4	126	176	18.0	9.6	15.8	248	29	0.126
3	891024	19	8.8	128	178	7.1	10.9	13.2	263	16	0.084
	AVERAGE	**17**	**8.7**	**121**	**172**	**12.1**	**21.4**	**10.2**	**287**	**29**	**0.130**
	STD	**2.7**	**0.2**	**5.1**	**5.6**	**3.5**	**6.2**	**3.2**	**37.5**	**10.0**	**0.027**

APPENDIX C. ALGAE IDENTIFIED IN LAKE CHARLESTON WATER SAMPLES.

GREEN ALGAE

Actinastrum sp.
Ankistrodesmus sp.
Aphanothece sp.
Chlamydomonas sp.
Chlorella sp.
Chodatella quadriseta
Closteridium sp.
Coelastrum sp.
Crucigenia alternans
Crucigenia apiculata
Crucigenia fenestrata
Crucigenia quadrata
Crucigenia rectangularis
Crucigenia tetrapedia
Dictyosphaerium sp.
Eudorina sp.
Golenkinia sp.
Gonium sociale
Kirchneriella lunaris
Oocystis sp.
Pandorina sp.
Pediastrum duplex
Pediastrum tetras
Polyedriopsis sp.
Scenedesmus acuminatus
Scenedesmus bijuga
Scenedesmus dimorphis
Scenedesmus quadricauda
Selenastrum sp.
Tetrastrum elegans
Tetastrum staurogeniform
Tetraedron sp.

DIATOMS

Cyclotella meneghiniana
Diatoma sp.
Fragilaria sp.
Melosira distans
Melosira granulata
Navicula sp.
Nitzchia acicularis
Nitzchia holsatica
Pinnularia sp.
Synedra tenera
Tabellaria sp.

BLUE GREEN ALGAE

Anabaena sp.
Aphanizomenon flos-aquae
Merismopedia sp.
Microcystis sp.
Oscillatoria sp.
Spirulina sp.

OTHER ALGAE

Ceratium sp.
Closterium sp.
Cosmarium sp.
Euglena sp.
Peridinium sp.
Phacus sp.
Staurastrum sp.
Trachelomonas hispida