GROWTH RESPONSE OF A STUNTED BLUEGILL POPULATION

FOLLOWING HYDROPHYTE REMOVAL BY DIURON

By

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PREFACE

The objective of this study was to ascertain whether annual growth rate of a stunted bluegill population could be affected by hydrophyte removal.

I wish to express my sincere appreciation to my major adviser, Dr. Robert C. Summerfelt, for his professional guidance and expertise. I also appreciate the advice and assistance of my committee members: Dr. Dale W. Toetz, Dr. Troy C. Dorris, and Dr. Austin K. Andrews. Thanks are also due to Dr. William A. Drew for assistance in identifying insects; Dr. Maureen Diggins for her help in analyzing and evaluating bluegill food habits; and to the many friends and employees of the Oklahoma Cooperative Fishery Unit who aided in data collection.

This study was financially aided by the Oklahoma Cooperative Fishery Unit, the Oklahoma State University Reservoir Research Center, and the Oklahoma State University Research Foundation. The diuron formulation "Aqua-trol" was donated and applied by Mr. Tom Graham, of the H and T Chemical Company, of Oklahoma City. The City of Stillwater granted use of Sanborn Lake for the study.

Lastly, I would like to thank my parents, Mr. and Mrs. Tony A. Mnich "just because," and our Lord Jesus Christ for His countless blessings.

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CHAPTER I

INTRODUCTION

Most literature on control of vascular aquatic plants (hydrophytes) emphasizes the effectiveness of control methods (herbicidal, biological, mechanical) and acute toxicities of herbicides to fish and other organisms. Effects of hydrophyte removal on fish growth have received little attention. Hydrophytes (1) provide cover for fish and fish-food organisms, (2) contribute production to the hydrophyte and detritus-based food webs, and (3) compete for plant nutrients which could be used in the phytoplankton-based food web.

Variation in susceptability (vulnerability) of prey species, such as the bluegill (*Lepomis machrochirus*), may be the major factor in determining their utilization as food by predaceous fishes (Lewis 1967). Hydrophytes provide cover for bluegill, and decrease their vulnerability to predation by largemouth bass (*Micropterus salmoides*). Hydrophyte removal may affect such predator-prey relationships. Growth of largemouth bass might improve following hydrophyte removal due to increased vulnerability and utilization of bluegill. A decrease in numbers of bluegill, especially young-of-the-year (YOY), should also indirectly benefit their growth by reducing intraspecific competition. Hydrophyte removal may also make invertebrate prey more available to bluegill. Improved bluegill growth through increased consumption of macroinvertebrates may be temporary, if predation decimates invertebrate production.

A decline in invertebrate production could then lead to diminished fish growth.

Hydrophyte removal may diminish the habitat and food source of many aquatic invertebrates. Detritophagic invertebrates may increase in abundance from decay of hydrophytes left in the lake, as is generally the case, following a herbicide treatment. Zooplankton which are grazers on phytoplankton may increase in abundance after the decay of hydrophytes if a phytoplankton bloom develops from release of plant nutrients previously bound in biomass of the hydrophytes. Resultant changes in fish growth would be species specific according to food habits and the degree of manifestation of any or all of the above mentioned processes.

Bluegill collected in 1969 and 1970 from Sanborn Lake, a 4.58 hectare pond in Stillwater, Payne County, Oklahoma, exhibited poor growth; and largemouth bass were too few to provide effective predatory control. As noted by Lewis (1967), failure of the bass bluegill combination is, in part, due to the high vulnerability of bass eggs to bluegill predation. In November 1969 the pond was treated with 1 ppb Antimycin A to selectively thin the sunfish population; and about 2000 fingerling largemouth bass about 75-100 mm total length, were stocked to enhance predator pressure. About 1.8 kg/ha of the small (150 mm) dead bluegill were collected, but this thinning produced only a 9 mm increase of growth increment in 1970 for the survivors of the 1969 YOY.

In 1971 the pond contained a nuisance growth of hydrophytes covering approximately 68 percent of the surface area. An initial application of 0.07 mg/l diuron (3-(3,4-dichlorophenyl)-1,1-dimethylurea) was made to the pond in April 1972 to obtain data on the efficacy and safety of this herbicide. A follow-up spot treatment of 0.18 mg/l was made six weeks later to ensure hydrophyte mortality. The removal of hydrophytes provided an opportunity to evaluate the effect on bluegill growth. This study compares the growth and condition of the bluegill population before and after removal of hydrophytes. Numerical and biomass densities of hydrophytes and macroinvertebrates, as well as bluegill food habits, are compared before and after herbicide treatment to provide insight into factors affecting bluegill growth and condition.

CHAPTER II

EFFICACY OF HYDROPHYTE CONTROL USING DIURON IN AN OKLAHOMA POND

Introduction

Diuron, 3-(3,4-dichlorophenyl)-1,1-dimethyurea, is a nonspecific, systemic herbicide that has been used variously as an aquatic herbicide since the early fifties. It is efficacious in controlling most aquatic plants at application rates of 0.2-1.5 mg/l, and its toxicity to non-target organisms is rather low (Johnson and Julin 1974). For these reasons, it appears that diuron should have a promising future in fisheries management as a means of controlling nuisance growths of algae and hydrophytes.

In spite of its desirable qualities, diuron has not been registered by the Environmental Protection Agency Registration Division, principally because data on persistance in the aquatic environment and chronic toxicity to non-target organisms is insufficient (Johnson and Julin 1974). Diuron was registered in 1974 for use as an aquatic herbicide under restricted use by Michigan, Texas, and Oklahoma (Johnson and Julin 1974). In Oklahoma, diuron was registered in 1972 and 1973 under the product name of Aqua-trol, a preparation containing 70 percent diuron, formulated by the H and T Chemical Company, Oklahoma City. Their recommended application rate was 2.24 kg/ha. Such an area-based application rate disregards volume and allows variation in concentrations applied. An area-based application rate may be valid, however, because diuron is not very soluble in water, so it sinks to the bottom where it is taken up by plants through the roots.

Aqua-trol was applied to Sanborn Lake 22 April, 1972. Hydrophyte biomass in total and by species was compared for September 1971 and October 1972 between two depth strata (0-1 m and 1-3 m). April 1972 (before treatment) 0-1 m samples of hydrophytes were obtained to indicate the normal winter "dieback" in hydrophyte biomass. Samples were taken May and July 1972 to describe the rate of biomass decrease due to Aqua-trol treatment. Temperature and dissolved oxygen were measured 40 times from 15 April - 7 July to monitor potential depletion from the decaying hydrophytes.

Description of Study Area

This study was done on Sanborn Lake (Figure 1) located adjacent to the Stillwater Municipal Airport, Payne County, Oklahoma. The pond, constructed in 1962, has a surface area of 4.58 hectares at spillway elevation (286.0 m mean sea level), an average depth of 1.7 m, and a maximum depth of 3.8 m. Water elevation from April to October 1972 declined from 285.7 m to 285.2 m. Elevation was not determined in 1971, but was approximately 285.6 m.

The pond is normally clear because of its well-sodded watershed--10.9 and 19.5 Jackson turbidity units at 0.5 m and 3.5 m depths on 28 April, 1972. The watershed is partially mowed prairie grassland with underlying silt loam and breaks-alluvial land complex soils. The pond was thermally stratified from 16 May, 1972 to 14 June, 1972 (Figure 3).

Aquatic vegetation (Figure 2) covered approximately 68 percent of

Figure 1. Bathymetric map of Sanborn Lake

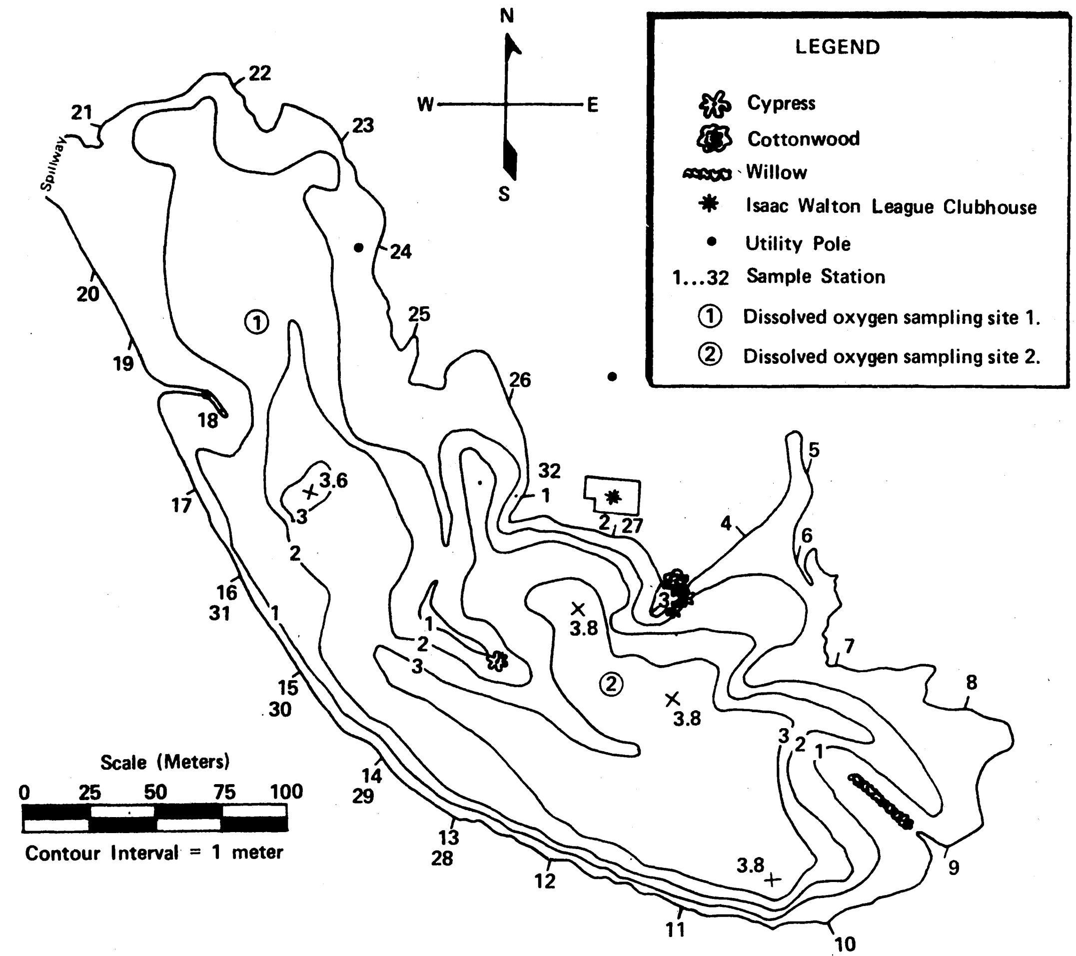
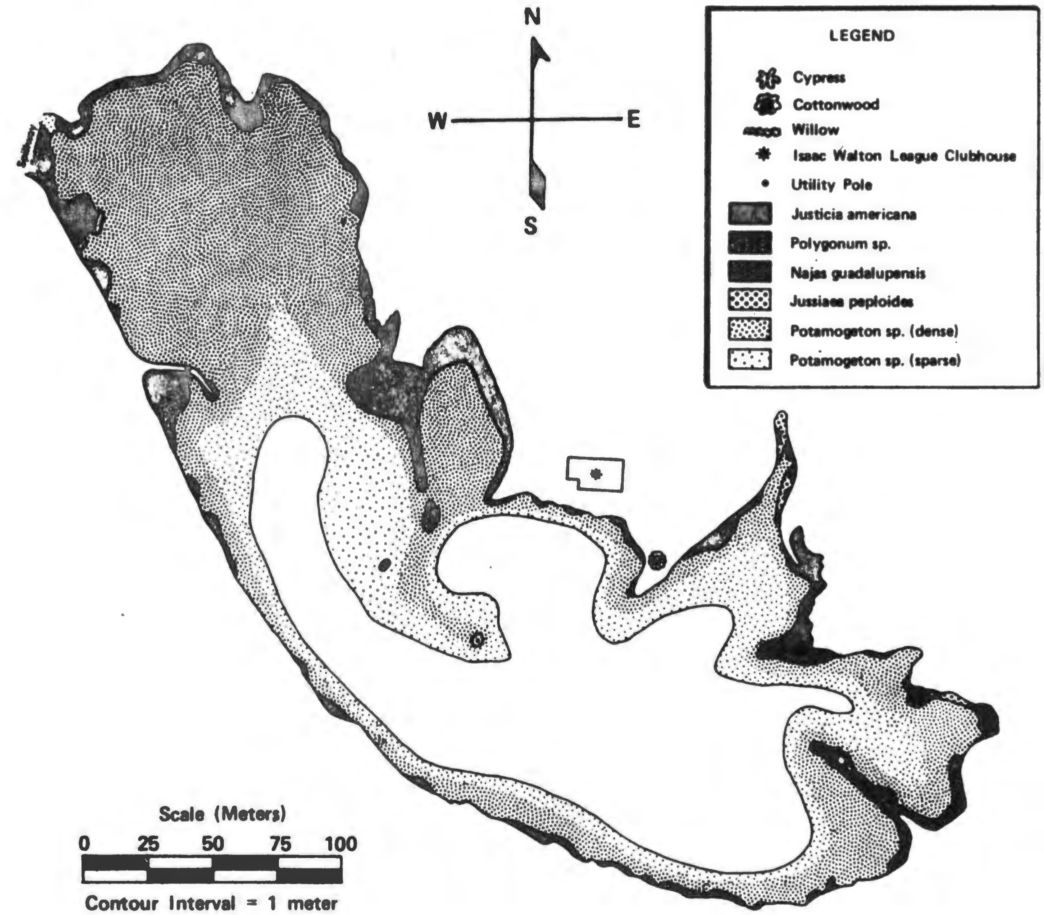

N
S
W
E
LEGEND
Cypress
Cottonwood
Willow
Isaac Walton League Clubhouse
Utility Pole
1...32 Sample Station
① Dissolved oxygen sampling site 1.
② Dissolved oxygen sampling site 2.
Spillway
Scale (Meters)
0 25 50 75 100
Contour Interval = 1 meter
3.6
3.8
3.8
3.8

Figure 2. Map of Sanborn Lake showing visually estimated areal coverage by hydrophytes, September 1971.


N
W
E
S
LEGEND
Cypress
Cottonwood
Willow
Isaac Walton League Clubhouse
Utility Pole
Justicia americana
Polygonum sp.
Najas guadalupensis
Jussiaea peploides
Potamogeton sp. (dense)
Potamogeton sp. (sparse)
Scale (Meters)
0 25 50 75 100
Contour Interval = 1 meter

the water surface area in September, 1971. Plants at the water's edge included *Typha sp.*, *Polygonum hydropiperoides*, *Polygonum pensylvanicum*, *Sagittaria latifolia*, *Eleocharis quandangulta*, *Eleocharis macrostachya*, *Cyperus erythrorhizos*, *Cyperus uniflorus*, *Juncus crassifolius*, *Juncus marginatus*, *Salix sp.*, *Phyla lanceolata*, and *Justicia americana*. *Justicia* also was an emergent in the water out to a depth of approximately one meter. Submergents were *Najas quadalupensis* and *Chara sp.*--the latter found in April, 1972, but not in October, 1971. *Potamogeton nodosus* and *P. amplifolius*, submergents with floating leaves, were the dominant macrophytes of the pond. *Jussiaea peploides* grew in a few, small isolated patches in the pond.

Methods and Materials

Herbicide Application

Sanborn Lake was initially treated 22 April 1972 with 2.01 kg/ha (0.10 mg/l) Aqua-trol by forced spraying of an aqueous Aqua-trol solution from a boat. This was equivalent to 0.07 mg/l diuron. At the time of the treatment, alkalinity ranged from 128-164 bicarbonate, pH from 7.4-8.0, water temperature was 16.5 C. Because it was felt that hydrophytes were not dying rapidly enough, a second treatment was applied 3 June 1972 to the north-northwest half of the pond that contained densely-growing *Potamogeton*. This application was made by hand-broadcasting Aqua-trol powder from the back of a boat at the rate of 3.04 kg/ha (0.26 mg/l Aqua-trol, 0.18 mg/l diuron) to this portion of the pond.

Hydrophyte Sampling

Aquatic plant samples were collected in October 1971 and 1972 (water temperature between 15-20 C), in April 1972 prior to herbicide treatment, May 1972 four weeks after the first treatment, and July 1972 ten weeks after the first and six weeks after the second treatment. Collections were made according to a stratified-random design for two depth contours, 0-1 m and 1-3 m. Samples were not taken for depths greater than 3 m because of the absence of substantial vegetation and the lack of means to obtain adequate samples from those depths. The number of samples taken from each depth contour was weighted according to the area in each contour. The area of the 1-3 m contour interval was twice the 0-1 m area, so twice the number of samples taken between 1-3 m (n=12) as between 0-1 m (n=6). April, May, and July, 1972 collections were made only in the 0-1 m strata.

Sampling stations were randomly selected from 26 predetermined points equidistant around the lake shore; sample stations 27-32 were used to sample the ridge extending from the cypress northward (Figure 1). Each station was assigned a predetermined landmark on which to draw a line of sight. From the randomly selected station on shore, the collector would sight on that station's landmark, and proceed to move along that line of sight one-half the surface distance to the next strata contour interval. The sampling cylinder was set into the pond bottom at this location. When the same station was randomly selected for two or more collection dates, the actual site was moved 1 m to the right to avoid sampling the exact location.

Quantitative estimates of the vegetation were obtained from inside the 1.22 m tall, 0.44 m diameter, 0.15 m^2 area sheet metal cylinder. The procedure and apparatus are a modification of that used by Gerking (1957). Hydrophyte samples were obtained by setting the cylinder into the pond bottom, using a 0.91 m tall extension for the 1-3 m stratum. When in place, depth was determined, and all vegetation, including roots, was removed using a spade and rake. Spade and rake were mounted on long poles for deep-water samples. The vegetation was placed in a plastic bag or bucket and later rinsed, sorted to genera, dried at 105 C, and weighed to the nearest 0.1 g.

The efficiency of the hydrophyte extraction procedure was tested using seven September 1971 0-1 m samples. Hydrophyte biomass was obtained by the extraction procedure mentioned above, then the cylinder was removed, and additional "missed" biomass was pulled by hand from the circle which the cylinder left in the mud. An average of 22.86 percent of the total biomass remained, so hydrophyte biomass obtained by the extraction procedure in the 1-3 m stratum was multiplied by 1.296 to correct for the biomass that was probably missed.

Water temperature and the concentration of dissolved oxygen were measured periodically at sunrise and sunset from 15 April - 7 July 1972, at 0.5 m depth intervals at two locations in the pond using a galvanic cell temperature-oxygen probe and meter. Station 1, 1.5 m depth, was located in the area of densly-growing Potamogeton at the northern end of the pond (Figure 1). Station 2, 3.5 m depth, was located in a non-vegetated area at the deeper, south end of the pond (Figure 1).

Results and Discussion

Hydrophytes

In October 1971, about 68 percent of the surface area of Sanborn Lake was covered with aquatic plants. Their biomass density, average dry weight biomass of all plant species, was 303.3 g/m^2 (481.2 g/m^3) in the 0-1 m stratum and 215.8 g/m^2 (150.3 g/m^3) in the 1-3 m stratum (Table 1). This biomass density is comparatively greater than that in several larger lakes: 179.2 g/m^2 (0-1 m) and 268.8 g/m^2 (1-3 m) in Lake Mendota (Rickett 1922); 175 g/m^2 (0.0-4.4 m) in University Bay, Lake Mendota (Lind and Cottam 1969); 67.2 g/m^2 (0-1 m) and 219.5 g/m^2 (1-3 m) in Green Lake, Wisconsin (Rickett 1924). Also in several lakes of the Illinois River Valley, Illinois, Low and Bellrose (1944) found an average of 127 g/m^2. In Par Pond, South Carolina, standing crops were higher than in Sanborn Lake for eight species, lower for five species, and nearly the same for three species (Polisini and Boyd 1972). The standing crops in Par Pond were mostly for emergents, whereas in Sanborn Lake, the majority of the plants were submergents. Standing crops of Chara and Myriophyllum in 4.6 hectare Lake Ösbysjön, Sweden, ranged from 130-1130 g/m^2.

In 1971, the difference in the biomass density of hydrophytes (g/m^3) between 0-1 m (481.2 g/m^3) and 1-3 m (150.3 g/m^3) was significant ($P<0.001$), but expected since penetration of light necessary for photosynthesis diminishes with depth. A correlation coefficient of -0.91 ($N=16$) for the regression $Y = 642.42 - 1.50 X$, where Y = Potamogeton biomass, X = depth, and N = sample size, further substantiates the relationship between depth and biomass hydrophytes.

Table 1. Average biomass density and ninety-five percent confidence limits (g/m^2 and g/m^3) of hydrophytes in Sanborn Lake for depth strata and dates indicated.

Unit	Sept. 1971 (0-1m) (6)[1]	Sept. 1971 (1-3m) (12)	April 1972 (0-1m) (6)	May 1972 (0-1m) (8)	July 1972 (0-1m) (7)	October 1971 (0-1m) (6)	October 1971 (1-3m) (12)
			Potamogeton amplifolius and *P. nodosus*				
g/m^3	358.1± 80.2	150.3±61.3	80.5± 91.5	50.3± 38.1	36.6± 44.9	0.3±0.8	0.4±0.4
g/m^2	220.4±125.8	243.0±79.0	44.2± 52.2	33.7± 23.0	21.1± 24.7	0.2±0.6	0.5±0.4
			Justicia americana				
g/m^3	108.7±125.8	0.0± 0.0	786.9±2023.1	498.1±1079.3	202.6±447.0	0.0±0.0	0.0±0.0
g/m^2	76.9±170.8	0.0± 0.0	320.0± 822.7	320.9± 551.4	114.5±249.4	0.0±0.0	0.0±0.0
			Najas quadalupensis				
g/m^3	14.4± 23.6	0.0± 0.0	1.6± 4.0	0.0± 0.0	0.0± 0.0	0.0±0.0	0.0±0.0
g/m^2	5.9± 8.0	0.0± 0.0	0.9± 2.3	0.0± 0.0	0.0± 0.0	0.0±0.0	0.0±0.0
			Chara sp.				
g/m^3	0.0± 0.0	0.0± 0.0	113.4± 220.5	6.2± 14.6	0.0± 0.0	0.0±0.0	0.0±0.0
g/m^2	0.0± 0.0	0.0± 0.0	56.0± 106.1	2.7± 6.3	0.0± 0.0	0.0±0.0	0.0±0.0
			Total				
g/m^3	481.2±140.0	150.3±61.3	982.4±1933.7	554.5±1058.8	239.2±433.8	0.3±0.8	0.4±0.4
g/m^2	303.3±136.0	243.0±79.0	420.9± 775.9	152.9± 205.2	135.6±242.1	0.2±0.6	0.5±0.4

[1] Sample size follows depth stratum for each date.

In April 1972 the average total biomass density of hydrophytes in the 0-1 m stratum was 982.4 g/m^3 compared to 481.2 g/m^3 in October 1971 (Table 1), but the difference was nonsignificant ($P>0.500$) due to the high variance. The high average biomass density in the April 1972 collections was due primarily to one sample which contained 4,721.3 g/m^3 of the emergent, Justicia americana. Also, overwinter emergence of Chara sp., not present in October, 1971, contributed to the increase in average biomass density in the spring. Apparently in the summer of 1971 Chara was shaded out by the Potamogeton, but during the winter, when much of the Potamogeton foilage died and collapsed, growth of Chara was substantial. The significant ($P<0.001$) September 1971 to April 1972 declined in average biomass of Potamogeton sp. from 150.3 g/m^3 to 80.5 g/m^3 is an apparently typical overwinter condition.

Average total biomass of hydrophytes in Sanborn Lake declined after the April 22 herbicide treatment. The decline was not significant at the five-percent level between April-May or April-July, because of the high variances and small sample sizes used; however, the decreased abundance of hydrophytes between the April-October, 1972, and October 1971-October 1972 samples were highly significant ($P<0.001$) for both 0-1 m and 1-3 m strata.

All plants in contact with the water were affected, but death of the emergent plants was slower than submergents. Other workers also have observed that emergents are more difficult to control than submergents (Johnson and Julin 1974). This study is apparently the first to report control of Justicia americana with diuron. No quantitative measurements were made of phytoplankton, but based on the visual change in clarity and color of the water, phytoplankton were killed by diuron.

Phytoplankton appeared to have repopulated within one month after the second treatment.

Because few hydrophytes appeared affected by the first application of 0.07 mg/l, a second treatment was made which provided a combined dosage of 0.25 mg/l. Other workers have used at least 0.5 mg/l diuron for submergents and at least 1.0 mg/l for emergents to gain effective control (Johnson and Julin 1974). However, the H and T Chemical Company had reported effective control at 2.24 kg/ha which would have given a dosage of 0.10 mg/l in Sanborn Lake. Diuron from the first treatment should not have been in solution at the time of the second treatment because it is not very soluble in water, but residues would have been expected in the hydrosol (Johnson and Julin 1974). I do not know whether the 0.18 mg/l concentration of the second treatment, the combined dosage of 0.25 mg/l, or an intermediate diuron concentration produced the control. In any event, the concentrations of diuron used in the present study are less than for most previously reported studies.

Dissolved Oxygen

Dissolved oxygen concentrations measured at sunrise at the surface of station one decreased gradually from 7.0 mg/l on 23 April to 5.0 mg/l on 23 May (Figure 3). Dissolved oxygen (DO) at the bottom decreased from 6.6-1.5 mg/l over the same period (Figure 3). Surface and bottom DO measurements at station 2 decreased from 7.7 to 5.5 mg/l and 7.2-0.8 mg/l, respectively (Figure 4). Dissolved oxygen at the surface of both stations increased between 25 May and 3 June to near pretreatment concentrations, but decreased following treatment two on 3 June from 6.3 mg/l (sta. 1) and 7.8 mg/l (sta. 2) to 5.1 mg/l and 5.9 mg/l on 10 June

(Figures 3 and 4).

The pond was thermally stratified in the deeper areas from approximately 13 May-22 June (Figures 4 and 7). A large storm and cold front passed through the area 20 to 22 June and apparently accounts for the uniform 4 mg/l DO concentration on the morning of 22 June (Figure 4). Dissolved oxygen was 7 mg/l at the surface by evening, however, and within a few days the pond was again stratified.

The Aquatic Life Advisory Committee of ORSANCO has recommended a dissolved oxygen concentration in warm-water fish habitats of not less than 5 mg/l for more than 16 hours per 24-hour period, and at no time less than 3 mg/l (McKee and Wolf 1971). Dissolved oxygen concentrations at the surface and in the water column at station one remained above 5 mg/l for approximately 25 days after treatment 1 (23 April-17 May), except for the bottom 0.25 m of the water column just above the decaying vegetation (Figure 5). From approximately 20-30 May, dissolved oxygen was equal to or less than 5 mg/l throughout the entire water column in the morning, but was 5.0-6.0 mg/l in the upper 1.25 m by the evenings. The 3 mg/l isopleth at station 1 included only the bottom 0.25-0.50 m (Figure 7).

Just prior to treatment two, DO concentrations measured at sunrise at station 1 were greater than 5 mg/l throughout the water column (Figure 5). Within six days after treatment two, the 5 and 3 mg/l oxygen levels were again evident in the lower 0.25-0.50 m of the water column. Dissolved oxygen concentrations were approximately 5 mg/l throughout the water column on the mornings of 8, 16, and 22 June, when the water column was mixed; DO concentrations were greater than 5 mg/l throughout most of the water column by sunset on 16 and 22 June, but remained

Figure 3. Dissolved oxygen concentration measured at sunrise at the surface and bottom of station one, 15 April-7 July, 1972.

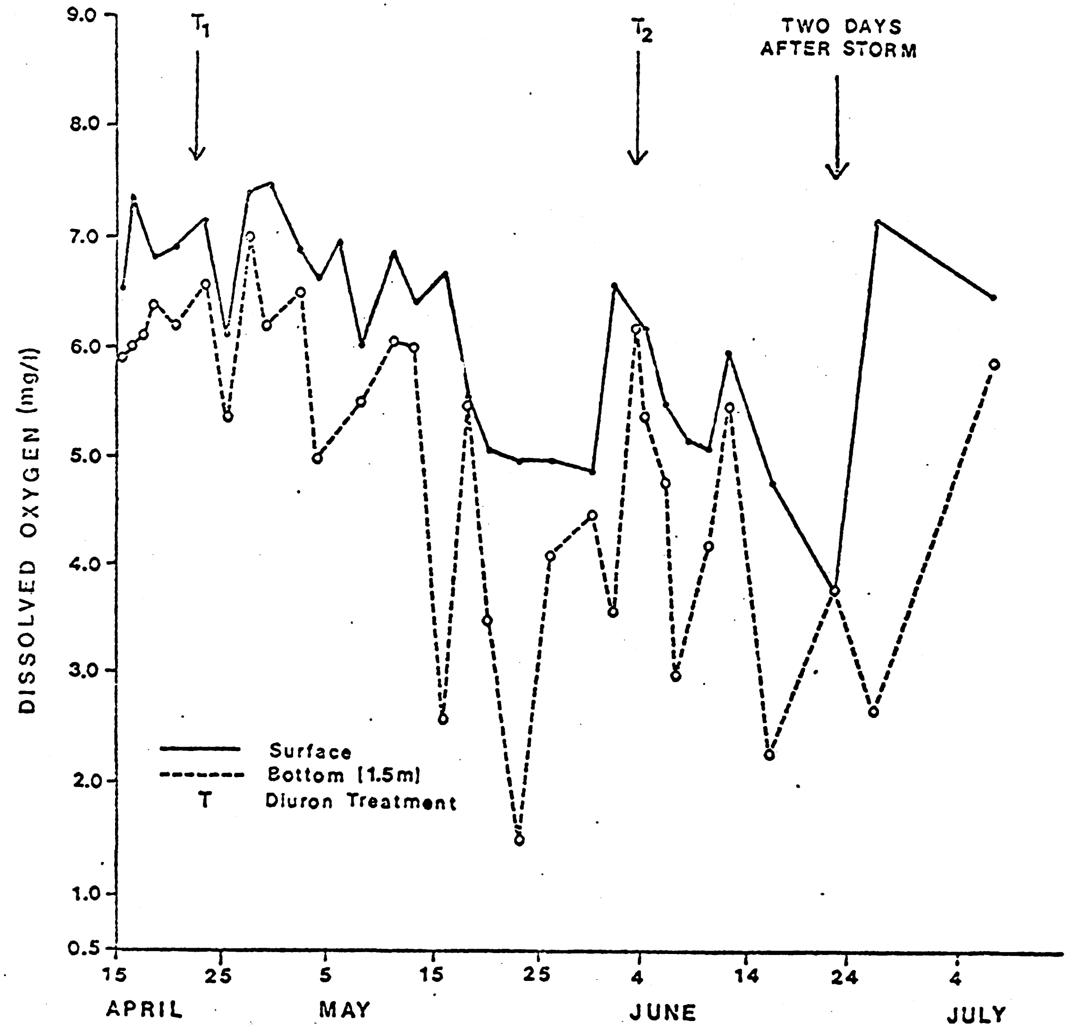
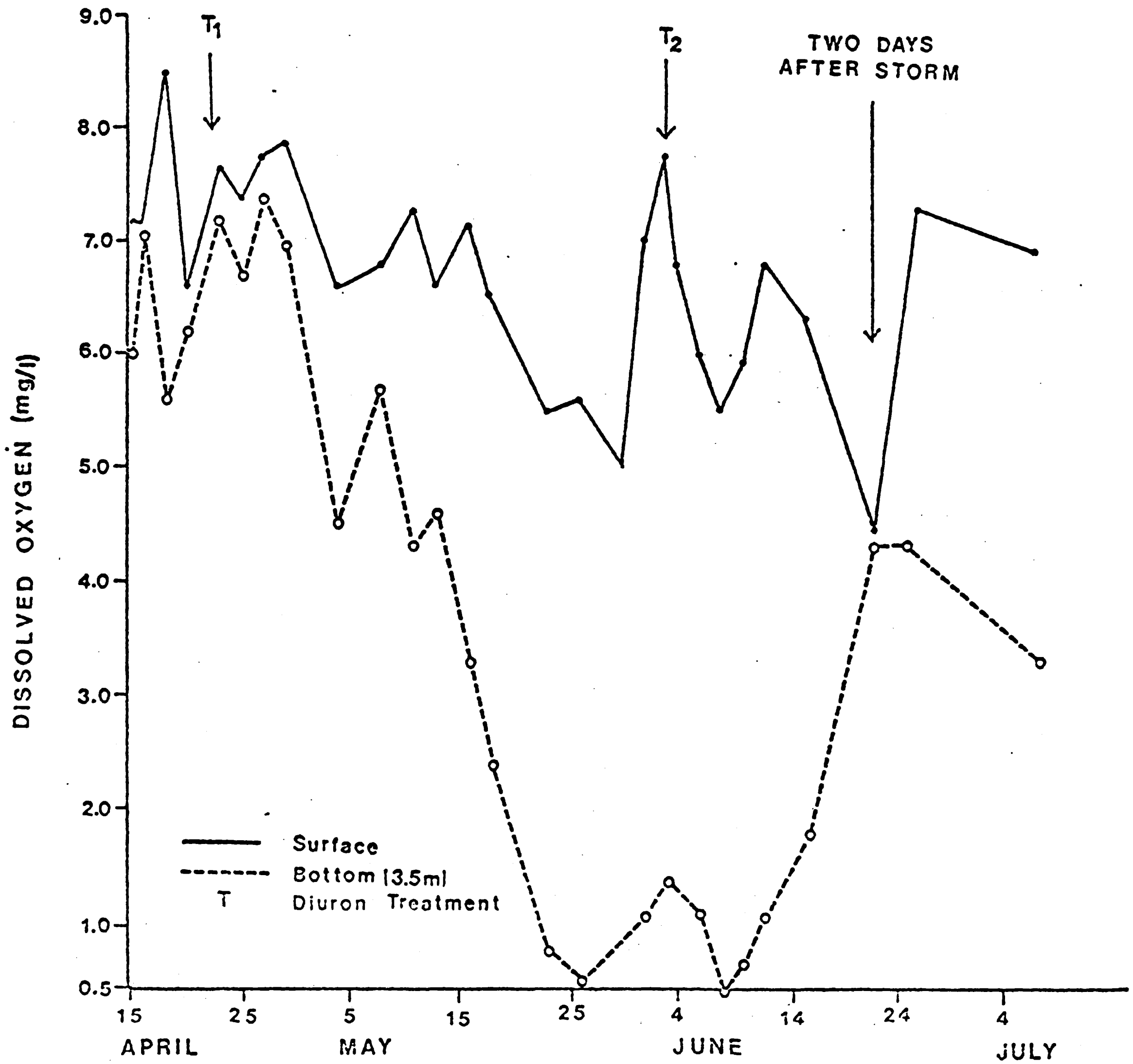

T1
T2
TWO DAYS AFTER STORM
DISSOLVED OXYGEN (mg/l)
9.0
8.0
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.5
Surface
Bottom (1.5m)
T Diuron Treatment
15
25
5
15
25
4
14
24
4
APRIL
MAY
JUNE
JULY

Figure 4. Dissolved oxygen concentration measured at sunrise at the surface and bottom of station two, 15 April-7 July, 1972.



DISSOLVED OXYGEN (mg/l)
9.0
8.0
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.5
T1
T2
TWO DAYS AFTER STORM
Surface
Bottom (3.5m)
T Diuron Treatment
15
25
5
15
25
4
14
24
4
APRIL
MAY
JUNE
JULY

Figure 5. Depth of the 5 mg/l dissolved oxygen isopleth at station one, 15 April-7 July, 1972.

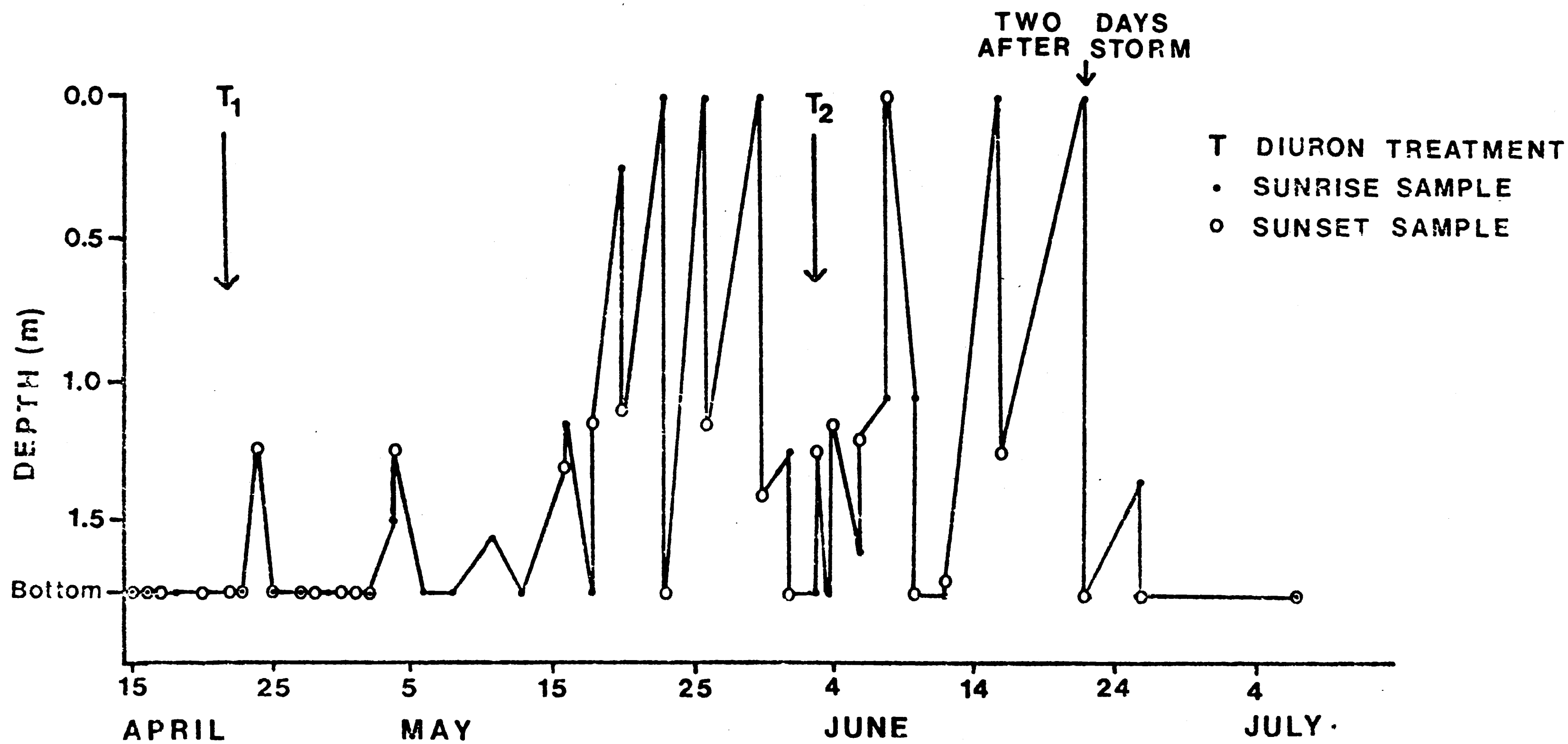

TWO DAYS
AFTER STORM
T_1
T_2
0.0
0.5
1.0
1.5
Bottom
DEPTH (m)
15
25
5
15
25
4
14
24
4
APRIL
MAY
JUNE
JULY
T DIURON TREATMENT
• SUNRISE SAMPLE
o SUNSET SAMPLE

Figure 6. Depth of the 5 mg/l dissolved oxygen isopleth at station two, 15 April-7 July, 1972.

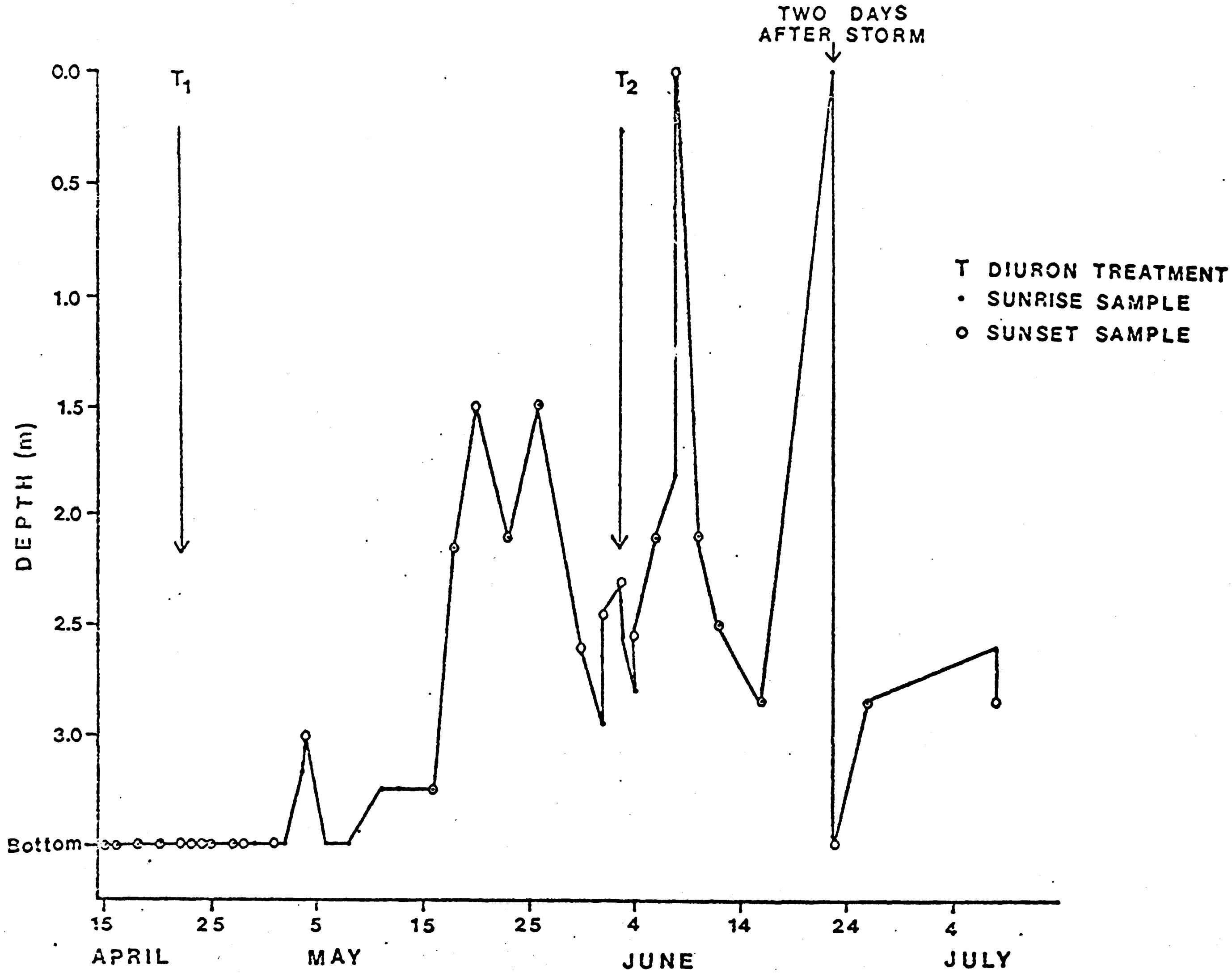
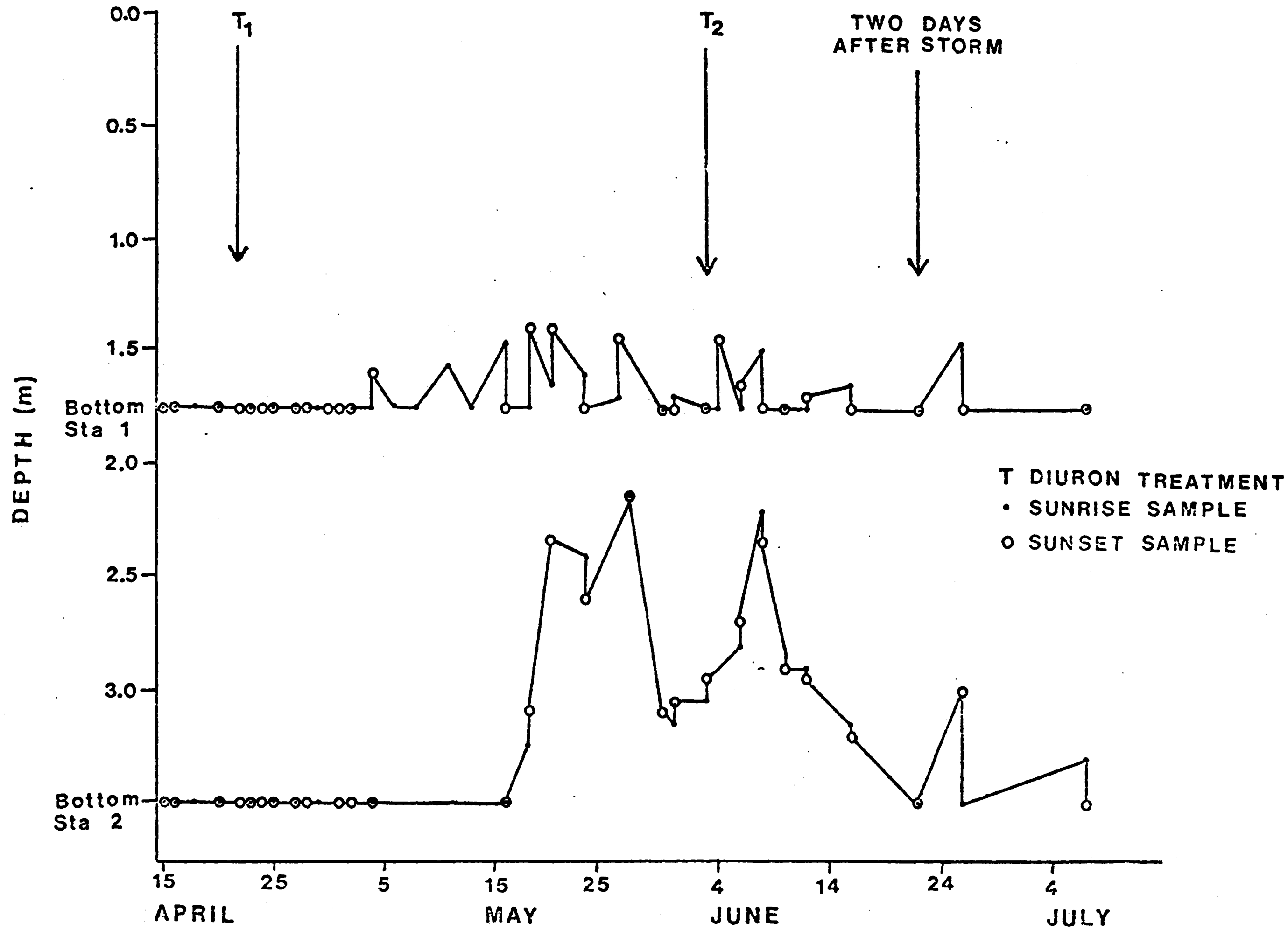

TWO DAYS AFTER STORM
T1
T2
0.0
0.5
1.0
1.5
2.0
2.5
3.0
Bottom
DEPTH (m)
T DIURON TREATMENT
• SUNRISE SAMPLE
o SUNSET SAMPLE
15
25
5
15
25
4
14
24
4
APRIL
MAY
JUNE
JULY

Figure 7. Depth of the 3 mg/l dissolved oxygen isopleth at stations one and two, 15 April-7 July, 1972.



T1
T2
TWO DAYS
AFTER STORM
DEPTH (m)
0.0
0.5
1.0
1.5
2.0
2.5
3.0
Bottom
Sta 1
Bottom
Sta 2
T DIURON TREATMENT
• SUNRISE SAMPLE
o SUNSET SAMPLE
15
25
5
15
25
4
14
24
4
APRIL
MAY
JUNE
JULY

between 4 and 5 mg/l throughout the day on 8 June.

In general, the 5 mg/l dissolved oxygen isopleth did not rise above 1.25 m in depth in the northern section of the pond following the two diuron treatments, except for 7 brief periods when the isopleth extended to the surface following weather changes which induced destratification (Figure 5).

Dissolved oxygen concentrations at station two were less than 5 mg/l in the lower one-half of the water column between 20 and 30 June (Figure 6). The 3 mg/l isopleth was found in the lower 0.5-1.5 m of the water column from approximately 20 May-14 June (Figure 7). Dissolved oxygen in a stratum of water 0.5 m above the bottom ranged between 0.5 and 1.5 mg/l between 20 May and 14 June. Such low concentrations of dissolved oxygen may have limited the vertical distribution of fish to the upper two-thirds of the pond for at least a month. These oxygen levels probably did not preclude brief feeding excursions by fish into the less oxygenated areas. No mortality of fish was observed.

Depletion of dissolved oxygen following diuron treatment and hydrophyte decay has been reported by many investigators (Johnson and Julin 1974). Oxygen depletion during summer stratification may reduce macroinvertebrate standing crops (Bergersen 1969, Fast 1971, Inland Fisheries Branch 1970). Between 20 May and 14 June 1972, only the upper 2-3 m of the water column of Sanborn Lake contained more than 3 mg/l dissolved oxygen, hence it is quite likely that the 38% reduction in numerical density of macroinvertebrates in the 1-3 m stratum between September 1971 and October 1972 was caused by low oxygen levels during the summer of 1972.

Conclusions

An April 1972 treatment of Sanborn Lake with Aqua-trol at 2.01 kg/ha (0.07 mg/l diuron) did not completely control submergent and emergent hydrophytes within five weeks; but a second "spot" treatment six weeks later at the rate of 3.04 kg/ha (0.18 mg/l diuron) resulted in complete control. The maximum total dosage to the entire pond of Aqua-trol in the two applications was 2.25 kg/ha (0.09 mg/l diuron); and the maximum "spot" dosage to vegetated areas was 5.05 kg/ha (0.25 mg/l diuron). The emergent, *Justicia americana*, not previously reported in diuron applications was effectively controlled by diuron at this rate with this formulation. Six 0-1 m samples taken in October 1973 show that hydrophyte biomass density returned to 38% of pretreatment levels during 1973 (R. C. Summerfelt, personal communication).

Dissolved oxygen concentrations were less than the 3-5 mg/l critical levels (McKee and Wolf 1971) in the lower 0.25-2.5 m of the water column in stratified deeper areas from 28 days after treatment one (20 May) until observations were ended, 7 July 1972. Fish mortality was not observed, and the DO levels in a substantial volume of the pond remained above critical levels such that fish were probably little affected. No fish were killed and numerical and biomass densities of macroinvertebrates in the 0-1 stratum were greater in 1972 than in 1971. There was a decrease in macroinvertebrate density for the 1-3 m samples which may indicate that dissolved oxygen became more limiting to the macroinvertebrates in the deeper strata. Species diversity declined from 1971 to 1972; apparently because of reduction in habitat provided by the hydrophytes and because of selective mortality of species sensitive to low dissolved oxygen.

In general, it seems that diuron is an effective and reasonably safe herbicide to use for one-season control of hydrophytes.

CHAPTER III

THE MACROINVERTEBRATE COMMUNITY OF AN OKLAHOMA POND BEFORE AND AFTER HYDROPHYTE CONTROL USING DIURON

Introduction

Findings have been contradictory on the relation between hydrophyte removal and the macroinvertebrate community. Declines in macroinvertebrate density following herbicide treatment have been attributed to acute herbicide toxicity (McCraren, Cope, and Eller 1969; Walker 1964, 1965; Cowell 1965); and to loss in vegetative substrate needed by plant-dwelling macroinvertebrates (Hilsenhoff 1966; Houser 1963; Walker 1964). Post-treatment increases in macroinvertebrate density have been attributed to increased availability of organic matter and the release of nutrients following hydrophyte decay (Walker 1963; Harp and Campbell 1964; Newman 1967; Tatum and Blackburn 1965). Many, perhaps most, reports indicate no substantial post-treatment change in macroinvertebrate density (Hilsenhoff 1966; Van der Weij 1967; Van der Weij, Hoogers, and Blok 1971; Blok 1967; Sills 1967; Johnson and Julin 1974; McCraren, Cope, and Eller 1969; and many others). Many, but not all, of these studies have been nonquantitative or poorly so; still, most of the literature indicates that hydrophyte control using herbicides has a negligible effect on total macroinvertebrate density if the herbicide is applied at recommended rates. Whether the effect of hydrophyte removal on invertebrates will be negative or positive depends on the species initially present (their food habits and habitat preferences), the rate and amount of oxygen depletion at decay, and the amount and selectivity of fish predation.

In April 1972, diuron was applied to a 4.58 hectare pond in Payne County, Oklahoma, in order to determine the effect on the macroinvertebrates of the loss in vegetative substrate. Diuron was applied initially at 0.07 mg/l, then reapplied as a "spot" treatment of 0.18 mg/l six weeks later. Concentrations of diuron in each application and the total of both should have been below the toxicity levels reported for macroinvertebrates (Johnson and Julin 1974). Thus, changes in the macroinvertebrate community should be attributable to the indirect herbicidal effects of habitat perturbation.

Macroinvertebrates were collected September 1971 and April 1972 before treatment and May, July, and October 1972 after treatment. Statistical comparisons were then made on macroinvertebrate numbers, biomass, and species diversity. Food habits of bluegill and dissolved oxygen were also measured to give indications of their effect on changes in the macroinvertebrate community.

Methods and Materials

Description of the study area, methods for sample site selection and stratification of samples, application of diuron, and determination of dissolved oxygen concentrations are discussed in Chapter II. Methods for Bluegill stomach analysis are described in Chapter IV.

Macroinvertebrate samples were obtained from inside a 1.22 m tall, 0.44 m diameter, 0.15 m^2 area galvanized steel cylinder. The cylinder was pushed into the lake bottom, using a 0.91 m extension for samples in the 1-3 m stratum. When in place, depth was determined, and all vegetation was removed using a spade and rake. The vegetation and attached organisms were placed in a bucket or plastic bag. Organisms attached to plants were separated from the vegetation with running water over a 35-mesh, 420-micron sieve. The enclosed water column was then swept from top to bottom, in circular motion, with a 1 mm^2 mesh aquatic net. Organisms and debris were placed in a jar of 10% formalin. Next, three 152 X 152 mm Eckman dredge samples were taken within the cylinder, the samples washed in the aquatic net, and preserved in 10% formalin. Invertebrates removed from the vegetation were pooled with those obtained from net and dredge samples and preserved in 10% formalin.

Preserved samples of macroinvertebrates were rinsed with water in a 35-mesh 420-micron sieve, then sorted using sugar flotation (Anderson 1959), counted, dried 24 hours at 150 C, cooled 24 hours in a desiccator, and weighed to the nearest 0.1 mg. Keys used for identification: Usinger (1971), Pennak (1953), and Edmondson, (1966).

Average and cumulatively pooled species diversities (Wilhm 1970) were determined for each collection date and depth strata. Species diversity was computed using the formula of Shannon: $\bar{d} = 3.322 \left[-\Sigma (n_i/n) \log_{10} (n_i/n)\right]$, where $\bar{d}$ = species diversity, n_i = number of individuals in the $\underline{i}^{th}$ taxa, and n = total number of individuals in the sample (Odum 1971).

Results and Discussion

Changes in Numbers and Biomass in the 0-1 Meter Stratum

Average total numerical density of macroinvertebrates in the 0-1 m stratum was 4009 individuals per m^3 in October 1972 and 4496 per m^3 in September 1971 (Table 2), but a *t*-test of the difference was nonsignificant (P>0.500). Also, difference in numerical density between April and October 1972 was nonsignificant (P = 0.400). The lack of change in total macroinvertebrate numerical density following treatment with diuron substantiates the findings of other investigators (McCraren et al. 1969; Blok 1967; Sills 1967; Van der Weij 1967; Van der Weij et al. 1971; Walker 1965; Johnson and Julin 1974).

Biomass density of macroinvertebrates in the 0-1 m stratum was 1.62 g/m^3 in September 1971 but 4.49 g/m^3 in October 1972 (Table 2). A *t*-test of the difference was significant (P = 0.036). This change in total biomass density between September 1971 and October 1972 was due to the change in biomass of pelecypods (Table 4). The numerical density of pelecypods was 3323% greater in October 1972 than in September 1971, but biomass density increased 131,350%. Although making up 9.6% of the total numerical density in October 1972, pelecypods comprised 46.9% of the total biomass density. Thus, a substantial part of the difference in biomass density between September 1971 and October 1972 was the shells of pelecypods. When pelecypods are deleted from the biomass calculations for the 0-1 m stratum, the difference in total biomass densities of macroinvertebrates between September 1971 and October 1972 is non-significant (P=0.268).

Although the difference in total numbers and biomass of macroinvertebrates in the 0-1 m stratum between 1971 and 1972 were not large, changes in numbers and biomass of certain taxonomic groups are quite evident (Tables 3 and 4). In general, density of detritophagic forms

Table 2. Numerical and biomass density (g/m^2 and g/m^3) of macroinvertebrates for depth strata and dates indicated.

Collection Date	Depth Strata	Sample Size	Numbers per m^3	Numbers per m^2	Biomass g/m^3	Biomass g/m^2
1971						
Sept.	(0-1 m)	6	4009 ± 1577[1]	2496 ± 1153	1.62 ± 1.51	1.09 ± 1.13
Sept.	(1-3 m)	10	1264 ± 440	2135 ± 686	1.32 ± 0.52	2.33 ± 1.00
1972						
April	(0-1 m)	6	4864 ± 3842	2579 ± 2282	7.11 ± 6.46	3.48 ± 3.20
May	(0-1 m)	7	5410 ± 2201	2897 ± 1302	5.70 ± 4.46	3.48 ± 2.91
July	(0-1 m)	7	4387 ± 1662	2447 ± 861	3.84 ± 1.93	2.22 ± 1.10
Oct.	(0.1 m)	6	4496 ± 1356	2090 ± 644	4.49 ± 2.60	1.94 ± 0.80
Oct.	(1-3 m)	12	771 ± 228	1010 ± 190	1.02 ± 0.55	1.45 ± 0.75

[1]The 95 percent confidence limits are given with each mean.

Table 3. Average numerical density (per m^3) by depth strata for dates shown.

Taxonomic group	September 1971 1-3 m No.	%	September 1971 0-1 m No.	%	1972 (0-1 m) April No.	%	May No.	%	July No.	%	October 1972 0-1 m No.	%	October 1972 1-3 m No.	%
Amphipoda	128	10.1	1081	27.0	775	15.9	1214	22.4	413	9.4	5	0.1	0	0.0
Hydracarina	5	0.4	119	3.0	18	0.4	57	1.1	13	0.3	2	0.0	0	0.0
Oligochaeta	127	10.0	554	13.8	305	6.3	273	5.0	294	6.7	515	11.5	66	8.6
Gastropoda	22	1.7	65	1.6	14	0.3	9	0.2	13	0.3	2	0.0	1	0.1
Pelecypoda	103	8.1	13	0.3	219	4.5	480	8.9	496	11.3	432	9.6	67	8.7
Insecta:														
Coleoptera	44	3.4	230	5.7	211	4.3	332	6.1	101	2.3	0	0.0	0	0.0
Trichoptera	76	6.0	105	2.6	76	1.6	11	0.2	127	2.9	24	0.5	1	0.1
Odonata:														
Anisoptera	34	2.7	215	5.3	56	1.2	49	0.9	45	1.0	32	0.7	3	0.4
Zygoptera	195	15.4	750	18.7	351	7.2	241	4.5	46	1.0	7	0.2	3	0.4
Ephemeroptera:														
Caenis	21	1.7	253	6.3	150	3.1	142	2.6	205	4.7	75	1.7	16	2.1
Hexagenia	0	0.0	0	0.0	6	0.1	27	0.5	23	0.5	32	0.7	1	0.1
Diptera:														
Chironomidae	301	23.8	404	10.1	2377	48.8	2323	43.0	2458	56.0	3055	58.0	524	68.0
Chaoborus	118	9.3	16	0.4	0	0.0	0	0.0	0	0.0	0	0.0	7	0.9
Chrysops	32	2.5	39	1.0	40	0.8	38	0.7	20	0.5	0	0.0	1	0.0
Palpomyia	24	1.9	8	0.2	154	3.2	158	2.9	75	1.7	297	6.6	77	10.0
Other Diptera	23	1.8	117	2.9	51	1.0	23	0.4	24	0.5	5	0.1	0	0.0
Others	11	0.8	40	0.9	62	1.3	33	0.6	34	0.8	13	0.3	5	0.6
Totals	1264	100.0	4009	100.0	4864	100.0	5410	100.0	4387	100.0	4496	100.0	771	100.0

[1] Depth stratum is 1-3 meters.

[2] Depth stratum is 0-1 meters.

Table 4. Average biomass density (g x $10^4/m^3$) by depth strata for dates shown.

Taxonomic group	September 1971				1972 (0-1 m)						October 1972			
	1-3 m		0-1 m		April		May		July		0-1 m		1-3 m	
	Bmass	%	Bmass	%	Bmass	%	Bmass	%	Bmass	%	Bmass	%	Bmass	%
Amphipoda	164	1.3	1009	6.2	2071	2.9	1308	2.3	494	1.3	7	0.0	0	0.0
Hydracarina	3	0.0	85	0.5	37	0.1	45	0.1	15	0.0	1	0.0	0	0.0
Oigochaeta	6866	52.0	1599	9.9	22888	32.2	13463	23.6	16355	42.6	14922	33.0	4020	39.5
Gastropoda	786	6.0	2589	16.0	6824	9.6	1258	2.2	2722	7.1	1	0.0	0	0.0
Pelecypoda	2127	16.1	16	0.1	22734	31.9	23530	41.3	5040	13.1	21016	46.9	2785	27.4
Insecta:														
Coleoptera	718	5.4	4202	26.0	5685	8.0	3293	5.8	1092	2.8	0	0.0	0	0.0
Trichoptera	148	1.1	86	0.5	304	0.4	32	0.1	319	0.8	62	0.1	1	0.0
Odonata:														
Anisoptera	389	2.9	3254	20.1	1514	2.1	2023	3.5	1683	4.4	1181	2.6	239	2.3
Zygoptera	800	6.1	1372	8.5	1536	2.2	796	1.4	378	1.0	30	0.1	9	0.1
Ephemeroptera:														
Caenis	32	0.2	198	1.2	701	1.0	692	1.2	561	1.5	70	0.2	24	0.2
Hexagenia	0	0.0	0	0.0	972	1.4	5487	9.6	5265	13.7	2762	6.2	40	0.4
Diptera:														
Chironomidae	456	3.5	299	1.8	3148	4.4	3413	6.0	3592	9.4	4172	9.3	2631	25.9
Chaoborus	127	1.0	13	0.1	0	0.0	0	0.0	0	0.0	0	0.0	12	0.1
Chrysops	381	2.9	797	4.9	471	0.7	644	1.1	470	1.2	0	0.0	8	0.1
Palpomyia	21	0.2	6	0.0	167	0.2	197	0.3	105	0.3	376	0.8	147	1.5
Other Diptera	44	0.3	128	0.8	155	0.2	111	0.2	178	0.4	2	0.0	0	0.0
Others	130	1.0	559	3.4	1938	2.7	724	1.3	171	0.4	255	0.5	255	2.5
Totals	13192	100.0	16212	100.0	71145	100.0	57018	100.0	38352	100.0	44857	100.0	10171	100.0

[1]Biomass multiplied by 10^4.

[2]Depth stratum is 1-3 meters.

[3]Depth stratum is 0-1 meters.

such as oligochaetes, most chironomids and pelecypods was greater after hydrophyte removal. Organisms associated with hydrophytes (phytomacrofauna) such as Hyallela azteca (Amphipoda), Caenis sp. (Ephemeroptera), Ochrotrichia sp. (Trichoptera), anisopterans, zygopterans, coleoterans and gastropods, diminished after hydrophyte removal.

Other workers have also observed that the density of oligochaetes and chironomids increased following organic enrichment (Harp and Campbell 1964; Walker 1963; Tatum and Blackburn 1965; Newman 1967; Howell 1942; McIntire and Bond 1962; Collins 1970; Wahlquist 1970; and others). Oligochaetes and most chironomids injest detritus; so both groups might be expected to benefit from hydrophyte decay (Pennak 1953; Fast 1971; Usinger 1971; Brinkhurst and Cook 1974; Roback 1974; Monakov 1972). Numerical and biomass density of chironomids increased in the 0-1 m stratum from $404/m^3$ and $0.03\ g/m^3$ in September 1971 to $3055/m^3$ and $0.42\ g/m^3$ in October 1972 (Tables 3 and 4). However, numerical density of oligochaetes declined in the 0-1 m stratum from $554/m^3$ in September 1971 to $515/m^3$ in October 1972 (Table 3); while biomass density rose from ($0.16\ g/m^3$ to $1.49\ g/m^3$ (Table 4). A decline in numerical and increase in biomass densities of oligochaetes was also found before treatment between September 1971 and April 1972 (Tables 3 and 4), which may indicate that as oligochaete numbers were reduced growth per individual in terms of biomass improved.

Numerical and Biomass densities of oligiochaetes declined from $305/m^3$ and $2.29\ g/m^3$ in April prior to treatment to $273/m^3$ and $1.35\ g/m^3$ one month later in May (Tables 3 and 4). This decline could be due to acute toxicity or suffocation from anoxia following hydrophyte decomposition. Since diuron is not very soluble in water, high concentrations may occur in the hydrosol (Johnson and Julin 1974); the low DO near the bottom of Sanborn Lake has already been discussed (Chapter II). Numerical density of oligochaetes increased between May and October 1972, but did not reach the level of the previous fall (Table 3). Biomass density in October 1972 surpassed the density in 1971 (Table 4). Thus, diuron toxicity or anoxia may have reduced the numerical density of oligochaetes following hydrophyte removal; but this reduction in numerical density, coupled with an increase organic matter from decaying hydrophytes, may have lessened intraspecific competition such that biomass density was enhanced.

Pelecypods (Family Sphaeridae) increased in the 0-1 m stratum from $13/m^3$ and $1.6\ mg/m^3$ in September 1971 to $432/m^3$ and $2.1\ g/m^3$ in October 1972 (Tables 3 and 4). Sphaerids are detritophagic, and often are found in organic or septic conditions (Pennak 1953; Fuller 1974; Monakov 1972), Thus, the increase in numerical and biomass densities of Sphaerids was probably due to an increase in food from decaying hydrophytes.

Densities of the burrowing mayfly, Hexagenia, were greater after herbicide treatment. This species may scavenge on detrital material (Craven 1968) which would be provided by decomposition of hydrophytes.

The biting midge larva, Palpomyia, increased after herbicide treatment. It has been reported to be a predator on chironomids (Robak 1974; Usinger 1971) which would account for its increase in abundance after herbicide treatment, since chironomids increased in Sanborn after treatment.

Several authors have observed a reduction in phytomacrofauna such as Hyallela azteca (Amphipoda), aninsopterans, zygopterans, coleopterans, and gastropods following removal of hydrophytic habitat (Hilsenhoff 1966;

Walker 1963, 1964; May Hestand, and Van Dyke 1973; Wahlquist 1970). Numerical density of these organisms also decreased in the 0-1 m stratum of Sanborn Lake: Amphipods from $1081/m^3$ in 1971 to $5/m^3$ in October 1972; odonates from $965/m^3$ to $39/m^3$; coleopterans from $230/m^3$ to 0; etc. (Table 3). Post-treatment declines in numerical densities of _Caenis_ (Ephemeroptera), _Ochrotrichia_ (Trichoptera) and hydrachnids were apparently related to habitat removal, since they are considered phytomacrofauna (Gerking 1957). Hydrachnids, however, were also a greatly preferred food item of the bluegill; so predation may have contributed to their decline (Chapter IV). Numerical densities of the principal trichopteran, _Cyrnellus_, and the predaceous dipterans _Chrysops_ and _Chaoborus_ also declined.

Changes in Numbers and Biomass in the 1-3 Meter Statum

The changes observed in numerical and biomass density of macroinvertebrates in the 1-3 m stratum after hydrophyte removal, were opposite that of the 0-1 m stratum. The decrease in total average numerical density between September 1971 and October 1972 was significant (P=0.035) (Table 2), but the decrease in biomass density was nonsignificant (P=0.350) even when pelecypods are deleted from the calculations (P=0.280). Explanation for this decrease could be partly due to the elemination of hydrophytic habitat preferred by phytomacrofauna, or to suffocation from anoxic conditions following hydrophyte decay.

Numerical and biomass density of phytomacrofauna (_Caenis_, hydrachnids, _Hyallela azteca_, anisopterans, zygopterans, coleopterans, and gastropods) decreased in the 0-1 m and 1-3 m stratum between September 1971 and October 1972 (Tables 3 and 4). Numerical and biomass density of Trichoptera and the dipterans *Chrysops* and *Chaoborus*, also decreased in both strata. Densities of hydrophytes and phytomacrofauna were greater in the 0-1 m stratum relative to the 1-3 m stratum. This would suggest that the likelihood of changes in macroinvertebrate density would be greater in the 0-1 m stratum than in the 1-3 m stratum after hydrophyte removal, and this was generally the findings. Numerical densities of Trichoptera and *Chaoborus* diminished more in the 1-3 m stratum, and density of *Chrysops* declined about the same amount in both 0-1 and 1-3 m strata.

As previously mentioned, detritophagic organisms apparently benefited from decaying hydrophytes. Post-treatment numerical density of detritophagic pelecypods increased in the 0-1 m stratum but decreased in the 1-3 m stratum (Tables 3 and 4). Chironomid, *Palpomyia*, and *Hexagenia* post-treatment densities were greater than pre-treatment densities in the 1-3 m stratum, but the increase was not as great as in the 0-1 m stratum. This finding coupled with the fact that Trichoptera, Oligochaeta, and *Chaoborus* densities diminished more in the 1-3 m stratum than in the 0-1 m stratum, indicates that some factor besides habitat loss affected 1-3 m macroinvertebrate densities.

Depletion of dissolved oxygen following diuron treatment and hydrophyte decay has been reported by many investigators (Johnson and Julin 1974). Oxygen depletion during summer stagnation may reduce macroinvertebrate standing crops (Bergersen 1969; Fast 1971; Inland Fisheries Branch 1970). Between 20 May and 14 June 1972 only the upper 2-3 meters of the water column in Sanborn Lake contained more than 3 mg/l dissolved oxygen, hence, it is quite likely that macroinvertebrates below these depths were adversely affected (Chapter II).

Dissolved oxygen in the lower 0.5 m of the water column was between 0.5 and 1.5 mg/l most of this period.

Most species of aquatic oligochaetes are able to withstand prolonged periods of low dissolved oxygen (Pennak 1953; Brinkhurst 1965); but their numerical density has been shown to increase when an anoxic hypolimnion of a eutrophic lake is aereated (Fast 1971; Inland Fisheries Branch 1970). Possibly, oligochaetes are able to survive, but not optimally utilize the bottom organic matter until anoxia or other toxic conditions are reduced. Numerical and biomass densities of oligochaetes in the 1-3 m stratum of Sanborn Lake diminished from $127/m^3$ and 0.69 g/m^3 in September 1971 to $66/m^3$ and 0.40 g/m^3 in October 1972 (Tables 3 and 4).

Fuller (1974) states that most species of <u>Sphaerium</u>, <u>Musculium</u>, and <u>Pisidium</u> are tolerant to nearly septic conditions. Morrison (1932) found the lowest pH tolerated by these genera was 6.8, 5.9, and 5.1, respectively. Perhaps, if oxygen was not limiting in Sanborn Lake, pH in the stratified areas over decaying hydrophytes may have been. Numerical density of pelecypods in the 1-3 m stratum of Sanborn Lake decreased from $103/m^3$ in September 1971 to $67/m^3$ in October 1972 (Table 3); biomass density increased slightly from 0.21 g/m^3 to 0.28 g/m^3 (Table 4).

Numerical densities of chironomids, <u>Palpomyia</u> and <u>Hexagenia</u> were enhanced 756%, 3713% and 3200% in the 0-1 m stratum between September 1971 and October 1972; but increased only 174%, 321%, and 100% in the 1-3 m stratum. The reason for the smaller increase in the 1-3 m stratum may be attributable to a smaller input of organic matter from decaying hydrophytes in the 1-3 m stratum as compared to the 0-1 m stratum.

Chironomids and *Hexagenia* would thus have had less of a detrital food source in the 1-3 m area. The success of the predaceous *Palpomyia* would be dependent on changes in chironomid density. Another possibility is that hypolimnial DO depletion in the 1-3 m stratum kept the above organisms from using their food sources as efficiently as in the 0-1 m stratum.

Many chironomids have been found to withstand DO concentrations near 1 mg/l, but it is generally agreed that they cannot withstand prolonged low levels (Curry 1965; Oliver 1971; Toetz, Wilhm and Summerfelt 1972). Others have noted that chironomid growth may be limited when DO is low (Jónasson and Kristiansen 1967; Fast 1971). *Hexagenia* have been shown to succumb when DO falls below 1.4 mg/l (Nebecker 1972; Britt 1955). *Palomyia* has been collected from waters with DO concentrations between 2-14 mg/l.

Numerical density of Trichoptera and *Chaoborus* decreased 99% and 94% in the 1-3 m stratum between September 1971 and October 1972; 77% and 100% in the 0-1 m stratum. Trichopterans were probably affected by low DO concentrations in the 1-3 m stratum of Sanborn Lake. Roback (1965), in studying the depth distribution of *Athripsodes*, found only one individual below 5 mg/l DO. *Cyrnellus* and *Athripsodes* are found in habitats where DO ranges 5-12 mg/l (Roback 1974).

Chaoborus is predatory on copepods, cladocerans, and chironomids (Stahl 1966). Post-treatment chaoborid densities would be expected to rise in Sanborn Lake since chironomids and zooplankton increased following hydrophyte decay. However, numerical density of chaoborids in the 1-3 m stratum declined from $118/m^3$ in September 1971 to $7/m^3$ in October 1972 (Table 3). The observed decline in chaoborids may be a result of their avoidance of anoxic areas and resultant migration to the epilimnion where exposure to fish predation reduced their density (Fast 1971). *Chaoborus*, however, was not found in bluegill stomachs either before or after hydrophyte removal (Chapter IV).

If anoxia were adversely affecting the macroinvertebrates in the 1-3 m stratum, it is possible that the decreases in the several taxa previously mentioned were due to migration into the epilimnion (Fast 1971; Bay et al. 1966; Hilsenhoff 1966). It is not known if this occurred in Sanborn Lake, but it is not suspected because numerical density of taxa that declined in the 1-3 m stratum also declined in the 0-1 m stratum.

Finally, diuron may have been directly toxic to the macroinvertebrates of Sanborn Lake. Concentrations of diuron may have been higher in the sediments (Johnson and Julin 1974), or degradation of the chemical under anoxic conditions may have been slower, thus allowing maximal concentrations to remain in contact with benthic macrofauna. This is, however, purely speculative.

Species Diversity

Changes in community structure are usually better indicators of environmental change than absolute changes in numerical density or number of species present (Rainwater 1969). In general, changes in the diversity value, $\bar{d}$, are a valid indicator of changes in community structure (Wilhm and Dorris 1966). Wilhm (1970) recommended the pooled $\bar{d}$ over the average $\bar{d}$, since the pooled $\bar{d}$, after reaching an assymptote, is less variable. Stable complex environments are characterized by high diversity values; unstable, more homogeneous environments by low diversity values (Odum 1971).

The differences in mean diversity values of the Sanborn Lake macroinvertebrate community between September 1971 and October 1972 (Table 5) were nonsignificant ($P>0.500$). Pooled $\bar{d}$ values in the 0-1 and 1-3 m strata diminished from 4.03 and 4.59 in September 1971 to 3.76 and 3.70 in October 1972 (Table 5); but all were characteristic of more stable, clean water conditions (Wilhm 1970). The reduction in macroinvertebrate community diversity following hydrophyte control is similar to that reported by Crossland and Elgar (1975). The reduction in diversity of the Sanborn Lake macroinvertebrate community following hydrophyte removal is probably due to a reduction in the numerical densities of several species of phytomacrofauna, coupled with an increase in the density of a few detritophagic species that were tolerant to low DO concentrations.

Correlations with Hydrophyte Biomass and Depth

Numerical density of macroinvertebrates in the 1-3 m stratum was significantly lower ($P<.001$) than in the 0-1 m stratum both before and after herbicide treatment (Table 2). The following multiple regressions were calculated for 1971 samples, giving correlation coefficients of -0.82, + 0.86, and -0.89, respectively:

$$Y = a_1 + B_1X_1 \qquad (1)$$

$$Y = a_2 + B_2X_2 \qquad (2)$$

$$Y = a_3 + B_1X_1 + B_2X_2 \qquad (3)$$

where Y = numerical density of macroinvertebrates, a = the Y-intercept, B = the slope of the regression, X_1 = depth of the water column, and X_2 = biomass of *Potamogeton*. Since regression 1 and 3 are both negative, it

Table 5. Numbers of species (s), numbers of individuals (n), successively pooled and mean diversity values for invertebrate samples taken at times and depth strata indicated.

Date	s	n	$\bar{d}$, pooled 1	2	3	4	5	6	7	8	9	10	11	12	Final pooled $\bar{d}$	Average
Sept. 71[1]	72	2246	3.22	3.16	3.56	3.72	4.01	4.03	-	-	-	-	-	-	4.03	3.35±.52
Sept. 71[2]	57	3203	4.27	4.22	4.35	4.45	4.38	4.45	4.45	4.51	4.57	4.59	-	-	4.59	3.83±.34
Apr. 72[1]	61	2321	4.25	4.30	4.43	4.40	4.43	4.44	-	-	-	-	-	-	4.44	3.84±.61
May 72[1]	59	3042	3.42	3.59	3.90	4.07	4.10	4.18	4.28	-	-	-	-	-	4.28	3.50±.41
July 72[1]	68	2569	3.89	4.34	4.53	4.56	4.59	4.56	4.49	-	-	-	-	-	4.49	3.76±.12
Oct. 72[1]	37	1881	3.46	3.72	3.73	3.76	3.76	3.76	-	-	-	-	-	-	3.76	3.45±.11
Oct. 72[2]	33	1818	3.73	3.62	3.84	3.78	3.72	3.76	3.69	3.69	3.71	3.71	3.68	3.70	3.70	3.13±.22

[1]Depth stratum is 0-1 meters.

[2]Depth stratum is 1-3 meters.

[3]The 95% confidence interval is given with each mean.

is postulated that depth of the water column is a more important overall determinant of macroinvertebrate numbers than is the biomass of Potamogeton. Vegetative biomass was negatively correlated with depth of the sample thus, as depth increased, vegetative biomass and invertebrate numbers decreased. Water depth and light penetration may be the overall controlling factors for the depth distribution of hydrophytes which in turn was the controlling factor for abundance of macroinvertebrates.

The correlation between numerical density of macroinvertebrates and depth is well supported by similar findings in other literature (Muttkowski 1918; Gerking 1962; Wilhm 1974; Fast 1971). The correlation between numerical density of macroinvertebrates and vegetative biomass is supported by the fact that there are often more macroinvertebrates (principally phytomacrofauna) in vegetated areas than in barren areas (Gerking 1962; Wahlquist 1970; Needham 1930). Burgess (1966) found 25-238 times more invertebrates in water hyacinths of Lake Apopka, Florida, than in the bottom mud; similar results were observed for density of invertebrates in water lettuce in Lake Panasoffkee (Burgess 1966). In Pena Blanca Lake, Arizona, however, the standing crop of littoral invertebrates was apparently not related to the standing crop of hydrophytes (Bergersen 1969).

Numerical density of the macroinvertebrates of Sanborn Lake was negatively correlated with depth and hydrophyte biomass in 1972 ($r = -0.91$). This correlation can be attributed to depth alone, since the biomass of hydrophytes in October 1972 was essentially zero. If this correlation indicates a relation between macroinvertebrate density and depth alone in 1972, then the correlation between macroinvertebrate density and hydrophyte biomass in 1971 may be due to the interrelationship between hydrophyte biomass and depth discussed in Chapter II.

Another possible reason for depth being the only factor controlling invertebrate density in 1972 is that the reduction of phytomacrofauna following hydrophyte removal left mainly benthic organisms which logically might be influenced more by depth than the phytomacrofauna. Gerking (1962) found that the numerical densities of benthos and phytomacrofauna in Potamogeton beds were equal. Thus, in 1971, hydrophyte biomass and depth could both have had a direct effect on numerical density of macroinvertebrates: Potamogeton biomass influencing phytomacrofauna, and depth influencing the benthos. In 1972, the macroinvertebrate community was primarily benthic, so depth may have had the only influence on the total fauna.

The quality of the bottom substrate and the amount of substrate organic matter may be more important controlling factors than were depth or hydrophyte biomass. Burgess (1965) found that hydrophyte fragments in various stages of decay supported one-third more organisms than did the living plants. Although living hydrophytes were removed in 1972, detritus was still present. Since living hydrophyte biomass was negatively correlated with depth in Sanborn Lake, it is logical that the depth distribution of detritus would parallel that of living plants. If the majority of Sanborn Lake's invertebrate community was of the type supported principally by litter, their distrubution may not have been substantially affected by the herbicide treatment.

Multiple regressions of the relationship between the biomass density of macroinvertebrates and depth and hydrophyte biomass were nonsignificant in 1971 and 1972 ($P>0.500$). This is presumedly due to the balancing effect of large numbers of smaller species in the shallow water versus fewer numbers of larger species in the deeper water.

A multiple regression for the relationship between the diversity of the macroinvertebrate community and depth and hydrophyte biomass was nonsignificant at the 0.05 level in 1971. The partial sum of squares for hydrophyte biomass was much smaller than that for depth, which indicates depth influenced diversity more than hydrophyte biomass. Species diversity in 1971 and 1972 did fit a negative quadratic correlation with depth ($r = -0.64$, $P = 0.055$, $r = -0.61$, $P = 0.03$). Species diversity is usually higher in the shallow depths of lakes (Wilhm 1974).

Conclusions

Most of the changes in the macroinvertebrate community of Sanborn Lake can be explained on the basis of environmental perturbations. In general, detritophagic organisms such as chironomids, pelecypods, and oligochaetes, benefitted from an increased supply of detritus from decaying hydrophytes. *Hyallela*, *Caenis*, *Ochrotrichia*, Hydracarina, Anisoptera, Zygoptera, Coleoptera, and Gastropoda declined following hydrophyte removal and resultant habitat loss. Species diversity decreased with the decline in importance of several species of phytomacrofauna and the greater importance of a few detritophages.

Total numbers of biomass and macroinvertebrates in the 0-1 m stratum did not change significantly. The decline of phytomacrofauna appears to have balanced the increase of detritophages. Numbers and biomass of macroinvertebrates in the 1-3 m stratum declined, probably due to oxygen depletion.

Diuron should not have been toxic to macroinvertebrates at concentrations used. Nevertheless, the decline of oligochaetes, *Chaoborus*, *Chrysops*, and possibly others, is not easily attributable to environmental changes.

CHAPTER IV

FOOD HABITS OF BLUEGILL (*Lepomis macrochirus*) IN AN OKLAHOMA POND BEFORE AND AFTER HYDROPHYTE REMOVAL USING DIURON

Introduction

Food habits studies can be used to assess the impact of hydrophyte removal on growth of fish by relating observed changes in the invertebrate community to utilization of this food source by fish. Several studies on hydrophyte removal have considered the effects on the invertebrate community (Harp and Campbell 1964; Walker 1963, 1964, 1965; McCraren, Cope and Eller 1969; and others); several have studied the effects on fish growth (Houser 1963; Cope, Wood, and Wallen 1970; McCraren et al. 1969; and others); but apparently there are no reports correlating the change in the invertebrate community with changes in fish growth. It is usually valid to assume that the growth response of fish to environmental perturbation is related to the response of the invertebrate community. Food habits studies can be useful in determining if, in fact, fish growth is related to changes in invertebrate density. For example, a food habits analysis can reveal which invertebrate taxa were the most utilized, and therefore most likely responsible for changes in fish growth; it can indicate to what degree fish took advantage of changes in abundance of food organisms in their environment; it can reveal that changes in fish growth were not related to changes in the invertebrate community if it is found that invertebrates were not eaten.

In April 1972, diuron was applied to a 4.58 hectare pond in Payne County, Oklahoma, to remove hydrophytes. Measurements were made on the macroinvertebrate community and bluegill growth in 1971 and 1972. The effect that changes in the macroinvertebrates had on growth of bluegill was assessed through food habits analysis before and after hydrophyte removal.

Methods

Stomachs from fish collected September 1970 and 1971, and October 1972, were extracted and placed in 70 percent isopropanol. Several fish were randomly selected for analysis from each age group (1-4). Numeric and gravimetric analyses were made.

Stomach and stomach content weight were obtained for the 1970 sample, but contents were not taxonomically identified. Wet weights were obtained by blotting with a paper towel and weighing to the nearest 0.1 mg. Wet weights for 1970 were converted to dry using the wet/dry ratio obtained from 1971 and 1972 samples.

In 1971 and 1972, stomach contents were washed into a petri dish for identification and numerical analyses using a 40x dissecting scope. The contents were separated into macroinvertebrate, zooplankton, and plant fractions. Each fraction was filtered, dried 24 hours at 105 C, and allowed to cool 24 hours in a desiccator. They were weighed to the nearest 0.1 mg.

Frequency of occurrence and relative abundance of various food items were compiled for each age class. Pooled values of Shannon's $\bar{d}$

were used to calculate diversity (Wilhm 1970). Ivlev's Electivity Index was used as a measure of selectivity (Ivlev 1961). Methods for determining the standing crops of macroinvertebrates and growth of bluegill are given in Chapters III and V, respectively.

Results and Discussion

Gravimetric Analyses

For all age groups, total weight of the stomach contents and the ratio, gut content/body weight, were higher in 1972 than in 1971 (Table 6). Differences were significant for all age groups ($P<0.005$) except age 4 fish ($P=0.180$) and the ratio content/body weight, of age 2 fish ($P>0.500$). Comparison of single grab samples from two years cannot represent completely the entire before-after time periods. Nevertheless, the average biomass of food in the stomachs of bluegill after hydrophyte removal was larger than before the treatment.

Total biomass of macroinvertebrates, zooplankton, and plant matter increased in the diet of bluegill in 1972 (Table 6), probably because these three dietary components were more numerous or vulnerable in the environment following hydrophyte removal. Density of macroinvertebrates in the littoral areas of Sanborn Lake after hydrophyte removal was greater than before treatment (Chapter III); and macroinvertebrates were probably more vulnerable to predation since the hydrophytic cover was eliminated. Zooplankton density was not measured in Sanborn Lake, but could be expected to rise with an increase of phytoplankton following hydrophyte decay (Walsh, Miller and Heitmuller 1971; Lawrence 1965; Hasler and Jones 1949; Wahlquist 1970). Plant fragments were presumably

Table 6. Gravimetric analysis of bluegill stomach contents, 1971 and 1972.

Year	Age	N	Gut content/ body wt. (% by wt.)	Invertebrates wt.	Invertebrates %	Zooplankton wt.	Zooplankton %	Plant wt.	Plant %	Total wt.	Total %
1971	1	10	.021[2] ± .015	Tr.[3]	26.3[4]	Tr.	73.7	Tr.	0.0	Tr.	100.0
1972	1	5	.079 ± .056	.00050 ± .00055	51.0	.00036 ± .00100	36.7	.00012 ± .00028	12.3	.00098 ± .00118	100.0
1971	2	10	.028 ± .019	.00059 ± .00064	61.5	.00037 ± .00023	38.5	Tr.	0.0	.00096 ± .00064	100.0
1972	2	5	.037 ± .059	.00196 ± .00341	69.0	.00066 ± .00079	23.2	.00022 ± 00039	7.8	.00284 ± .00406	100.0
1971	3	10	.015 ± .008	.00094 ± .00088	49.5	.00045 ± .00032	23.7	.00051 ± 00064	26.8	.00190 ± .00100	100.0
1972	3	5	.052 ± .031	.00248 ± .00248	33.3	.00120 ± .00205	16.1	.00376 ± .00464	50.6	.00744 ± .00539	100.0
1971	4	2	.003 ± .032	.00045 ± .00568	100.0	Tr.	0.0	Tr.	0.0	.00045 ± .00568	100.0
1972	4	5	.086 ± .088	.00380 ± .00489	17.7	.00402 ± .00695	18.8	.01294 ± .01524	60.4	.02142 ± .02177	100.0

[1]Average weights of content fractions and gut content wt. per body wt. percentages are given with their 95% confidence intervals.

[2]Gut contents (dry wt. in g) were compared to wet weight (g) of body weight in all age classes except age 1 (1971), in which estimated dry weights (based on the average wet weight-dry relationship) were used; the actual dry weights were below the sensitivity of the balance used.

[3]Tr. = trace = <0.0001.

[4]In all age classes except age 1 (1971) the average dry weights were used to obtain percentage of each fraction to the total gut content. Age 1 (1971) used wet weights since dry weights were below the sensitivity of the balance.

more numerous in 1972 after hydrophytes began to decay; filamentous algae were attached to decaying hydrophytes in 1972, but not substantially to living plants in 1971.

Zooplankton comprise a smaller and macroinvertebrates a greater percentage of the total biomass of the stomach contents as bluegill grow larger (Gerking 1962; Leonard 1940; Howell 1942; Patriarche and Ball 1949, Emig 1966). Zooplankton comprised 73.7%, 38.5%, 23.7%, and 0.0% of the stomach contents of age 1, 2, 3, and 4 bluegill in 1971; and 36.7%, 23.2%, 16.1%, and 18.8% in 1972 (Table 6). The percentage of macroinvertebrates in the diet does not show a clear trend with age (Table 6).

The relative importance of invertebrates in the stomach contents of age 1 and 2 bluegill rose from 26.3% and 61.5% in 1971 to 51.0% and 69.0% in 1972; the importance of zooplankton for ages 1 and 2 dropped from 73.7% and 38.5% to 36.7% and 23.2% (Table 6). The diet of age 1 and 2 bluegill in 1972 shifted toward invertebrates. Patriarche and Ball (1949) found the diet of bluegill in four Michigan ponds shifted from entomostraca to chironomids at about 51 mm length, and age 1 bluegill in Sanborn Lake were closer to this size in 1972 than in 1971. The age 1 fish of Sanborn Lake were 39 mm in total length in 1971 and 47 mm in 1972; age 2 fish were 68 mm in 1971, and 73 mm in 1972 (Chapter V). It is postulated that bluegill less than 51 mm length which feed almost exclusively on zooplankton had a greater source of food in 1972 after hydrophyte removal, and thus were able to more quickly grow to a size at which they could utilize chironomids.

There was a greater biomass and percentage of plant matter in the stomach contents of bluegill in 1972 than in 1971 (Table 6). Emig (1966) states that plants are frequently eaten and are sometimes the dominant food, particularly in the summer. Patriarche and Ball (1949) speculated that plants would be used as food only when animal food is unavailable. Gerking (1962) believed plants have little nutritive value but may be used as "roughage" to scour out chitinous debris. It is also quite possible that plant matter is inadvertently consumed in capturing organisms inextricably bound in bottom debris or algal mats. I have observed such with darters in aquaria. If so, one would expect plant matter to be greater in the diet of larger fish, since it has already been demonstrated that they consume more macroinvertebrates while smaller fish prefer zooplankton.

Numeric Analysis

There were higher total numbers of food items in the stomachs of bluegill in 1972 than in 1971 (Table 7). Chironomids, ceratopogonids, pelycepods, and plant matter increased in relative abundance and frequency of occurrence (Tables 7 and 8). Increase of these items in bluegill stomachs would likely follow observed increase of these items in the environment in 1972 (Chapter III) if consumption was mainly due to their abundance. Trichopterans, ephemeropterans, hydrachnids, amphipods, and gastropods all declined in abundance in the environment in 1972 (Chapter III), but their abundance and frequency of occurrence in the bluegill ration was greater in 1972 (Tables 7 and 8). Apparently, loss of habitat left these organisms more vulnerable to bluegill predation.

It cannot be determined from this study if bluegill predation or habitat loss per se is the reason for the decline of trichopterans, ephemeropterans, hydrachnids, amphipods and gastropoda in the

Table 7. Relative abundance of food items in four age classes of bluegill, 1971 and 1972.

Food Item	Age 1		Age 2		Age 3		Age 4	
	1971	1972	1971	1972	1971	1972	1971	1972
Crustacea:								
Amphipoda	0.0	0.2	0.0	0.0	0.1	0.2	0.0	0.0
Cladocera	15.3	653.8	11.8	467.0	54.3	1602.2	39.5	790.4
Cladocera eggs	0.3	3.2	2.4	32.0	4.5	30.0	3.5	111.0
Copepoda	0.7	8.6	2.0	3.0	1.9	1.0	2.5	1.2
Ostracoda	1.5	4.8	10.0	2.8	7.6	3.6	0.0	1.0
Insecta:								
Coleoptera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4
Trichoptera	0.0	0.2	0.9	0.2	0.3	0.4	0.0	0.8
Ephemeroptera	0.1	0.2	0.2	0.2	0.2	0.4	0.0	0.0
Odonata	0.1	0.0	0.7	0.2	0.1	0.0	0.0	0.0
Diptera:								
Chironomidae	1.4	1.6	1.9	7.6	10.0	23.0	1.5	21.0
Ceratopogonidae	0.0	0.0	0.2	0.6	0.0	0.2	0.0	0.0
Terrestrial insects	0.0	0.2	0.0	0.6	0.2	0.2	0.0	0.2
Miscellaneous:								
Hydracarina	0.2	0.6	0.6	1.0	1.7	1.0	0.0	0.4
Pelecypoda	0.0	0.0	0.0	0.6	0.0	0.0	0.0	2.0
Gastropoda	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.2
Oligochaeta	0.0	0.0	0.0	0.2	0.0	0.2	0.0	0.2
Plant sprouts	0.0	0.0	0.4	1.8	5.9	N.C.	N.C.	N.C.
Fish scales	0.1	0.6	0.3	0.2	0.6	0.0	0.0	0.4
Total	19.7	674.0	31.4	518.0	87.5	1662.4	47.0	929.0
Sample size	10	5	10	5	10	5	2	5

[1]Abundance is given as average number of that item in stomachs of that age class.

[2]N.C. = not counted.

Table 8. Frequency of occurrence of food items in four age classes of bluegill, 1971 and 1972.

Food Item	Age 1		Age 2		Age 3		Age 4	
	1971	1972	1971	1972	1971	1972	1971	1972
Crustacea:								
Amphipoda	0.0	20.0	0.0	0.0	10.0	20.0	0.0	0.0
Cladocera	90.0	100.0	100.0	100.0	90.0	90.0	100.0	80.0
Cladocera eggs	20.0	30.0	40.0	100.0	60.0	40.0	100.0	60.0
Copepoda	50.0	100.0	70.0	80.0	60.0	60.0	100.0	40.0
Ostracoda	80.0	30.0	90.0	60.0	80.0	40.0	0.0	20.0
Insecta:								
Coleoptera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	20.0
Trichoptera	0.0	20.0	60.0	20.0	30.0	40.0	0.0	40.0
Ephemeroptera	10.0	20.0	20.0	20.0	10.0	40.0	0.0	0.0
Odonata	10.0	0.0	40.0	20.0	10.0	0.0	0.0	0.0
Diptera:								
Chironomidae	40.0	60.0	50.0	80.0	80.0	100.0	50.0	100.0
Ceratopogonidae	0.0	0.0	10.0	40.0	0.0	20.0	0.0	0.0
Terrestrial insects	0.0	20.0	0.0	60.0	10.0	20.0	0.0	40.0
Miscellaneous:								
Hydracarina	20.0	40.0	10.0	40.0	50.0	40.0	0.0	40.0
Pelecypoda	0.0	0.0	0.0	40.0	0.0	0.0	0.0	60.0
Gastropoda	0.0	0.0	0.0	0.0	10.0	0.0	0.0	20.0
Oligochaeta	0.0	0.0	0.0	20.0	0.0	20.0	0.0	20.0
Plant matter	80.0	40.0	80.0	100.0	100.0	100.0	50.0	100.0
Fish scales	10.0	20.0	30.0	20.0	60.0	0.0	0.0	40.0
Sample size	10	5	10	5	10	5	2	5

environment. Hall, Cooper and Werner (1970) found that the biomass density of benthos was similar in ponds with and without bluegill. Hayne and Ball (1956), however, found standing crops of benthic food organisms two to five times greater in ponds without sunfish than in those with sunfish. The biomass densities of the above organisms in Sanborn Lake in 1971 were 33-216 times greater than in 1972, except for Trichoptera and Ephemeroptera, which were about five and four times more numerous in 1971 (Chapter III).

Coleoptera, Mollusca, Oligochaeta, and Amphipoda were not numerous in the stomach contents of bluegill in 1971 or 1972 (Table 7). This is probably due to negative selection of bluegill for Coleoptera and Mullusca, and rapid digestion of oligochaetes (Gerking 1962; Ivlev 1961). Oligochaetes may be rejected as food items by bluegill (Hayne and Ball 1956), but amphipods are usually selected for (Gerking 1962; Emig 1966).

Most food items comprised a smaller numerical percentage of the stomach contents of bluegill in 1972 than in 1971, because the percentage of Cladocera in the diet increased greatly in 1972 (Table 9). Cladocera were 92% of the total number of items in the ration of all age classes of bluegill in 1972; only 65% in 1971 (Table 9). This increased utilization is presumably due to an increase in abundance of Cladocera in the environment rather than greater vulnerability to predation, although the selectivity of bluegill for Cladocera is well documented (Gerking 1962; Emig 1966; Patriarche and Ball 1949; Wahlquist 1970). The principle Cladocerans in the diet of bluegill were the limnetic genera Daphnia and Bosmina, so hydrophyte removal would not be expected to change concealment or vulnerability. Copepoda also increased in the diet but were not numerous in 1971 or 1972 (Table 7). The greater

Table 9. Percentage abundance of food items in four age classes of bluegill, 1971 and 1972.

Food Item	Age 1 1971	Age 1 1972	Age 2 1971	Age 2 1972	Age 3 1971	Age 3 1972	Age 4 1971	Age 4 1972
Crustacea:								
Amphipoda	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
Cladocera	77.7	97.0	37.6	90.1	62.1	96.0	84.1	85.4
Cladocera eggs	1.5	0.5	7.6	6.2	5.1	1.8	7.4	11.8
Copepoda	3.6	1.3	6.4	0.6	2.2	0.1	5.3	0.1
Ostracoda	7.6	0.7	32.2	0.5	8.7	0.2	0.0	0.1
Insecta:								
Coleoptera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Trichoptera	0.0	0.0	2.5	0.1	1.3	0.0	0.0	0.1
Ephemeroptera	0.5	0.1	0.6	0.1	0.2	0.0	0.0	0.0
Odonata	0.5	0.0	1.9	0.1	0.1	0.0	0.0	0.0
Diptera:								
Chironomidae	7.1	0.2	6.4	1.5	11.3	1.4	3.2	2.3
Ceratopogonidae	0.0	0.0	0.6	0.1	0.0	0.0	0.0	0.0
Terrestrial insects	0.0	0.0	0.0	0.1	0.2	0.0	0.0	0.0
Miscellaneous:								
Hydracarina	1.0	0.1	1.9	0.2	1.9	0.1	0.0	0.0
Pelecypoda	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.2
Gastropoda	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
Oligochaeta	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.0
Plant sprouts	0.0	0.0	1.3	0.3	6.7	0.0	0.0	0.0
Fish scales	0.5	0.1	1.0	0.0	0.6	0.0	0.0	0.0
Sample size	10	5	10	5	10	5	2	5

[1]Abundance is given as percentage of total number of items found in all stomachs of that age class.

growth in 1972 for age 1 bluegill may be more of a result of changes in the zooplankton population than of changes in the macroinvertebrate food supply (Chapter V). Wahlquist (1970) found bluegill production in ponds without water hyacinths, in which the fish ate zooplankton, was greater than in non-hyacinth ponds, in which they ate principally phytomacrofauna.

Lastly, there was a decrease in utilization of ostracods in 1972 (Table 7), even though nutrient enrichment following hydrophyte decay should have promoted their abundance by increasing their food supply (Tressler 1966). Ostracods are mainly benthic, so they may have been "passed over" by bluegill feeding principally on the limnetic Cladocera *Daphnia* and *Bosmina*, or anoxia near the water-substrate interface may not have allowed ostracods to flourish. If bluegill in 1971 were feeding among the hydrophytes more than in 1972, it is possible they consumed more ostracods in 1971, since ostracods also are found attached to hydrophytes (Tressler 1966).

Ration Diversity

Diversity of the total ration declined sharply in 1972 (Table 10), probably because of the greater percentage abundance of Cladocera in the ration in 1972 (Table 9). Hall et al., (1970) stated that bluegill are opportunistic feeders, and that their ration diversity decreases when food levels are high and they are able to select for the most abundant or vulnerable prey. Apparently, the bluegill of Sanborn Lake in 1972 shifted their diet to primarily Cladocera presumably because they were more abundant in the environment the fall after hydrophyte removal.

Table 10. Diversity of food items in bluegill and the macroinvertebrate community of Sanborn Lake, 1971 and 1972.

Age Class	Date	Sample Size	$\bar{d}_1$ [1]	$\bar{d}_2$ [2]
1	1971	10	1.44	3.03
1	1972	5	0.27	2.93
2	1971	10	2.54	2.93
2	1972	5	0.70	3.83
3	1971	10	2.23	3.15
3	1972	5	0.46	3.49
4	1971	2	--	--
4	1972	5	0.36	2.41
Envr.[3]	1971	6	--	3.50[4]
Envr.[3]	1972	6	--	3.62[4]

[1] Diversity based on all items in the ration.

[2] Diversity based on macroinvertebrate portion of the ration.

[3] Diversity of macroinvertebrates in the environment.

[4] Values are lower than in Chapter III, since Odonata, Oligocaeta and Hydracarina are "lumped" as they were for stomach analysis.

Diversity of the macroinvertebrate portion of the diet was slightly less for the age 1 fish in 1972; slightly greater for the age 2 and age 3 fish (Table 10). The decrease in ration diversity for age 1 fish may more nearly reflect the change in total diet diversity, because smaller fish consume primarily zooplankton (Patriarche and Ball 1949). The increase in ration diversity for age 2 and 3 fish may reflect increased vulnerability of macroinvertebrates to fish predation following removal of hydrophytes; or it may reflect the increase in diversity of macroinvertebrates in the environment (Table 10). Diversity in the environment actually decreased from 4.03 in 1971 to 3.76 in 1972 (Chapter III), but when odonates, oligochaetes and hydrachnids are "lumped" to conform to categories in stomach analyses, diversity increases from 3.50 in 1971 to 3.62 in 1972 (Table 10). Thus, "lumping" produces a misleading trend opposite to that actually found in the environment. Still, the trend of increasing diversity of the ration from 1971 to 1972 is similar to the trend of increasing environmental diversity based on "lumped" data. Perhaps, if items in the stomach could have been keyed to genera, diversity changes in the diet between 1971 and 1972 would similarly have reflected the actual decrease in diversity in the environment.

Selectivity of Food Items

Selectivity for Trichoptera, Ephemeroptera, and Hydracarina increased in 1972 (Table 11), even though their abundance in the environment in 1972 was less than 1971. Electivity for the plant associated ephemeropteran Caenis changed from -0.06 in 1971 to +0.61 in 1972 for age 1 fish, and -0.18 to +0.06 for age 2 fish (Table 11). Electivity

Table 11. Ivlev's electivity indices for macroinvertebrates in the diet of bluegill, 1971 and 1972.

Food Item	Age 1		Age 2		Age 3		Age 4	
	1971	1972	1971	1972	1971	1972	1971	1972
Coleoptera	-1.00	-1.00	-1.00	-1.00	-1.00	-1.00	-1.00	+1.00
Trichoptera	-1.00	+0.94	+0.77	+0.58	-0.04	+0.52	-1.00	+0.73
Ephemeroptera	-0.06	+0.61	-0.18	+0.06	-0.59	-0.03	-1.00	-1.00
Odonata	-0.62	-1.00	-0.21	+0.36	-0.94	-1.00	-1.00	-1.00
Diptera:								
Chironomidae	+0.77	-0.09	+0.61	+0.03	+0.78	+0.14	+0.82	+0.11
Ceratopogonidae	-1.00	-1.00	+0.91	-0.07	-1.00	-0.78	-1.00	-1.00
Hydracarina	+0.57	+1.00	+0.63	+0.99	+0.97	-0.04	+0.94	-1.00
Mollusca	-1.00	-1.00	-1.00	-0.12	-0.41	-1.00	-1.00	-0.09
Amphipoda	-1.00	+0.97	-1.00	-1.00	-0.94	+0.78	-1.00	-1.00
Oligochaeta	-1.00	-1.00	-1.00	-0.72	-1.00	-0.87	-1.00	-0.87

for the plant-associated trichopteran Ochrotrichia changed from negative in 1971 to positive in 1972 for age classes 1, 3 and 4 (Table 11). Gerking (1962) found Trichoptera favored by bluegill, but consumption of Ephemeroptera and Hydracarina proportional to their abundance in the environment. Electivity for Trichoptera, Ephemeroptera and Hydracarina in Sanborn Lake was greater in 1972 than 1971, which is probably because they became more vulnerable to predation after loss of vegetative cover.

Chironomids were abundant in the environment in 1972 relative to 1971 (Chapter III), but electivity for them decreased in all age groups (Table 11), possibly because bluegill shifted more toward Cladocera as their primary food source. Although electivity for chironomids declined in 1972, they were still a favored food item, as the generally positive electivity indices for both years attests (Table 11). Other workers have also shown that bluegill have a predilection for chironomids (Gerking 1962; Emig 1966; Patriarche 1949; Howell 1942).

Electivity was negative for oligochaetes, molluscs, ceratopogonids, amphipods, coleopterans, and odonates. Oligochaetes may be avoided as a source of food (Hayne and Ball 1956), or underestimated because of rapid digestion (Gerking 1962). Molluscs are often avoided as food items (Gerking 1962), but small sphaerids and gastropods may be eaten by bluegill (Hayne and Ball 1956). Amphipods are usually preferred food items (Gerking 1962; Emig 1966). Ceratopogonids increased in the environment in 1972, but were still infrequently eaten by bluegill. Coleoptera were found in the stomach of only one fish. Odonates were found to be of small importance in the bluegill diet by Gerking (1962) and Patriarche and Ball (1949), but they ranked third in importance in a study by Howell (1942).

Conclusions

The average biomass and number of nearly all macroinvertebrate taxa increased in the diet of all age classes of bluegill in 1972 following hydrophyte removal and decay. The bluegills' consumption of chironomids, pelecypods, and ceratopogonids increased in 1972, perhaps because of the 7 to 37-fold increase of these organisms in the environment as a result of detrital enrichment. The bluegills' consumption of Trichoptera, Ephemeroptera and Hydracarina was greater in 1972, perhaps because these organisms were more vulnerable to predation after loss of vegetative cover. Bluegill selected primarily for chironmids and hydrachnids in 1971 and 1972; Trichoptera and Ephemeroptera were also selected for in 1972, but all other organisms were not substantially sought either year. Chironomids were the most numerous and most selected macroinvertebrate food item; thus, of the macroinvertebrates, it seems that only changes in the environmental density of chironomids would have had much effect in determining the growth response of Sanborn bluegill to hydrophyte removal.

Numerically, Cladocera comprised an average of 79% of the stomach contents of bluegill throughout 1971 and 1972; percentage abundance of Cladocera in the diet increased from 65% in 1971 to 92% in 1972. Percentage abundance of Cladocera in the stomach contents of age 1 bluegill rose from 78% in 1971 to 97% in 1972. This greater consumption of Cladocera in 1972 probably aided growth of age 1 fish more than older ages because zooplankton are nearly the sole source of food for bluegill less than 51 mm (Patriarche and Ball 1949), whereas larger, older fish can also rely on macroinvertebrates as a food source.

Gravimetrically, the zooplankton portion of the diet of age 1 fish was greater than the macroinvertebrate portion in 1971, but less than the macroinvertebrate portion in 1972. Age 1 fish were 39 mm in total length in 1971 but 47 mm in 1972; it is therefore postulated that fish in 1971 were too small to consume anything but zooplankton, whereas fish in 1972 had grown to a size at which chironomids could be used. Hall et al. (1970) stated that selectivity of bluegill is usually based on size of the food particle, and small fish are limited to zooplankton because other items are too large. The fact that growth of age 1 bluegill was greater in 1972 generally presupposes that the available food supply was greater in 1972; thus it is further postulated that entomostraca (mainly Cladocera) must have been more available in 1972 since consumption of them increased, and since they were probably the only source of bluegill less than 51 mm had for effecting an increase in growth. As age 1 bluegill in 1972 approached 51 mm total length, they were able to utilize chironomids; thus, biomass of the macroinvertebrate fraction of the diet increased, and growth on the more energy efficient diet of chironomids was enhanced.

Growth of larger bluegill was little different in 1971 and 1972 (Chapter V). The principle food items in the diet of older bluegill were Cladocera and chironomids. Apparently, these items were not large enough to make food seeking efficient, nor were environmental increases following hydrophyte removal great enough to effect a substantial change in growth.

CHAPTER V

GROWTH AND CONDITION OF BLUEGILL BEFORE AND AFTER HYDROPHYTE REMOVAL USING DIURON

Introduction

Fish growth changes following hydrophyte removal with herbicides have been described by several workers, but conclusions regarding the effect have not been consistent. Increased growth has been observed, and attributed to (1) increasing the biomass and availability of food organisms, and (2) decreasing interspecific competition through toxic or predatory reduction of sunfish numbers (Surber 1945; Cope, Wood and Wallen 1970; Cope, McCraren and Eller 1969; Walker 1963). Decreased growth has been observed and attributed to (1) decrease of food organisms through poisoning, oxygen depletion or habitat removal, (2) chronic toxicity to the herbicide (Lawrence 1958, 1965; Gilderhus 1966; Houser 1963; McCraren, Cope and Eller 1969; Johnson and Julin 1974). Other workers have found growth little affected (Gilderhus 1967; Walker 1964). In most studies growth has not been measured, but the claim made that "no significant effects on fish was observed" (Van der Weij, Hoogers and Blok 1971; Van der Weij 1967; Blok 1967; Riemer 1964; Lawrence, Funderburk, Blackburn and Beasley 1965; Walker 1964, 1965; and others). Studies using the herbicide diuron (the chemical used in the present study) generally fall into the last class. McCraren et al. (1969), however, found standing crops of sunfish decreased in ponds treated with 0.5-3.0 mg/l diuron.

In 1972, diuron was applied to a 4.58 hectare pond in Payne County, Oklahoma, in order to study the effects of hydrophyte removal on the growth and condition of a stunted bluegill population. The diuron concentrations used in the present study were not acutely toxic to fish or fish-food organisms (Chapter II). Changes observed in standing crop of macroinvertebrates and fish growth should probably be attributed to removal of hydrophytes per se.

Growth and condition of bluegill for the summers of 1971 and 1972 were compared using samples collected September 1971 and October 1972. Hydrophyte and macroinvertebrate biomass, numbers and species, bluegill food habits, and dissolved oxygen were measured before and after treatment. These parameters were studied because of their obvious potential causal relation to changes in bluegill growth.

Methods

Bluegill were collected in September 1970 and 1971, and monthly from April through October 1972, using a boat-mounted electrofishing apparatus powered by a 230-volt, 3000 watt AC generator. In 1972, catch by electrofishing was supplemented with catches from two 1.27 cm mesh barrel traps and six 0.64 cm wire mesh minnow traps. Length, weight, scale, and stomach samples were obtained on all fish. Fish caught in 1970 and 1971 were preserved in 10% formalin for one week, then transferred to 70% isoproponal. Fish caught in 1972 were frozen. Samples frozen five months before processing gave an overestimate of live length and an underestimate of live weight. Estimates of live length and weight based on samples of 43 and 74 fish, respectively, were made for frozen samples, using equations 1 and 2.

$$L_1 = 1.4546 + 0.9606L_f, \ R^2 = 0.997 \qquad (1)$$

$$W_1 = 0.2105 + 1.0325W_f, \ R^2 = 0.999 \qquad (2)$$

where: L = length, W = weight, 1 = live, and f = frozen.

A directly proportional change in the frozen-live relationships was assumed for samples frozen less than five months before processing. Total and standard lengths were measured to the nearest millimeter for the 1970, 1971, and April, 1972 samples. Total lengths for the remaining samples were obtained from an estimating equation derived from the linear relationship between total and standard lengths in a sample of 73 fish from the 1970 collection:

$$TL = 0.1411 + 1.2955 \ SL, \ R^2 = 0.999$$

Weights were measured to the nearest 0.1 g. Condition was determined for each age class on each date using the formula $K = W \cdot 10^5/L^3$.

Ten to fifteen scales were taken from the left side below the lateral line where the tip of the pectoral fin meets the body. Scales were mounted between glass slides, or impressions were made on plastic slides using a roller press. Scales or impressions were examined at 80 mangifications using a scale projector with a 16 mm micro-tessar lens. The number and location of scale annuli used to estimate age and growth were determined according to the criteria of Regier (1962). Linear and curvilinear (2nd degree polynomial) length-weight and body-scale relationships were calculated for fish on each collection date.

Empirical total lengths and growth increments calculated by subtracting back-calculated length at annulus from empirical length at capture in the fall were both used to compare the growth of all age classes of bluegill before and after hydrophyte removal. Statistical methods used were from Snedecor and Cochran (1967). Growth history (in total length attained and growth increment) of bluegill from 1966 to 1972 was based on a combination of empirical and back-calculated values. Back-calculation was based on Lee's formula with a correction for intercept (Lagler 1956).

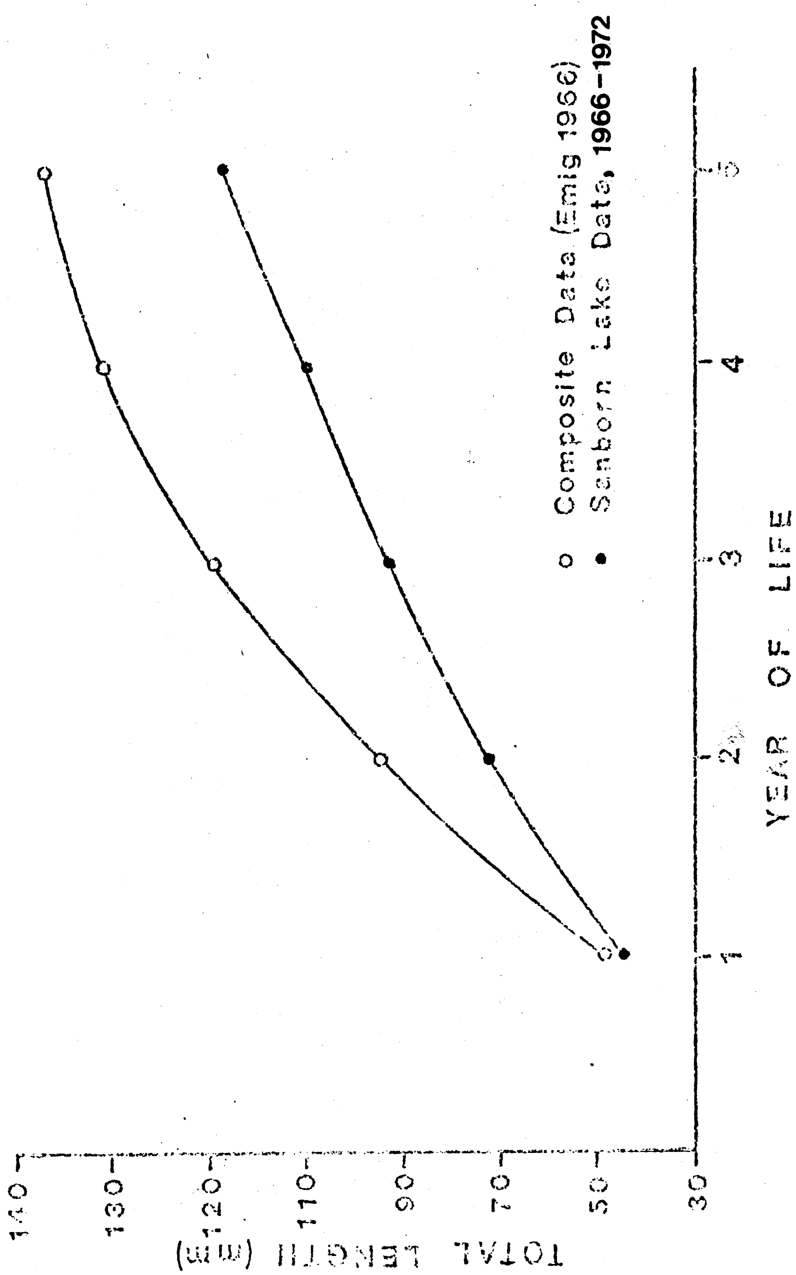
Methods for assessment of hydrophyte control and sampling, macroinvertebrate community response, and bluegill food habits changes are given in Chapters II, III, and IV, respectively.

Results and Discussion

Before Hydrophyte Removal

Growth of bluegill in Sanborn Lake between 1966 and 1972 was considerably less than the average growth from various locations in the United States (Figure 8). For age 1 and 2 bluegill, growth was about equal to the slowest growth reported in Oklahoma, but for older fish it was less than the slowest reported (Houser and Bross 1963; Jenkins, Elkin and Finnell 1955). Bennett (1971) states that 127 mm-203 mm bluegill with an index of condition of 7.1-8.0 denotes normal plumpness. The average index of condition for Sanborn bluegill was 7.4; however, the average length was only 86 mm. Annual total mortality ranges from 60-87.7 percent for bluegill populations (Emig 1966). Age frequency data from Sanborn indicated an annual total mortality of about 45 percent. Mortality was highest for Sanborn bluegill between the second and third years (63 percent), and no specimens were found over five years old. Introgressive hybridization is pronounced in Sanborn Lake, with the

Figure 8. Comparison of bluegill growth in Sanborn Lake with an estimate of the national average.



Composite Data (Emig 1966)
Sanborn Lake Data, 1966–1972
YEAR OF LIFE
1
2
3
4
5
TOTAL LENGTH (mm)
140
130
120
110
90
70
50
30

hybrids showing growth characteristics little different from the stock bluegill population. Trautman (1957) states such hybridization can be found where there is overcrowding. It is obvious that in Sanborn Lake too many bluegill were surviving, causing acute competition, poor growth, and a short life span.

In November, 1969, the pond was treated with Fintrol-5 (1 ppb Antimycin-A) to thin the stunted population. Since antimycin is selectively lethal for smaller fish, one would expect a thinning of the 1969 young-of-the-year population to result in decreased competition the following summer which would appear later as increased second-year growth. Total length attained by the end of the second year of life was 69 mm, 79 mm, and 68 mm for year classes 1968, 1969, and 1970 (Table 12); the 1968-69, 1969-70 differences were significant ($P = 0.002$; $P<0.001$). Second-year growth increment in 1970 was 9 mm greater ($P<0.001$) than in 1969 and 5 mm greater than in 1971 ($P<0.001$) (Table 13). Thus, second-year growth of the 1969 year class did improve after numerical reduction of the year-class using antimycin.

The effects of the antimycin treatment, however, did not last more than a season, nor were other age classes substantially affected. Average total length of the 1969 year class in the third year of life (1971) was slightly greater than third-year length of the 1968 and 1970 year classes ($P=0.248$, $P=0.094$), but growth increment was slightly less ($P=0.300$, $P=0.450$). The 1969 year class apparently reached a size by the third year of life at which food was insufficient because of competition with the four and five-year olds, or some other factor became limiting to growth. Other year classes which were comprised of larger fish at the time of the antimycin treatment seem to have responded little

Table 12. Total length (mm) of bluegill attained at the end of each year of life, 1966-1972.

Year Class	Length to Annulus 1	2	3	4	5
1972	47 ± 3(44)[1]	-	-	-	-
1971	39 ± 3(26)[1]	73 ± 3(38)[1]	-	-	-
1970	41 ± 2(22)[1]	68 ± 3(41)[1]	93 ± 3(30)[1]	-	-
1969	44 ± 2(26)[2]	79 ± 3(26)[1]	98 ± 4(11)[1]	109 ± 8(10)[1]	-
1968	43 ± 5(13)[2]	69 ± 6(13)[2]	93 ± 7(13)[1]	107 ± 44(2)[1]	128 ± 8(6)
1967	42 ± 4(8)[2]	69 ± 5(8)[2]	87 ± 5(8)[2]	104 ± 6(8)[1]	-
1966	46 ± 4(6)[2]	74 ± 5(6)[2]	91 ± 8(6)[2]	107 ± 8(6)[2]	119 ± 12(6)[1]

[1]These are total lengths observed from fall collections. Ninety-five percent confidence interval and (sample size) are given.

[2]These lengths are back-calculated from the Fall, 1970, sample.

Table 13. Mean annual growth increments of bluegill, 1966-1972.

Year of Growth	Age 1	2	3	4	5
1972[2]	47 ± 3(44)[1]	30 ± 2(38)	23 ± 3(30)	16 ± 6(10)	13 ± 6(6)
1971[2]	39 ± 3(26)	30 ± 2(41)	21 ± 5(11)	21 ± 102(2)	-
1970[2]	41 ± 2(22)	35 ± 2(26)	24 ± 5(13)	17 ± 8(8)	12 ± 13(6)
1969[3]	44 ± 2(26)	26 ± 4(13)	18 ± 6(8)	16 ± 16(6)	-
1968[3]	43 ± 5(13)	27 ± 4(8)	17 ± 3(6)	-	-
1967[3]	42 ± 4(8)	27 ± 3(6)	-	-	-
1966[3]	46 ± 4(6)	-	-	-	-

[1]The 95 percent confidence interval is given with sample size in parentheses.

[2]The increments for growth are obtained from empirical length in fall of year indicated minus back-calculated length at previous annulus for the same fish.

[3]Increments based on lengths back-calculated from the September, 1970, collection.

to the antimycin treatment. The 1968 year class, in its third year of life in the summer of 1970, attained a slightly larger total length and had a larger growth increment than previous year classes for their third year of life, but these differences were insignificant ($P=0.286$, $P=0.151$). Apparently, numerical densities of larger fish (older age classes) were not reduced substantially by the antimycin, so competition continued and prevented any growth response; moreover, the large fish were not vulnerable to capture by the average size predator so the numbers were not thinned by predation following hydrophyte removal.

Changes in condition between 1970 and 1971 may also be used to describe the effect of the antimycin treatment (Table 14). The decreases in condition between 1970 and 1971 for ages 2, 3, and 4 may represent increased competition as the 1969 year class reached a size by its third year of life at which it became capable of competing with 4 and 5 year-olds for larger macroinvertebrates. The only significant decrease in condition was between 1970 and 1971 age 2 fish ($P=0.019$); and this would be expected because the population of the age 2 fish (1969 year class) were thinned by the antimycin treatment. The significant ($P<0.001$) increase in the condition between 1970 and 1971 age 1 fish (young-of-the-year, or YOY) is not easily explainable. However, if the 1970 year class experienced better survival because of decreased competition for zooplankton with the reduced 1969 year class, the increased density of YOY could cause increased intra-age (sibling) competition with resulting low condition. Since population estimates were not made in 1970 or 1971, the explanation for the low condition of 1970 YOY must remain speculation.

The 9 mm increase in second-year growth increment for the 1969

Table 14. Condition coefficients of bluegill, 1970-1972.

	Date of collection			t-test	
Age	1970	1971	1972	'70 vs '71	'71 vs '72
1	1.38	1.60	1.64	($p<.001$)	($p=.373$)
2	1.51	1.45	1.63	($p=.019$)	($p<.001$)
3	1.49	1.43	1.61	($p=.129$)	($p<.001$)
4	1.49	1.40	1.57	($p=.282$)	($p=.038$)
5	1.59	-	1.64	-	($p=.444$)[1]

[1]Test is '70 vs '72.

year class is not considered of practical significance; nor are the 10 and 5 mm increases in total length attained for age 2 and 3 fish in 1970. Changes of 5-10 mm in average size would scarcely be perceptible to fishermen. Growth of bluegill in Sanborn Lake, 1966-1971, remained stunted prior to hydrophyte removal.

Effects of Hydrophyte Removal

The increases in total length attained between 1971 and 1972 (Table 12) of 8 mm for age 1 fish and 5 mm for age 2 were significant ($P<0.001$; $P=0.002$; Table 15). However, growth increment (a better measure of change in growth) was significantly different only for age 1 fish ($P<0.001$). Changes in growth increment and total length for other ages were nonsignificant (Table 15). Thus, hydrophyte removal substantially affected only growth of age 1 bluegill.

Condition was greater for all age classes after hydrophyte removal in 1972, but the increases over 1971 were significant only for ages 2-4 (Table 14). Enhanced condition in 1972 may be related to the 10-fold increase in biomass of the stomach contents, the 14-fold increase in the biomass of chironomids in the 0-1 m stratum, and a possible environmental rise in density of Cladocera.

The growth response of bluegill to hydrophyte removal would have to be included in the "no significant effects on fish" category. This would agree with the findings of most workers, as mentioned in the chapter introduction. One would have to consider hydrophyte removal a poor method for improving a stunted bluegill population. A 9 mm increase in growth increment of only one age class does not cause a size change perceptible to fishermen. It should be noted, however,

Table 15. Comparison of changes in three parameters indicating bluegill growth between 1971 and 1972.[1]

Table	Parameter	Age 1	Age 2	Age 3	Age 4
12	total length	+($p < .001$)	+($p = .002$)	-($p = .107$)	+($p = .300$)
13	growth increment	+($p < .001$)	0($p > .500$)	+($p = .450$)	-($p = .450$)
14	condition coefficient	+($p = .373$)	+($p < .001$)	+($p < .001$)	+($p = .038$)

[1]Increase in growth denoted by +, decrease by -, no change by 0. Probabilities that there is no significant difference are given.

that growth response in a severely stunted, overcrowded population might logically be less than in a normal population. Perhaps the largemouth bass population of Sanborn was too small to reduce the density of bluegill sufficiently to have reduced intraspecific competition. Population estimates of the bluegill and bass populations would have been useful in this study.

On the positive side, it is evident that hydrophyte removal had no substantial adverse effects on growth or condition of bluegill. Although dissolved oxygen concentrations in Sanborn Lake were lowered to less than 5 mg/l at depths below 2.25-3.00 m, 60-80% of the water column remained well-oxygenated; and fish were probably able to avoid stressful areas. Diuron applied in the concentrations used caused no apparent acute toxicity, at least through the end of that growing season. Hydrophyte removal by diuron for recreational enhancement appears safe and effective.

<u>Correlation with Environmental Changes</u>

The growth response of Sanborn bluegill to hydrophyte removal would have to be included in the "no significant effects on fish" category. This would agree with the findings of most authors, as mentioned in the chapter introduction. It should be noted, however, that growth response in a severely stunted, overcrowded population might logically be less than in a normal population.

Biomass density of macroinvertebrates (minus pelecypods) increased by 50% in the 0-1 meter stratum of Sanborn pond following hydrophyte removal. Chironomids, the most numerous macroinvertebrate food item in the stomachs of bluegill in 1971 and 1972 increased in the environment from 404 per m^3 in 1971, to 3055 per m^3 in 1972; probably in response to increased organic enrichment following hydrophyte decay. Although numerical densities of phytomacrofauna declined following hydrophyte removal, abundance of these organisms in the diet of bluegill increased in 1972. This probably reflects increased vulnerability to capture, due to loss of vegetative cover. The macroinvertebrate portion of the bluegill diet increased in response to increased abundance and vulnerability of food organisms. Condition was enhanced and growth increased substantially for YOY bluegill.

Zooplankton populations were not monitored in this study, but the great increase of Cladocera in the bluegill diet, from 65% of the numerical count in 1971 to 92% in 1972, indicates that they must have increased substantially in the environment. Wahlquist (1970) found greater sunfish production in ponds without water hyacinths, than in those with hyacinths. Wahlquist attributed the greater production in non-hyacinth ponds to a greater abundance of plankton, whereas production of food organisms in the hyacinth ponds was principally of organisms associated with hyacinth roots. Wahlquist did not measure benthos, nor did he consider the importance of detritus to zooplankton or phytomacrofauna.

In this study, zooplankton comprised 38-97% of the total number of food items in stomachs of bluegill in 1971 and 1972. Gravimetrically, phytomacrofauna and benthos comprised a greater percentage of the stomach contents as fish grew larger. Zooplankton were probably more important to age 1 fish, because their diet until 51 mm total length is primarily entomostraca (Patriarche and Ball 1949). Gravimetric composition of the diet of age 1 bluegill was 73% zooplankton in 1971, but was 51%

macroinvertebrates in 1972. Age 1 fish in 1972 were 8 mm larger than in 1971, so the dietary shift from zooplankton to macroinvertebrates may reflect the young fishes enhanced capability to utilize the larger macroinvertebrate food items. Increased consumption of zooplankton early in 1972 may have helped age 1 fish to more rapidly attain a size at which they could effectively use chironomids.

Growth of age 2-4 fish in 1971 was not significantly different from the growth of age 2-4 fish in 1972 despite a 10-fold increase in the biomass of the stomach contents. Gravimetric composition of the diet of age 2-4 fish in 1971 and 1972 was principally macroinvertebrate and principally chironomids, although Cladocera were the most numerous item. In this study, environmental response of zooplankton and chironomids to hydrophyte removal was apparently not great enough to affect growth of the principal portion of Sanborn's bluegill population.

Conclusions

In Sanborn Lake, annual recruitment of bluegill sunfish exceeds the food supply such that poor growth is an annual problem. Predation by largemouth bass on bluegill has not been effective in reducing sunfish numbers to a density which would reduce competition and provide better growth, and a longer life span.

An attempt was made to rectify this situation by selectively killing smaller fish with antimycin-A in the fall of 1969. In 1970, the yearlings which were survivors of the 1969 year-class grew 9 mm better than yearlings grew in 1969 prior to the antimycin treatment. This enhancement was not obvious by the third year of life; nor were other year classes affected significantly.

Hydrophyte removal using diuron in 1972 also failed to improve poor growth substantially, except for the fish in their first year of life; and that increase was too small to be perceptible by fishermen. Condition increased significantly for ages 2, 3, and 4, and insignificantly for age 1 ($P<0.001$, $P<0.001$, $P=0.038$, $P=0.373$). This may be a result of the increased consumption of Cladocera and macroinvertebrates in 1972 over 1971. Growth increment in 1972 was 8 mm greater than 1971 for age 1 fish, but growth of other ages in 1972 was not different from 1971. These changes can hardly be considered of practical significance. The fact that growth did not decline and condition improved does, however, indicate that hydrophyte removal using diuron caused no overly adverse effects on bluegill. Hydrophyte removal using diuron does seem a useful managerial tool for improving recreational waters.

Certain findings in this study suggest that hydrophyte control may lead to improved growth of bluegill, while others could diminish growth. Biomass and numerical densities of some food organisms (chironomids) increased in the 0-1 m stratum after hydrophyte removal; others (phytomacrofauna) decreased. Zooplankton may have increased in the environment and been largely responsible for the improved growth of age 1 fish in 1972. Availability of macroinvertebrate food organisms in the 0-1 m stratum apparently increased after loss of hydrophytes. Numbers, biomass, and availability of food organisms in the 1-3 m stratum decreased, probably as a result of oxygen depletion. Fish could potentially have been harmed by dissolved oxygen concentrations below 5 mg/l, however, since concentrations less than 5 mg/l occurred only in the bottom 20-40% of the water column in 1972, bluegill were probably able to avoid stressful areas. Toxicity to diuron might explain the decline in numbers of oligochaetes, Chrysops and Chaoborus; but most changes in macroinvertebrates seem related to hydrophyte decomposition. The net effect of these changes seems to have "balanced out" such that bluegill growth response to hydrophyte removal in Sanborn Lake must necessarily be placed in the "no significant effects on fish" category.

The fact that perturbations both potentially harmful and potentially beneficial to macroinvertebrates and bluegill occurred in Sanborn Lake after hydrophyte removal may indicate that variations in the herbicide and techniques used, weather, the pond ecosystem, and so on, are probably responsible for the varying results obtained by other workers. For example, if a herbicide is improperly applied in toxic quantities, or if bass predation is significant, bluegill numbers would decrease with resultant decreased competition and increased growth (Cope et al. 1970; Cope et al. 1969). If the pond contains large amounts of hydrophytes, death and decay will increase numbers of detritophagic and planktonic food organisms through organic enrichment (Harp and Campbell 1964; Walker 1963; Newman 1967). This is true, assuming that decomposition of the hydrophytes does not occur so quickly that severe oxygen depletion occurs, as is often the case when diuron is applied above 0.5 ppm. (Johnson and Julin 1974). If the food supply in a pond is principally phytomacrofauna, hydrophyte removal could decrease the food supply of fish (Hilsenhoff 1966; May, Hestand and Van Dyke 1973). If plants were not numerous in the pond, hydrophyte removal would logically have little effect. Oxygen depletion would less readily occur unless the phytoplankton community were large.

The negligible growth response of fish observed by most investigators following hydrophyte removal indicates that in most cases, careful application of reasonably safe herbicides will produce perturbations which can be effectively handled by the pond community's ecological homeostatic mechanisms. The Sanborn community was able to "balance" a decrease in phytomacrofauna with an increase in detritophages, such that little change in total numerical density of macroinvertebrates resulted. Numerical density of macroinvertebrates declined in the 1-3 meter stratum, but increased in the 0-1 meter stratum. Also, an increased vulnerability of phytomacrofauna seems to have balanced a decline in environmental abundance, such that their importance in the bluegill diet was not significantly changed. Decrease in plant cover harmed many organisms, but decomposition of the plant biomass provided detritus for others. Such changes as observed are, I feel, mere reflections of the chemical, plant, and animal interactions that took place to stabilize the pond environment.

CHAPTER VI

SUMMARY

The objective of this study was to determine growth response of a stunted bluegill population to hydrophyte removal by diuron. Efficacy of the Aqua-trol formulation of diuron was tested. Changes in dissolved oxygen concentration, the macroinvertebrate community, and bluegill food habits were measured to determine possible reasons for observed fish growth response.

The Aqua-trol formulation of diuron was applied 22 April 1972, at 0.10 mg/l (0.07 ppm diuron) and again six weeks later to areas of concentrated hydrophytes at 0.26 mg/l (0.18 mg/l diuron). These treatments were effective in reducing hydrophytes to 0.1% of pretreatment densities. Dissolved oxygen concentrations in Sanborn dropped below 5 mg/l in the lower one-third of the water column following onset of stratification in the deeper areas of the pond. Stratification was not maintained constantly, but occurred through most of the period from 15 May-7 July, when measurements were ended. Numerical density of macroinvertebrates in the 1-3 meter stratum declined between 1971 and 1972, perhaps as a result of oxygen depletion. No effects on fish were observed or suspected, since most of the water column contained sufficient dissolved oxygen. Mortality of fish was not observed, and toxic effects on fish are not suspected since growth of bluegill was little different following treatment. Unexplained decreases in oligochaetes, <u>Chaoborus</u>, and

Chrysops could reflect diuron toxicity, but total numerical density of macroinvertebrates changed little in the 0-1 meter stratum. The overall effect of herbicide treatment seems to indicate that the Aqua-trol formulation of diuron is a reasonably safe, effective aquatic herbicide.

Numerical and biomass densities of macroinvertebrates between 1971 and 1972 were not significantly different; however, this was a dynamic balance as substantial changes occurred in most taxonomic groups. Detritophagic organisms such as chironomids and pelecypods increased in abundance. Numerical density of oligochaetes declined, but biomass density diminished. Phytomacrofauna such as odonates, amphipods, coleopterans and gastropods decreased in abundance. Species diversity declined following simplification of the pond environment, elimination of several species of phytomacrofauna, and increased importance of a few species of detritophages.

Bluegill responded to increased chironomid production and greater availability of displaced phytomacrofauna by consuming more in 1972. Chironomids and Cladocera comprised most of the diet of bluegill in 1971 and 1972. Bluegill condition improved as a result of increased consumption of invertebrates and cladocera in 1972, but changes in growth increment over 1971 were minor. Only YOY bluegill showed a significant increase in growth increment of 8 mm ($P<0.001$). The improved growth response of YOY bluegill in 1972 may be due to greater zooplankton production following hydrophyte decay, release of nutrients, and phytoplankton growth. Cladocera were more numerous in the diet of YOY bluegill in 1972, and because zooplankton comprise most of the diet until bluegill attain 51 mm total length, it is likely that increased consumption of Cladocera in 1972 effected the 8 mm increase in growth.

Insignificant changes in growth for other age classes may indicate that zooplankton and chironomid increases in the environment were not great enough to effect significant growth response in a stunted bluegill population. It is also possible that the growth response of bluegill was caused by effective thinning of the YOY and ineffective thinning of larger fish by the small largemouth bass population.

It is possible that perturbations as observed in this study might have a greater effect on a more balanced fish population. Nevertheless, hydrophyte removal used alone, as in this study, did not substantially rectify the problem of stunting in bluegill sunfish. Growth response of bluegill to hydrophyte removal in the present study must be placed in the "no significant effects on fish" category, as has been the case with most studies by other workers. Results following hydrophyte removal will be situation specific, but, in most cases, the pond community apparently has the ability to negate such perturbations effectively through its ecological homeostatic mechanisms.

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VITA

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