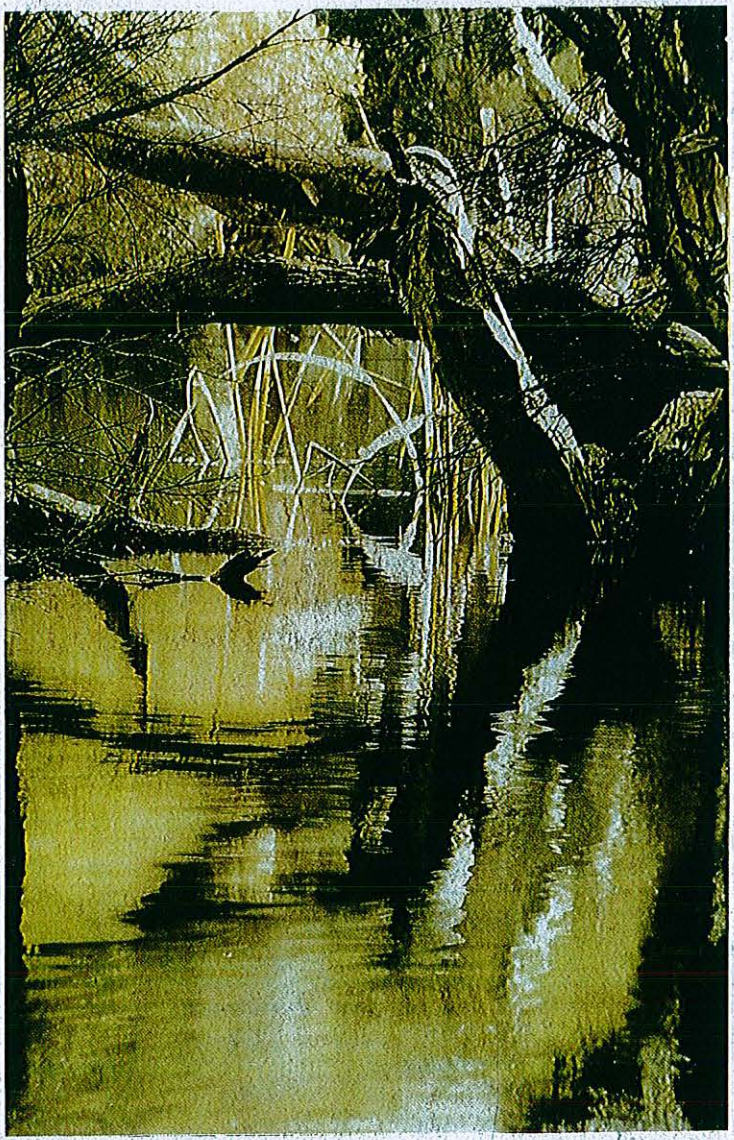
CHIRONOMID CONTROL IN PERTH WETLANDS

FINAL REPORT AND RECOMMENDATIONS



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Cover Photo: Star Swamp
By Craig Kinder

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Wetlands that are not highly enriched with nutrients and have had little of the natural vegetation removed, such as Star Swamp, generally do not support chironomids in sufficient abundance to cause nuisance problems. Rehabilitation of wetlands for chironomid control will involve reduction of nutrient inputs and replanting of fringing vegetation.

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Chironomid Control in Perth Wetlands

Final Report and Recommendations

Report on research carried out between 1987 and 1991

Prepared for — The Midge Research Steering Committee

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Acknowledgements

A Midge Research Steering Committee was established in Perth in August 1987 to oversee research into more effective and environmentally acceptable methods of reducing the nuisance caused by non-biting midges to residents living near wetlands in the Perth metropolitan region. During the 1990/91 research program the steering committee comprised representatives from the Department of Conservation and Land Management, the Environmental Protection Authority, the Cities of Armadale, Cockburn, Melville, Perth, South Perth, Stirling, Swan, and Wanneroo and the University of Western Australia. The Department of Planning and Urban Design were also represented on the steering committee from 1987 to 1990.

This study was funded by the three state and eight local government authorities listed above and their interest and support is gratefully acknowledged.

<u>Composition of the 1990/91 Midge Research Steering Committee</u>

Mr. J. Lane	CALM (Chairperson)
Mr. J. Sutton	EPA
Mr. N. Hume	City of Armadale
Mr. D. Ashby/ Mr. C. Lister	City of Cockburn
Mr. G. Dunne/Mr. R. Keegan	City of Melville
Mr. A. Van Leeuwen	City of Perth
Mr. B. Burnett/Mr. S. Camillo	City of South Perth
Mr. D. Rajah	City of Stirling
Mr. P. Kampen/ Mr. M. Pasalich	City of Swan
Ms. L. Schwarzbach/ Mr. P. Wesley	City of Wanneroo
Dr. D. Edward	University of Western Australia

The chairperson, Mr Jim Lane, is especially thanked for the valuable guidance and support he provided to the study team throughout the midge research program. Representatives of local government authorities also provided assistance in many ways and their contributions were greatly appreciated.

Throughout the study many people have worked on the project. Foremost among these is Susan Harrington who set high standards during the first two and a half years of the project. Other employees have provided invaluable assistance during the summer months; John Arena, Mark Garkaklis, Natalie Hill, Jonathon Mercer, Karen McRoberts, Belinda Robson, Carolyn Williams and Vicki Wilson.

We also thank the residents near North Lake who allowed us to place light traps on their properties and those at Forrestdale Lake who collected adult midges.

We gratefully acknowledge the provision of Sumilarv®, in addition to helpful advice and discussions, by Mr. Steve Broadbent of Wellcome Australia Ltd.

Nutrient analyses were carried out by Mr. Mark Lund and staff of the Centre for Water Research at Murdoch University. Weather data for Perth was obtained from the Bureau of Meteorology and for North Lake from the Murdoch University weather station.

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Debby Kirby investigated the relationship between nutrient enrichment and midge production at Perth wetlands as part of an Honours project in the School of Biological and Environmental Sciences at Murdoch University. Jan Otto van Erpers Roijaards, of the Prof. H.C. van Hall Instituut, Groningen, The Netherlands, investigated the influence of chironomid larvae on the release of nutrients from wetland sediments as part of a work experience program at Murdoch University. We are extremely grateful that both Debby and Jan were prepared to undertake research projects that would contribute to the knowledge of the relationship between nutrients and nuisance midges.

The provision of infrastructure, laboratory and computing facilities provided by the School of Biological and Environmental Sciences at Murdoch University is also gratefully acknowledged.

Synopsis

A Midge Research Steering Committee was established in 1987 in response to an apparent decline in the effectiveness of the insecticide temephos (Abate®) against chironomid midges in Perth wetlands, and environmental concerns about the continuing use of chemical control measures. This committee, which comprises representatives from local authorities and state government agencies, has funded a four year research project undertaken by scientists at Murdoch University. A two pronged approach has been taken by the research group. Investigations have been undertaken into short term alternatives to temephos and midge populations and their environment have been monitored with the aim of finding longer term control options. The results of this research and their implications for midge control are summarized below.

1) The composition and abundance of chironomid populations vary considerably between wetlands indicating that a universal solution to midge problems is unlikely to be found. However some general patterns have emerged. Those wetlands that are most nutrient enriched and support the most algal growth have the worst problems with midges. Variations in the abundance of midges between years appears to be related to the amount of algal growth (food availability) and to lake depth (which influences temperature and salinity) all of which are ultimately dependant upon year to year climatic variation . The results of monitoring at Forrestdale Lake suggest that more accurate predictions of the timing of midge problems, based on water temperature and lake depth, will be possible at some wetlands, leading to more efficient use of insecticides.

2) A standard, though flexible, monitoring program has been designed that can be used by authorities involved in midge control. This program, which is now established at many Perth wetlands, will allow midge populations to be monitored, nuisance problems to be predicted with a reasonable degree of accuracy, pesticides to be used more judiciously, and the effectiveness of pesticide applications to be determined.

3) Short term alternatives to the use of temephos have been investigated. Light traps may assist in midge control at some wetlands, however large numbers of traps may be required. Fogging of adults using a synthetic pyrethroid insecticide is not considered to be a viable control method because of poor midge control potential and possible adverse environmental effects. Chemical alternatives to temephos that have been tested include another organophosphate (chlorpyrifos), a synthetic pyrethroid (permethrin), a bacterial insecticide (B.t.i.), and an insect growth regulator (Sumilarv®). None of these were ideal. However, Sumilarv® was found to offer the best combination of effectiveness, environmental safety and

cost. This pesticide is expected to be registered for use in Australia by 1994.

4) Limited testing of Sumilarv® under laboratory and field conditions suggests that rates necessary to control the nuisance chironomid *Polypedilum nubifer* are safe for a variety of non-target organisms including invertebrates and fish. Since overseas studies have demonstrated some deleterious effects upon non-target invertebrate populations, further testing on indigenous fauna is being carried out.

5) An integrated approach to midge control is recommended for adoption in Perth. This will involve minimizing the use of insecticides and an increasing emphasis upon non-chemical methods. Restoration of wetlands with a special emphasis on water quality, is recommended as the basis for long term chironomid control. In particular, reduction of nutrient inputs to wetlands so as to reduce the growth of algae should be considered the most urgent aspect of wetland restoration. Other important strategies include the replanting of fringing vegetation and buffer zones of woodland and more appropriate land use practices within wetland catchments.

Summary

This report describes the results of the fourth year (1990/91) of a research program undertaken to determine more effective and environmentally acceptable methods for control of nuisance midges near urban wetlands. Because this report also represents the final report of four years of study reference is made to information obtained in previous years wherever appropriate.

Chironomid and Environmental Monitoring Programmes

• The regular larval monitoring program first undertaken by the Murdoch research group in 1987 was continued for a fourth year at North Lake and Forrestdale Lake in 1990/91. Monitoring was undertaken to investigate the relationship between larval densities and the level of adult nuisance and to identify the most appropriate times for pesticide treatments. Physico-chemical and nutrient data were also recorded at the same time as larval samples were collected. Monitoring of environmental variables was undertaken to identify key factors that might enable the onset of midge problems to be predicted simply, quickly and with a high degree of confidence. Determination of the relationship between chironomid abundance and evironmental variables would also provide the basis for development of longer term strategies for midge control. A regular larval sampling program was continued at Lake Monger by the Perth City Council with identifications provided by the Murdoch research group.

• Quantitative data on adult midge abundance and species composition were collected using light traps placed near four residences at North Lake and submerged emergence traps at North Lake and Forrestdale Lake.

North Lake

• The larvae of 17 species of chironomids were recorded at North Lake during the study. Larvae of the pestiferous species *Polypedilum nubifer* accounted for more than 70% of total larval density on most occasions. Most species more abundant during the warmer months. Densities of *P. nubifer* fluctuated substantially during spring, summer and autumn each year, partly in response to applications of Abate® or Sumilarv® and partly as a result of natural changes in population structure.

• The lowest spring/summer density of *P. nubifer* ever recorded at North Lake over the four year study occurred during 1990/91 and was accompanied by low rates of adult emergence and few reports of problems with nuisance swarms. In contrast, much higher larval densities had been recorded during the previous two spring/summer periods (1988/89 and 1989/90). The use of Sumilarv® at North Lake on two occasions during

1989/90 appeared to be at least partly responsible for a reduction in adult nuisance problems at the lake whereas applications of Abate® during 1988/89 appeared to be ineffective.

• The Murdoch University veterinary farm drain was diverted away from North Lake to an on-campus soak in autumn 1990 to meet the requirements of the EPA. The drain had been found to contribute a large proportion of the total external nutrient load to North Lake and its diversion was seen as an important step towards the improvement of water quality at the lake. Mean spring and summer concentrations of total phosphorus were lower in 1990/91 than those recorded in the previous four years and chlorophyll-*a* concentrations were lower than those recorded in the previous two years. However concentrations as low as those recorded in 1990/91 have been recorded in the past six years and indicate that internal loading fom the sediments may also be an important source of nutrients to North Lake. Further monitoring is considered necessary to fully determine the success of the drain diversion.

• The decrease in densities of larval *P. nubifer* and the reduction in adult nuisance problems at North Lake in 1990/91 appeared to be related to lower abundance of algae which in turn may be attributed to the reduced nutrient concentration in the water. Mean summer density of *P. nubifer* was significantly correlated ($r^2 = 0.97$) with mean spring and summer chlorophyll-*a* concentrations recorded for the four years of the study.

• The lower lake depth during 1990/91 may also have inhibited high population density of *P. nubifer*. Diversion of the Vet. Farm drain resulted in less water entering North Lake in 1990. Summer water levels subsequently fell below the area of sandy sediments which had supported large populations of *P. nubifer* in previous summers. The fine muds which formed a layer over the sand of the littoral zone this summer may have been less favourable to *P. nubifer* which tends to prefer cleaner sandy regions. The rapidly receding water level may also have stranded substantial numbers of egg masses and larvae on the exposed shoreline.

• The mean nightly catch of adults per light trap at North Lake was not significantly different between the three summers of 1988 to 1991 despite much lower larval densities and adult emergence at the lake in 1990/91. However the very high densities of *P. nubifer* recorded at nearby Bibra Lake in 1990/91 indicated that adults from this lake may have been contributing to catches at North Lake. Therefore midge control efforts at a single lake may not be successful if problems at nearby lakes are not also considered.

• The most constant predictor of larval midge density at North Lake was maximum temperature. Densities exceeding 8 000 larvae/m^2 generally occurred three to seven weeks after the maximum weekly temperature reached 20 ^oC. Although this information is of some practical value in indicating when midge problems may be about to develop it is too imprecise for decisions regarding the timing of pesticide treatments. Monitoring of larval densities, in conjunction with adult emergence, remains the best method for assessing whether or not a pesticide treatment is appropriate at North Lake.

Forrestdale Lake

• The larvae of 16 species of chironomid were recorded at Forrestdale Lake during the study. *P. nubifer* was far less dominant at this lake than North Lake, particularly during the summers of 1987/88 and 1990/91. *C. occidentalis, C. alternans, D. conjunctus, T. fuscithorax* and *P. villisomanus* also occurred in relatively high proprtions. Midge nuisance problems were less severe during the spring and summer of 1990 than during the three preceding years and appeared to be mainly due to the much earlier drying of the lake.

• The onset and duration of midge problems at Forrestdale Lake appears to be determined primarily by water temperature and conductivity, both of which are related to lake depth. A rapid rise in the density of *P. nubifer* occurs within four to six weeks of the median water temperature rising above 22 ^{o}C, unless the lake depth has fallen below 40 cm (in which case the lake should dry before *P. nubifer* reaches nuisance levels). If the lake depth is between 40 and 60 cm when the median temperature rises above 22 ^{o}C a problem may still occur but it is likely to be shortlived. The longer the lake contains water during January and February after the critical median weekly temperature has been exceeded the longer problems with *P. nubifer* are likely to continue. Abate® still appears to be an effective compound for midge control at Forrestdale Lake but proper timing of treatments is essential for good control.

• Manipulation of lake depth leading to earlier summer drying may be an effective means of controlling midge problems at Forrestdale Lake. While it may be argued that this approach is not wholly compatible with the nature conservation role of the lake, nor is the use of pesticides. In addition such manipulation may not be compatible with the aims of Typha (bullrush) control. A similar approach is currently being investigated for the control of mosquitoes and ceratopogonids (biting midges).

Lake Monger

• Larval densities of *P. nubifer* were substantially lower at Lake Monger in the summer of 1990/91 than during the previous summer (1989/90) and few problems with nuisance adults were reported. The reason for the difference between the two summers is not known. However, data from a variety of sources revealed that algal abundance was much lower at the lake in 1990/91 than in previous summers.

Other Lakes

• Further evidence of a positive relationship between chironomid abundance and nutrient enrichment of wetlands has been provided by data collected from 40 wetlands on the

Swan Coastal Plain as part of a separate project on wetlands classification. These data indicated that wetlands at which total phosphorus concentrations of greater than 250μg/l were recorded during summer also contained populations of *P. nubifer* at densities high enough to result in nuisance swarms of adults. Exceptions included wetlands which were tannin stained, polluted with organochlorine pesticides or almost dry.

Experimental work on nutrient-chironomid relationships

• An experimental test of the relationship between algal blooms and midge production was undertaken as an honours project by Debby Kirby in collaboration with the midge research group in 1990/91. Six experimental ponds were created within plastic lined water tanks. Nutrients, trace elements and an inoculum of algal cells were added to three ponds while the other three remained untreated and so acted as controls. The enriched ponds were found to contain significantly higher abundances of algae and chironomids than the control ponds. The presence of large numbers of zooplankton appeared to reduce algal abundance by grazing. These results have important implications for the management of wetlands because they support previous indications that midge problems are a response to nutrient enrichment and algal blooms.

• An experimental test of the influence of chironomids on the release of nutrients from wetland sediments was carried out by Jan Otto van Erpers Roijaards as a work experience project in collaboration with the midge research group. Twelve microcosms (glass cylinders) containing lake sediments and water from Bibra Lake were set up in the laboratory. Four treatments, with three replicates of each, corresponding to varying densities of *P. nubifer* were employed. The concentration of inorganic phosphorus in the water column increased in response to higher larval density. These results indicated that high densities of *P. nubifer* larvae of can contribute to the transfer of phosphorus from the sediments to the water column and therefore contribute further to nutrient problems within wetlands.

Emergence and movement of adult chironomids

• Diel patterns of emergence of chironomids at North Lake were investigated by removing trapped insects from emergence traps every two hours over 24 hours on several occasions during the spring and summer of 1990/91. The majority of adults of *Polypedilum nubifer* and *Tanytarsus bispinosus* emerged shortly after sunset. The main factor influencing the time of emergence appeared to be light intensity. These results indicate that for routine monitoring of adult emergence (which will be necessary if Sumilarv® is widely used as a control agent), emergence traps only need to be set from late afternoon to early evening. They also suggest that fogging of adults would be more effective after sunset than before.

• Comparison of 1990/91 North Lake light trap data with wind data collected at the Murdoch University Meteorological Station revealed that although wind appeared to have some influence on the movement of adult chironomids the attraction to lights is probably an overriding factor at even moderate wind speeds and that movement crosswind or even upwind is possible.

Investigations into the toxicity of Sumilarv®

• Sumilarv®, an insect growth regulator with juvenoid activity, has been identified as an alternative to the organophosphate pesticide, Abate®. Particularly for short term control of midges at wetlands where the latter no longer provides reliable control (eg. North Lake and Lake Monger). Laboratory trials indicate that an application rate of 10 kg product/ha of Sumilarv® 0.5G (assuming 50cm depth) should achieve good control in Perth wetlands. The suppliers of Sumilarv®, Wellcome Australia, have indicated that it will be available at a cost comparable to Abate® on a per hectare basis. Registration proceedings will start in 1992 and it is expected to be registered for use in Australia by 1994. Further trial use in wetlands may be permitted before then.

• The effects of Sumilarv® on both chironomids and non-target fauna were further tested in the field at North Lake between November and December 1990. No significant differences were detected between chironomid emergence or larval densities in the treated or control enclosures over the course of the experiment. However the experiment was marred by the fact that chironomid emergence rates fell unexpectedly just before the experiment began. Sumilarv® also appeared to have very little effect upon the populations of non-target organisms that were present in the enclosures. Two copepods, *Calamoecia tasmanica tasmanica* and *Mesocyclops* sp., the amphipod, *Austrochiltonia subtenuis*, the mayfly, *Tasmanocoenis tilyardi* , the caddisfly, *Ecnomus pansus*, the ostracod, *Cypricercus salinus*, nematodes, oligochaetes and the ceratopogonid, *Nilobezzia* sp. demonstrated very little sensitivity to the Sumilarv® treatment.

• A series of laboratory bioassays were performed to determine the relative toxicity of the active ingredient of Sumilarv®, pyriproxyfen, to species which coexist with chironomids in wetland habitats. Species tested included the ostracod, *Candonocypris novaezelandiae*, the amphipod, *Austrochiltonia subtenuis*, the corixid, *Micronecta robusta*, the mayfly *Cloeon fluviatile* and the goby, *Pseudogobius olorum*. The median effective and lethal concentrations of pyriproxyfen for the organisms tested ranged from 0.12 to 6.21 ppm. The concentrations of the pesticide required to detrimentally effect 50% of the gobies, ostracods and corixids were 621, 392 and 125 times higher, respectively than the proposed field rate of 0.01 ppm (0.01 ppm pyriproxifen = 10 kg Sumilarv® /ha). A concentration of pyriproxifen at ten times the field rate would kill or harm 50 % of the mayflies and amphipods, respectively. The emergence of mayflies did not appear to be inhibited by pyriproxifen (0.01ppm). Overall the bioassay results suggested that 0.01 ppm pyriproxifen was not acutely toxic to any of the non-target organisms tested.

• Validation studies are needed to determine whether the organisms tested in the laboratory respond in the same way in the field. The field test undertaken at North Lake supported these results. However the test was limited in its capacity to validate the laboratory tests as it represented only a single application rate.

• The results of the field and laboratory testing of Sumilarv® indicate that it appears to be a relatively safe compound for use in the urban wetlands as part of an integrated approach to the control of nuisance chironomids. Widespread use however cannot be recommended until further investigations into the effects of this pesticide on important aquatic organisms such as cladocerans (water fleas) and odonates (dragonflies) are completed. These organisms did not occur in large enough numbers at North Lake to be tested during this study, but a recent study from the United States found that both groups were adversely affected by Sumilarv®. Their results have serious implications because both cladocerans and odonates potentially exert either direct or indirect biological control on chironomid populations. Larval odonates directly prey on chironomids while cladocerans (in particular *Daphnia* sp.) reduce algal biomass in wetlands through grazing and so reduce the abundance of the chironomid food source. Further investigations into the effects of Sumilarv® on cladocerans are currently being undertaken by a member of the midge research group (K. Trayler) as part of an honours project at Murdoch University which is scheduled for completion in December 1991.

• Considerable care needs to be exercised with the large scale application of Sumilarv® to wetlands. The impact of an accidental spill on the aquatic ecosystem would be serious given that ten times the application rate could kill 50% of the mayflies and harm 50% of the amphipods present. We do not have any information on the effect that a spill of Abate® would have on a wetland where it had not been previously used. However, a similar or even greater effect from this broad spectrum pesticide could be expected. Great care must be exercised with the use of any pesticide in natural aquatic ecosystems.

Integrated chironomid control

• The results of this research program indicate that short term control of midge problems will be improved by monitoring larval densities, water depth and temperature so that larval thresholds and critical values can be used to predict appropriate times to treat. Pesticides (either Abate® or Sumilarv®) should not be applied unless the means to monitor the effectiveness of a treatment are also available. Monitoring programs similar to those already undertaken by the City of Stirling (Smyth 1991) and other municipal authorities must be established by all agencies responsible for the management of wetlands where midge problems occur.

• Differences between lakes in terms of the degree of enrichment, physico-chemical attributes and composition and abundance of midge populations mean that no universal formula for the short term solution of midge problems can be given. Reliance on pesticides will not solve problems indefinitely as resistance will develop. In addition, all pesticides, including narrower spectrum compounds such as Sumilarv®, will affect wetland food chains in an adverse way. An integrated approach to midge control is required where several control techniques are applied simultaneously. For example the use of light traps and limited pesticide treatments together with reduced nutrient inputs and the replanting of fringing vegetation are all techniques which can be used in combination at many wetlands. Manipulation of wetland water levels, as already noted for Forrestdale Lake, may represent an alternative, non-chemical, short term solution to midge problems.

• The problem of nutrient enrichment must be addressed as the key component of an ecologically sensitive midge control strategy for the wetlands of the Swan Coastal Plain. Research and monitoring undertaken during the final year of this study has demonstrated that midge problems are largely fuelled by high primary production. Midge problems are a symptom of a disturbed system and an effect of poor water quality. The water quality of wetlands must be improved for a longer term solution to midge problems. Reduction of external inputs by diversion of inflowing drains, similar to the recent diversion of the veterinary farm drain at North Lake, indicates one possible approach. The conversion of septic tanks to mains sewerage needs to be considered at Forrestdale where septic leachate may be the main source of diffuse nutrient input to the lake. Other approaches that warrant serious consideration include the reduction of fertiliser use in wetland catchments and the establishment of buffer zones of fringing vegetation. Buffer zones should be of sufficient width, height and structure to maintain wetland ecosystem function, to filter nutrients from surface and subsurface runoff and to act as a physical barrier to midge swarms.

• Appropriate urban planning is vital to preventing midge problems occurring at future residential developments sited near wetlands. In the absence of any specific research it is recommended that no development should be sited closer than 800 metres to a wetland and that any landuse activity proposed for within that distance should be wholly compatible with the objectives of wetland conservation. The removal of fringing vegetation must be avoided at all cost. Vegetation provides the basis of all wetland food chains and is integral to natural wetland structure and function. Its removal for urban or agricultural development is usually accompanied by the onset of excessive nutrient enrichment and subsequent midge problems. Problems with midges should be rare in non-enriched wetland habitats where little fringing vegetation has been removed.

Chapter One

Introduction

Non-biting midges (Chironomidae) have a world-wide distribution and occur in both running waters (streams and rivers) and standing waters (lakes and wetlands) and in waters that range from fresh to saline. Larval chironomids are a major component of the invertebrate fauna of the wetlands of the Swan Coastal Plain and the adults often form large nuisance swarms in the suburbs adjacent to wetlands during the summer months. A Midge Research Steering Committee was established in Perth in August 1987 to oversee research into more effective and environmentally acceptable methods of reducing the nuisance caused by non-biting midges to nearby residents of Perth's major wetlands.

The use of the pesticide Abate (temephos) to kill the aquatic larvae has been the main approach to chironomid control in Perth. However, after 20 years of use there is now evidence of resistance to this pesticide in some chironomid populations. Initial research concentrated on finding an effective alternative to Abate that was also economically and environmentally acceptable.

The wetlands around which the worst nuisance problems were experienced were observed to be those with the worst water quality problems. Thus, the investigation of the relationship between eutrophication and chironomid abundance has also been a major focus of the research.

This report describes the results obtained from the fourth and final year (1990/91) of the research program in conjunction with the first three years of the study. More detail of the first three years of the study were reported previously (Davis *et al.* 1988; 1989 and 1990) and readers should consult those reports to obtain a background to the work described here.

The objectives for the fourth year of study were as follows:

1. To continue monitoring larval densities and relevant environmental parameters at North Lake and Forrestdale Lake.

The regular larval monitoring program first undertaken by the Murdoch research group in 1987 was continued for a fourth year at North Lake and Forrestdale Lake in 1990/91. Monitoring was undertaken to investigate the relationship between larval densities and the level of adult nuisance and to identify the most appropriate times for pesticide treatments. Physico-chemical and nutrient data were also recorded at the same time as larval samples were collected. This information was required to facilitate the identification of critical factors which might enable the onset of midge problems to be predicted simply, quickly and with a high degree of confidence. Determination of the relationship between chironomid abundance and environmental variables might also provide the basis for development of longer term strategies for midge control.

2. To undertake laboratory testing of Sumilarv® against target and non-target fauna

Results obtained in previous years (1987-1989) indicated that an alternative to the organophosphate pesticide Abate was needed for effective short term control of midge problems at many urban wetlands. The high rates of Abate needed to achieve high mortality of *P. nubifer* under controlled conditions in the laboratory indicated that some resistance to Abate was present in the gene pool of some populations of this species. In addition, Abate is also considered to be less effective in eutrophic waters.

Abate is the only compound registered for use against midges in Western Australia and alternative compounds are not easily found. The choice of compounds to be tested largely depended upon whether or not they could be obtained in sufficient quantity and appropriate formulation for both our trials and subsequent use by local authorities on a scale similar to Abate. Initial studies carried out in 1987 to 1989 suggested that of the available alternatives Sumilarv® (pyriproxyfen) was the only one that appeared to be both effective against chironomid larvae and was relatively acceptable on environmental grounds. This compound, which is expected to be registered for use in Australia in 1994, was chosen for further more intensive investigation.

Sufficient granular and technical grade material for both field and laboratory trials was made available to us by Wellcome Australia. The results of field and laboratory trials of Sumilarv® undertaken in 1989/90 indicated that it had considerable potential as an alternative to Abate for the control of midges. Sumilarv® applied at a rate of 10 kg/ha (0.05 kg AI/ha; assuming 50cm depth) effectively inhibited chironomid emergence by more than 80% for a minimum of three weeks.

The effectiveness of Sumilarv® against the main nuisance species *Polypedilum nubifer* was investigated by field and laboratory experiments in 1989/90. In this fourth year the effectiveness of Sumilarv® on other nuisance species was also to be tested. Several laboratory bioassays were attempted using *Chironomus occidentalis*. However, this species proved difficult to keep alive in the laboratory and after several unsuccessful experiments this aspect of the study was discontinued.

A series of laboratory bioassays were to be performed to determine the relative toxicity of the active ingredient of Sumilarv®, pyriproxyfen, to species which coexist with chironomids in wetland habitats. Species to be tested included the ostracod, *Candonocypris novaezelandiae*, the amphipod, *Austrochiltonia subtenuis*, the corixid, *Micronecta robusta*, the mayfly *Cloeon fluviatile* and the goby, *Pseudogobius olorum.*

3. To undertake whole lake field trials of Sumilarv® using adequate controls.

The effects of Sumilarv® on both chironomids and non-target fauna were to be further tested in the field at North Lake in 1990/91. To properly determine the effectiveness of a pesticide under field conditions it is important to compare both treated and untreated areas so that the effects of the pesticide can be separated from any natural fluctuations that may have occurred in larval and adult populations. For this reason the effectiveness of Sumilarv® in the field was to be determined using plastic enclosures similar to those that had been used at North Lake during the previous year (Davis *et al.* 1990).

Whole lake field trials of Sumilarv® were also planned for Lake Monger but the unexpectedly low densities of chironomids throughout spring and summer of 1990/91 meant that treatment with Sumilarv® was not warranted and good experimental data were unlikely to be obtained. For this reason Sumilarv® was field tested in November 1990 at North Lake, where larval densities in early spring indicated that a treatment with an insecticide was warranted. Further field experiments were planned at North Lake, however the density of larvae decreased to very low levels for the rest of the summer period and no further treatments were required.

4. To investigate the relationship between algal blooms and midge production in Perth wetlands.

Initial work on midge problems at Perth wetlands (Davis *et al.* 1988) suggested that high densities of larval midges occurred as a response to nutrient enrichment. The most eutrophic wetlands, for example, North Lake, Bibra Lake and Lake Monger supported very high densities of larval chironomids and experienced severe problems with adult midges. Although the link between high nutrients, algal blooms and midge problems was recognised, time and budgetary constraints have previously prevented the establishment of a definite cause and effect relationship.

An experiment designed to test the effects of nutrient enrichment and algal blooms on chironomid abundance was undertaken during 1990/91 as an Honours project at Murdoch University by Debby Kirby in collaboration with the midge research group. The results of the project are summarised in this report and described in full in a separate Honours thesis which is available at the Murdoch University Library.

5. To determine the effect of wind speed and direction on the abundance of midges caught at light traps.

Data on the abundance of adult midges caught at light traps set at North Lake was to be compared with wind data collected by the Murdoch University Meteorological Station (located only several hundred metres to the north of the lake) to determine how much influence prevailing winds may have on the occurrence of nuisance swarms of midges in residential areas near wetlands.

6. In addition to these initial objectives the opportunity also arose to investigate the influence of chironomid larvae on the release of nutrients from lake sediments and to examine the patterns of emergence of adult chironomids.

a) Influence of chironomid larvae on sediment nutrient release: High densities of larval chironomids have been reported to contribute to the transfer of phosphorus from lake sediments to the water column. A small study was undertaken by Jan Otto van Erpers Roijaards (a work experience student from The Netherlands) to investigate whether the presence of large numbers of chironomid larvae in eutrophic wetlands in Perth may contribute to nutrient enrichment. Experimental microcosms containing lake sediments and water from Bibra Lake and a range of densities of larval chironomids were to be set up in the laboratory to test of the influence of chironomids on the release of nutrients from wetland sediments.

b) Patterns of emergence of adult chironomids: Diel patterns of emergence of chironomids at North Lake were investigated by collecting adult flies with emergence traps at two hourly intervals over 24 hours on different occasions at North Lake. This information is required to determine when traps need to be set for routine monitoring of emergence. This type of monitoring will be necessary if Sumilarv® is widely used as a control agent at Perth wetlands in future years.

Chapter Two

Chironomid and Environmental Monitoring Programs at Selected Perth Wetlands

INTRODUCTION

Chironomid populations have been monitored at several Perth wetlands for the three and a half year period between October 1987 and March 1991. North Lake and Forrestdale Lake have been monitored by the midge research group and the City of Perth has monitored chironomids at Lake Monger.

There were several main objectives of the monitoring programs. Changes in chironomid populations through spring and summer were to be investigated to examine the relationship between larval density and the level of nuisance caused by the adults. This information could then be used to time applications of insecticides more efficiently. Regular monitoring also allowed the effectiveness of pesticide applications to be investigated. With the aim of finding physico-chemical predictors of the onset of rapid growth of chironomid populations, various environmental variables were monitored concurrently with the chironomids. The final objective was to investigate the relationships between the general abundance of chironomids and environmental variables. It was hoped that this might suggest methods by which long term control of chironomids could be achieved.

This final report brings together all the monitoring data to provide an overall analysis with regard to the above aims.

METHODS

The methods used in the monitoring programmes conducted in 1987/88, 1988/89 and 1989/90, together with illustrations and descriptions of the lakes, have been covered in detail in previous reports (Davis *et al.* 1988, 1989 and 1990). The following is a brief summary of methods used during these years and in 1990/91. Maps of the three most intensively studied wetlands are given in Figure 2.1.

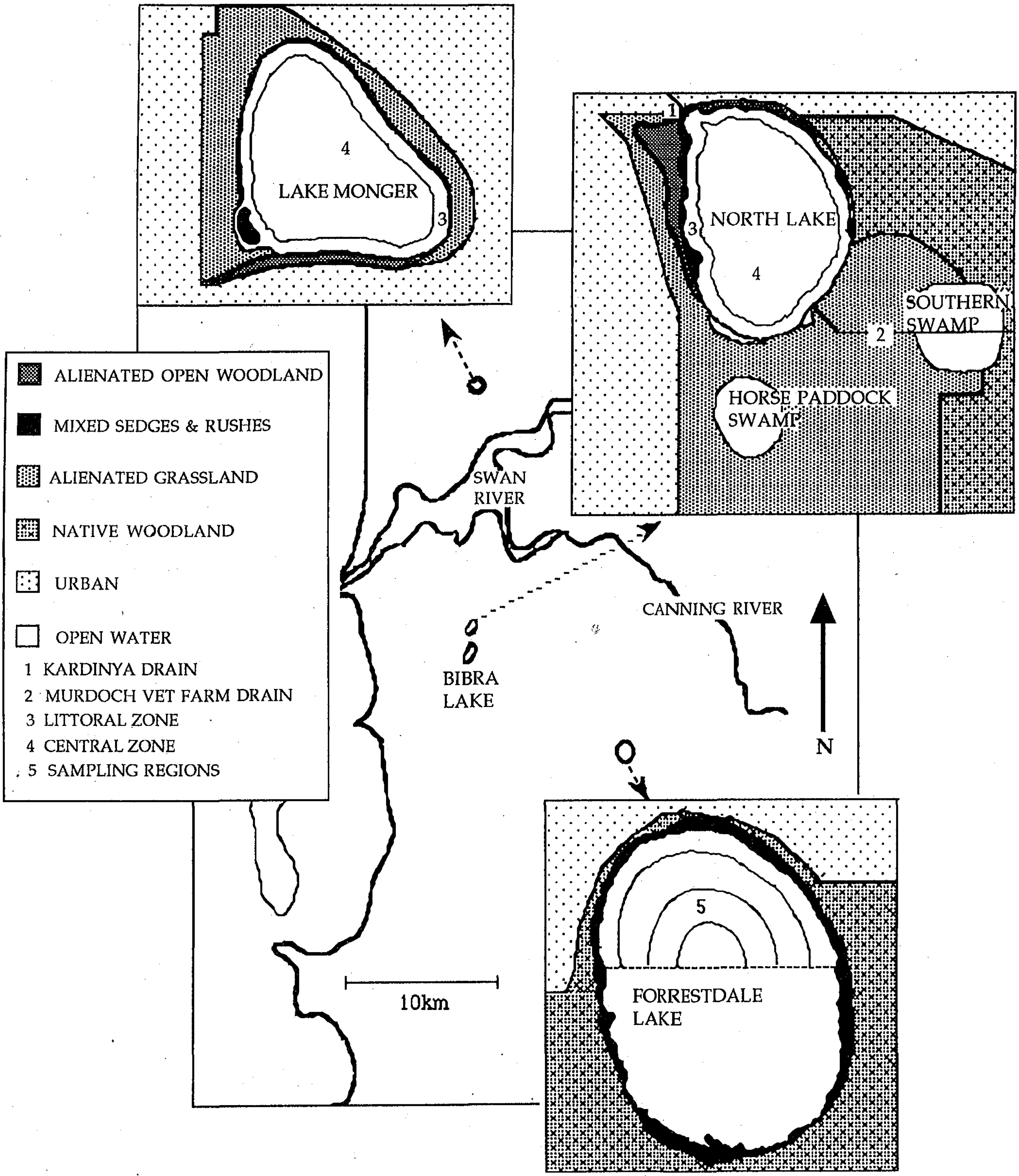

Figure 2.1 Map showing the locality of Lake Monger, North Lake and Forrestdale Lake on the Swan Coastal Plain

North Lake

On 76 occasions between October 1987 and March 1991, 20 cores of the sediment, containing chironomid larvae, were collected from randomly chosen locations within the littoral region of North Lake (Fig. 2.1). On most of these occasions an additional 10 cores were taken from the deeper (>1m) central region of the lake (Fig. 2.1).

During late spring, summer and early autumn of 1988/89, 1989/90 and 1990/91 submerged benthic emergence traps were used to determine adult emergence rates. Between six and fifteen traps were set in the littoral area for 24 hours on 86 occasions. Over the same time period four light traps were stationed at residential properties adjacent to North Lake, and were used to trap adult midges three nights per week between 7.30pm and 4.30am. Two of these were stationed on the west side of the lake and two on the north side and all four were approximately 50m from the shoreline. These light traps are described and illustrated in Davis *et al.* (1989).

Forrestdale Lake

This lake was sampled on 64 occasions between October 1987 and December 1990. On each occasion 20 core samples were collected from the northern half of the lake. This half of the lake was divided into four concentric semi-circles with the largest being the littoral region and the smallest the central area (Fig. 2.1). In these regions 1, 4, 6, and 9 cores were taken at random compass coordinates from within the deepest central area to the shallowest region respectively. This method ensured that the sampling locations were spread over the entire sampling area. An additional six cores were taken from within the stands of *Typha* (bullrushes) on the northern region of the lake while the stands were inundated.

Six emergence traps were used to trap adult chironomids at this lake during the spring and summer of 1988/89 and 1989/90. These were set in two groups of three located in open water near the *Typha* and were set for the 24 hour period prior to the collection of larval samples from the lake.

Methods common to both North and Forrestdale Lakes

Midge larvae were separated from the sediment by differential sorting using a saturated solution of calcium chloride, prior to counting and identification. This technique is described in Davis *et al.* (1990). Water samples were also collected on each sampling occasion and analysed for pH, conductivity, and the concentrations of total nitrogen, total phosphorus and chlorophyll-*a*(a measure of the abundance of phytoplankton). Maximum and minimum air and water temperatures were recorded by thermometers, which were either hung near the lake (for air temperature) or placed on the sediment in the littoral area of the lake (for water temperature). Lake water depth was measured from standard Water Authority depth gauges.

Lake Monger

Prior to April 1989 a non-quantitative method of sampling larval chironomid populations at Lake Monger was employed by the City of Perth. These data have been presented in previous reports and will not be repeated here. Since April 1989 a quantitative sampling technique (coring) has been employed. Ten to 14 core samples were taken from the littoral and/or central regions (Fig. 2.1) of Lake Monger on regular sampling occasions between April 1989 and March 1991. These samples were collected, sorted and identified by employees of the City of Perth.

Statistical analysis

Oneway analyses of variance with aposteriori Tukeys HSD tests were employed to investigate inter-annual differences in the spring/summer (including March) environmental data. The Tukeys HSD tests were performed at the 95% confidence level when the condition of homogeneity of variance (as measured by the Cochrans C test) is satisfied. Where heterogeneity of variance existed the results of the ANOVA were only considered to be significant at the 0.01 level of probability.

RESULTS AND DISCUSSION

Chironomid Populations

North Lake

The species composition and cumulative abundance of chironomid larvae in the littoral region of North Lake are presented in Figure 2.2, while the relative abundance of each species in this region (represented as percentage of total abundance) is given in Figure 2.3. Species richness for the whole lake was calculated for all sampling occasions and is plotted in Figure 2.4, The species composition and cumulative abundance of larvae in the central region is presented in Figure 2.5 and Figure 2.6 illustrates the changes in the density of each species in the littoral region.

Species composition of the larvae
Littoral region: The larvae of *P. nubifer* accounted for more than 70% of total larval density on most occasions and often constituted 80 to 99% of total abundance (Figs. 2.2 and 2.3). However, other species (particularly *Chironomus* spp.) were also relatively abundant on various occasions. For much of winter and spring 1988, *P. nubifer* accounted

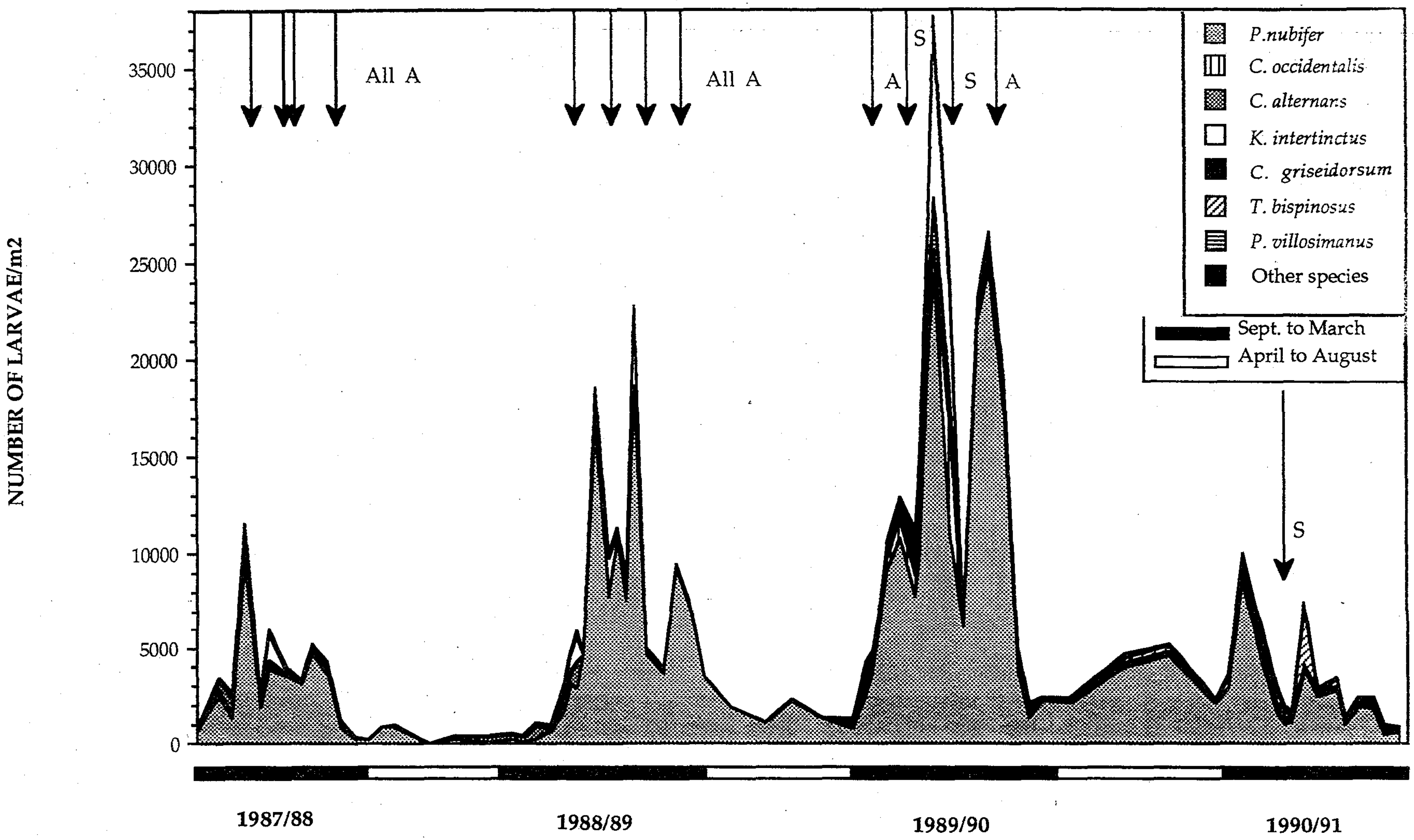

Figure 2.2. Changes in the abundance and species composition of larval Chironomidae in the littoral region of North Lake, between October 1987 and March 1991. The dates on which Abate (A) or Sumilarv (S) were applied to the lake are indicated.

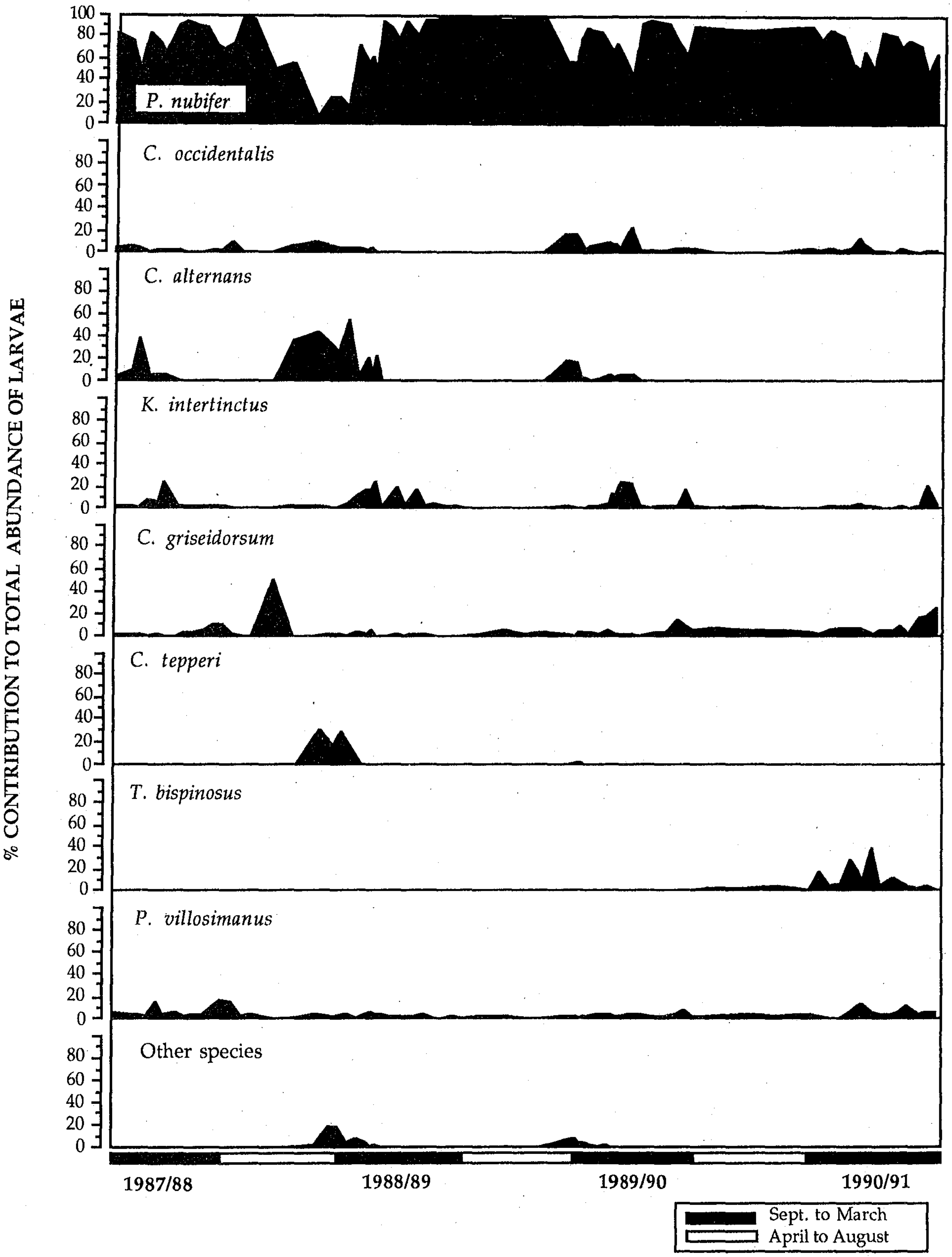

Figure 2.3. Changes in the percentage contribution of individual chironomid species to total larval abundance in the littoral region of North Lake, between October 1987 and March 1991.

for between 6 to 60% of larval density while *Chironomus alternans, Chironomus griseidorsum, Chironomus tepperi* and *Kiefferulus intertinctus* accounted for much of the remaining abundance (Figs. 2.2 and 2.3). During much of spring and early summer 1989 *Chironomus occidentalis, C. alternans* and *K. intertinctus* were relatively abundant, accounting for up to 30% of larval abundance. Finally, during the spring and summer of 1990/91 *Tanytarsus bispinosus, C. occidentalis, Procladius villosimanus* and *C. griseidorsum* were relatively abundant at various times while *P. nubifer* contributed less than 75% to total larval abundance.

Central region: *C. occidentalis, P. nubifer* and *P. villosimanus* were the most abundant species in the deeper (>1m) region (Fig. 2.5). During the 1987/88 season *P. nubifer* dominated, while *C. occidentalis* was the most abundant species during the following three years.

Species richness: The maximum species richness recorded on any one sampling occasion was eleven species and the minimum was two (Fig. 2.4). Species richness showed a highly seasonal trend with maximum number of species present in late winter and spring followed by a decline during summer and autumn. This decline occurred during the period when pesticides were used, although whether the use of pesticides contributed to the decline cannot be determined.

<u>Abundance of larvae</u>
Littoral region: Most species occurred at higher densities during the warmer months (late spring, summer and early autumn) than during the cooler months (Fig. 2.6), with the exception of some of the rarer species which were only present during winter.

During two of the four summers (1987/88 and 1990/91) *P. nubifer* rarely occurred at densities higher than 4 000 larvae/m^2 and reached maximum densities of 9500 and 8500/m^2 respectively. In the intervening two years *P. nubifer* density was regularly recorded at over 4 000/m^2 and exceeded 15 000/m^2 on more than one occasion. The mean density of *P. nubifer* (± standard error) recorded each spring and summer over the period 1987 to 1991 was as follows;

	spring and summer		**summer only**	
1987/88 :	3017 ± 723	(n=12)	3511 ± 900	(n=9)
1988/89 :	5695 ± 1264	(n=15)	8930 ± 1615	(n=10)
1989/90 :	10755 ± 1928	(n=17)	11016± 3261	(n=8)
1990/91 :	2425 ± 526	(n=15)	1838 ± 401	(n=8)

(Where n = number of sampling occasions)

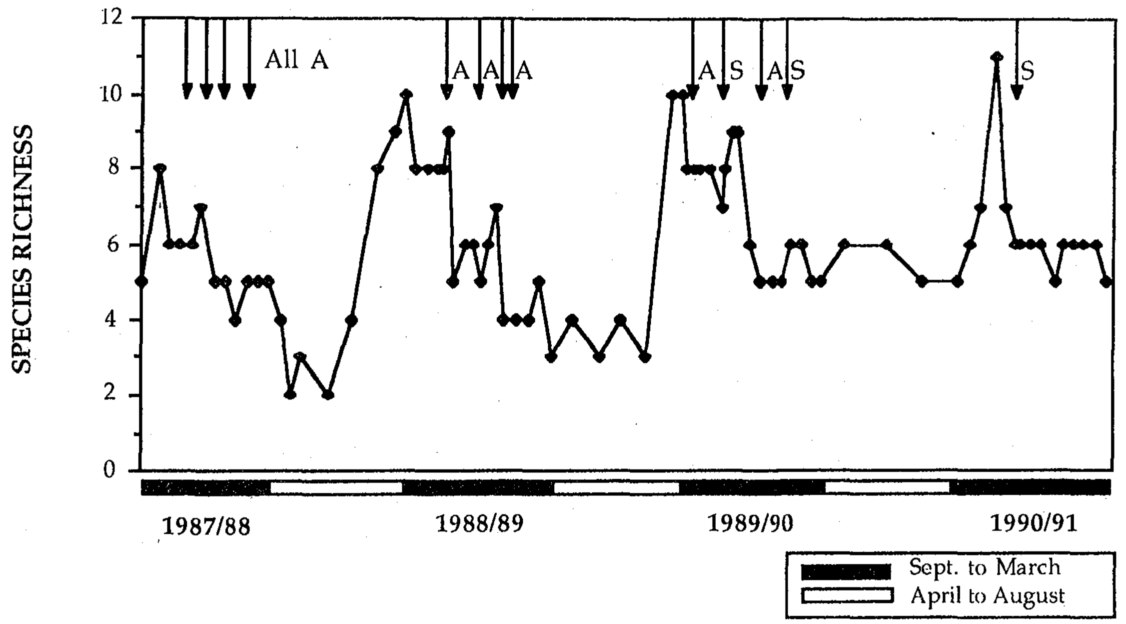

Figure 2.4 Changes in species richness of chironomid larvae at North Lake, between October 1987 and March 1991. Applications of Abate (A) or Sumilarv (S) are indicated.

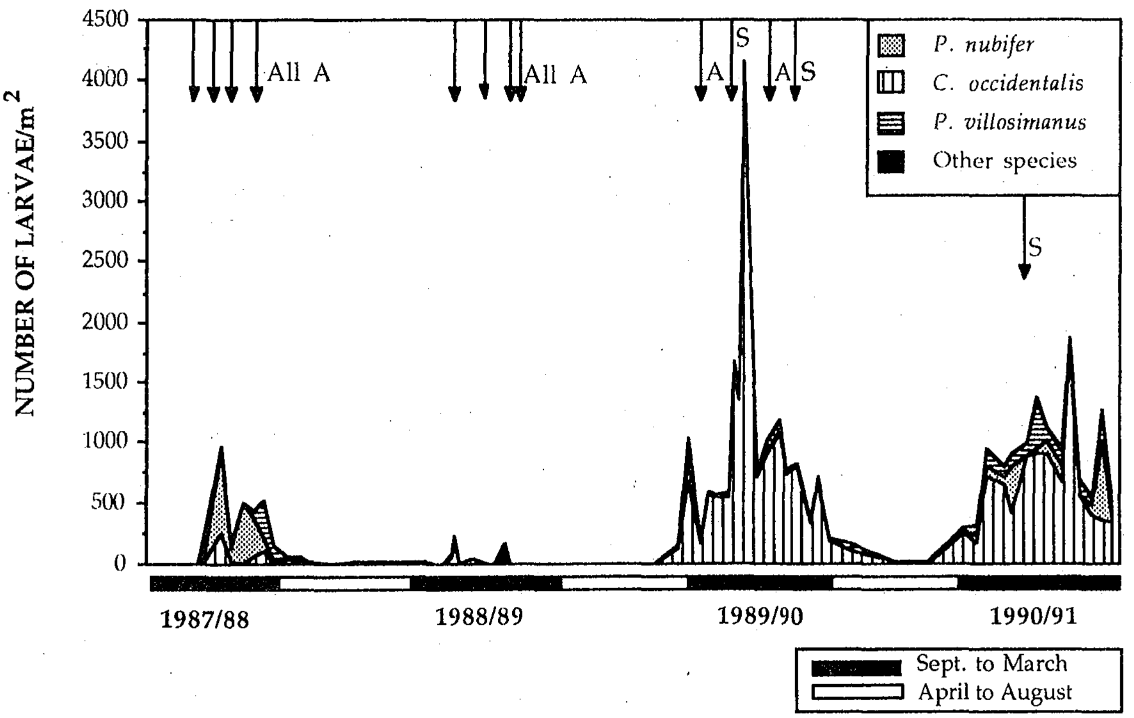

Figure 2.5. Changes in the abundance and species composition of chironomid larvae in the central region of North Lake, between October 1987 and March 1991.

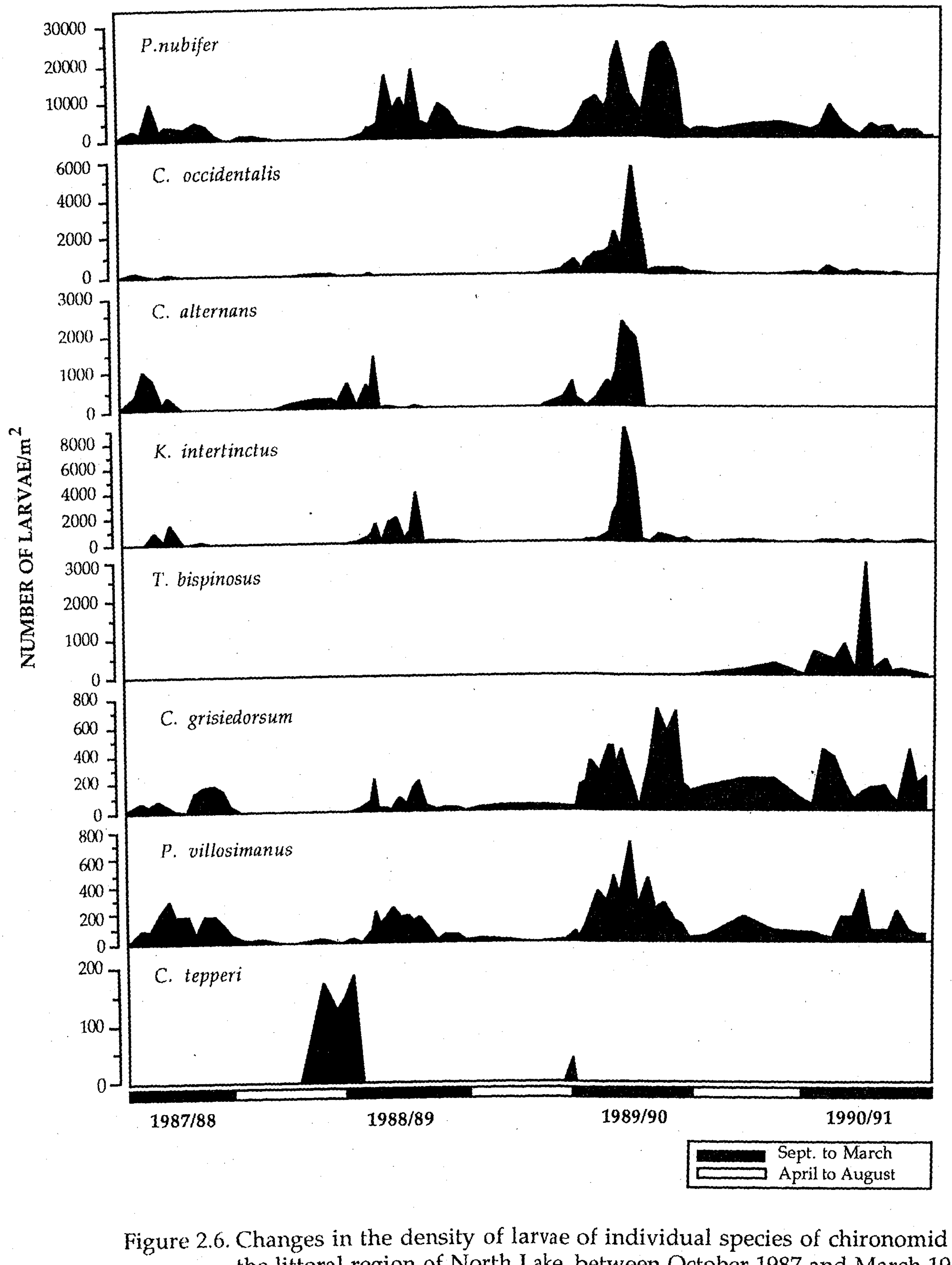

Figure 2.6. Changes in the density of larvae of individual species of chironomid in the littoral region of North Lake, between October 1987 and March 1991.

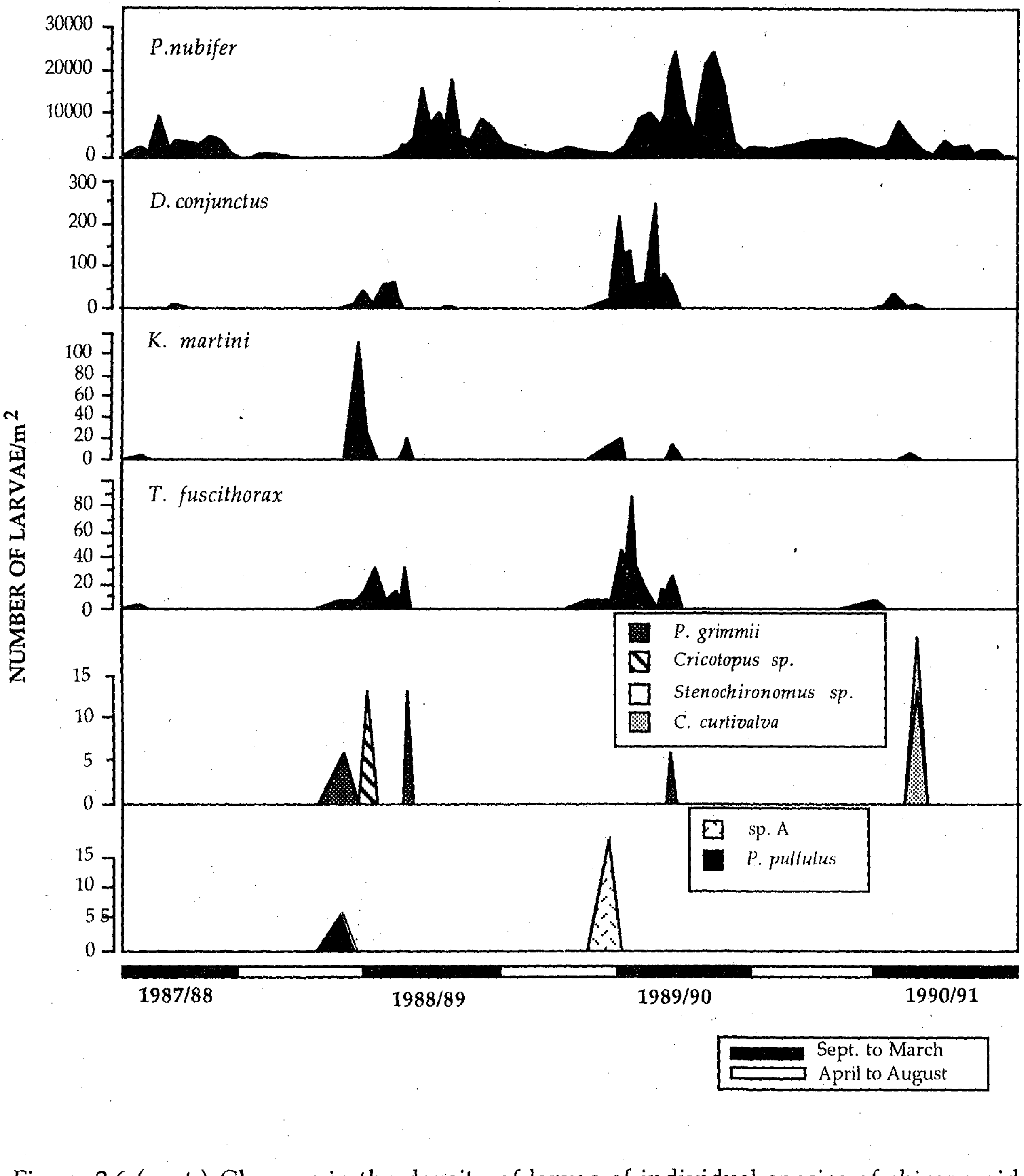

Figure 2.6 (cont.). Changes in the density of larvae of individual species of chironomid in the littoral region of North Lake, between October 1987 and March 1991.

During spring summer and autumn the density of *P. nubifer* fluctuated substantially (Fig. 2.6), some of the peaks appeared to be due to natural changes in population structure, while others were most likely due to the application of Abate or Sumilarv®.

In October 1990 the density of *P. nubifer* was recorded at over 8000 larvae/m^2 and so a decision was made to treat the littoral area of the lake with Sumilarv® in mid-November 1990. However, by early November the density of *P. nubifer* larvae and rate of adult emergence at the lake were already in decline and both larval density and adult emergence remained low for the rest of spring and summer. Thus, the effects of the November 1990 Sumilarv® application on the chironomid populations at North Lake are less clear than those of previous trials. The results of the 1989/90 trials suggested that this insecticide inhibited emergence for a minimum of three weeks but this was not accompanied by a decrease in larval density. Sumilarv® may have contributed to the low emergence in 1990/91 for several weeks, but is unlikely to have been the cause of the persistently low density of larvae.

Some species of chironomids were only abundant in the littoral region of North Lake for limited periods (Fig. 2.6). Whereas, *C. occidentialis* was present at densities above 500/m^2 only during 1989/90, *C. alternans* was recorded at over 500/m^2 in the first three summers but was not present at all during 1990/91. *T. bispinosus* was found only during 1990/91 where it contributed up to 39% of the total larval abundance (Figs. 2.3). *Chironomus tepperi* was most abundant (up to 185 larvae/m^2) during the winter of 1988 and only occurred once outside of this period (Fig. 2.6).

Central region: In the deeper region of North Lake chironomid larvae generally occurred at densities less than 2000 larvae/m^2, except in November 1989 when larval density was recorded at over 4000/m^2 (Fig. 2.5). *C. occidentalis* was generally the dominant species (up to 4053/m^2). *P. nubifer* and *P. villosimanus* occurred at maximum densities of 675 (January 1988) and 420 (December 1990) larvae/m^2 respectively.

Adult Emergence

The abundance and species composition of adult chironomids emerging from North Lake are shown in Figure 2.7, while Figure 2.8 presents a comparison of adult emergence, larval densities and light trap catches of *P. nubifer*.

The species composition of emerging adults was found to be different for each of the three years during which emergence was measured (1988/89 to 1990/91), and to generally reflect differences in the larval populations. During 1988/89 *P. nubifer* (with an average rate of emergence of 80 ± 28/m^2/night) and *K. intertinctus* (89 ± 37/m^2/night) tended to dominate both the larval and emergence trap samples, while during 1989/90 *P. nubifer* alone (188 ± 37/m^2/night) accounted for most of the emergence (Fig. 2.7). *K. intertinctus*

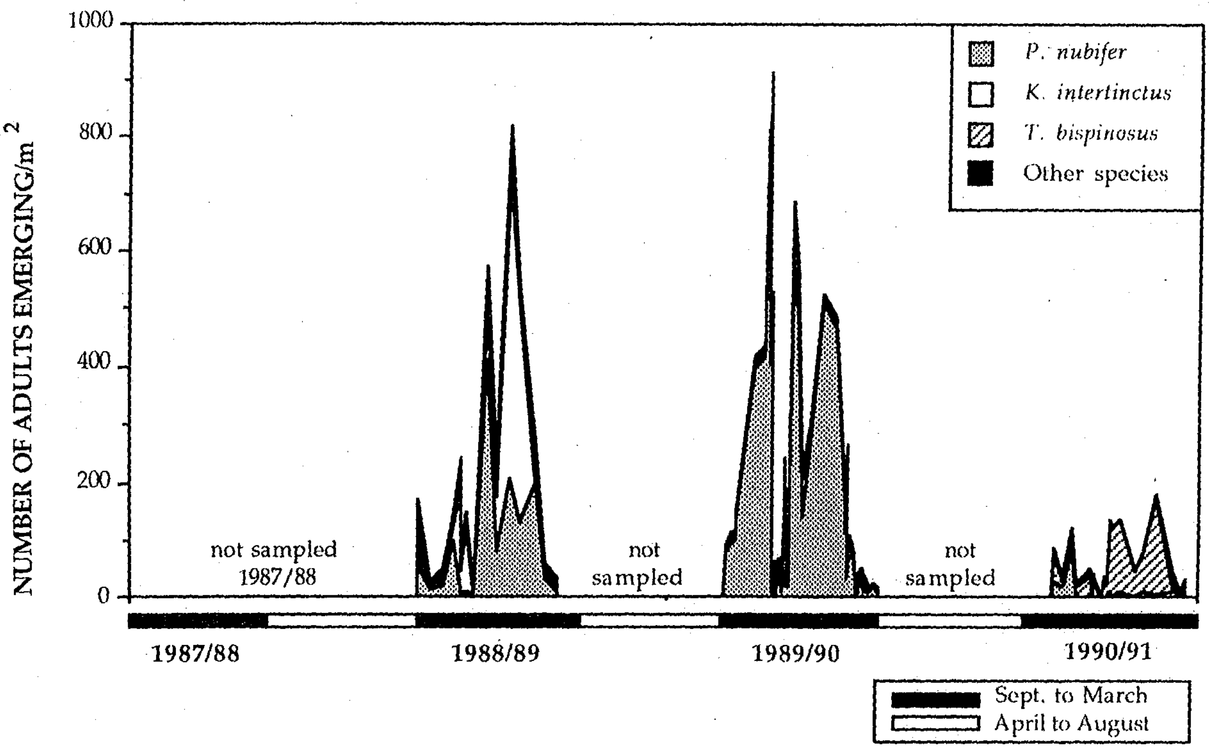

Figure 2.7. Changes in the species composition and abundance of adults emerging from the littoral region of North Lake between September 1988 and March 1990.

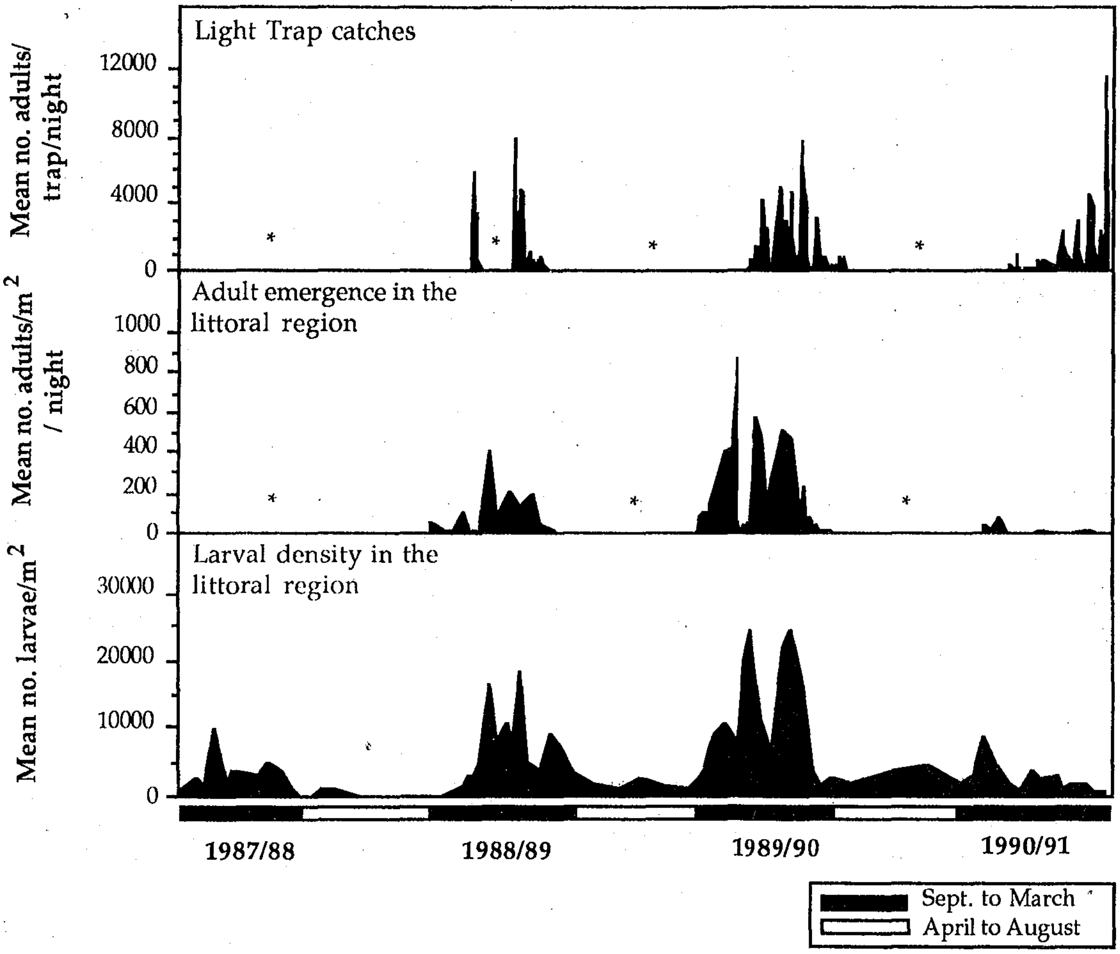

Figure 2.8. Changes in the density of larvae, rate of emergence and abundance of adults in light traps of *P. nubifer* at North Lake, October 1987 to March 1991. * = not sampled during this period.

larvae were present in high numbers in 1989/90 only after the application of Sumilarv® and so the emergence of this species might have been affected by this insecticide. During the final year (1990/91) the emergence of *P. nubifer* was generally lower (with an average rate of 8 ± $3/m^2$/night) than that of *T. bispinosus* (37 ± $9/m^2$/night). The rate of emergence for *T. bispinosus* is not unexpectedly high (an average of approximately 8% of larval density) but the rate of *P. nubifer* emergence is unusually low with an average of only 0.3% of larval density (Figure 2.8). In previous years, the daily rate of emergence of this species was approximately 4-5% of larval density. The reasons for the low emergence are not clear, although the lower phytoplankton growth, receding shoreline and the Sumilarv® application in November 1990 are possible explainations.

While the feeding biology of *P. nubifer* has not been properly studied, some preliminary examinations of gut contents during 1990 suggest at least part of the food intake of this species comprised whole live algal cells. Therefore, the lower abundance of phytoplankton in 1990/91 than in previous years may have affected the amount of food available to *P. nubifer*. In addition, the bare sand sediment in the littoral area that has supported high densities of *P. nubifer* larvae in the past was exposed by the receding shoreline in 1990/91. During this year the sand in the littoral region was covered by a layer of the fine silty mud that has been restricted to the central regions in previous years. This may have had a detrimental effect on the survival of *P. nubifer*. Although the rate of emergence was already declining in early November, the application of Sumilarv® in mid-November 1990 may also have contributed to the reduced rate of emergence.

<u>Light Traps</u>

Changes in the numbers of adult chironomids collected in the four light traps at North Lake between late spring 1990 and early autumn 1991 are presented in Figure 2.9. The light trap catches for the three years 1988/89 to 1990/91 are shown in Figure 2.8 together with larval densities and adult emergence. The effects of wind on light trap catches are discussed in Chapter Five.

The mean daily catch of adults per trap for the three years 1988/89 to 1990/91 were 1664 ± 284, 1326 ± 173 and 1125 ±166 respectively. There were no significant differences in the mean light trap catches between any of the years, despite the large differences in larval density and emergence at North Lake (Fig. 2.8). *P. nubifer* was the most abundant species in the light traps in all years.

In 1990 light trap catches of *P. nubifer* were lower during late spring and early summer than during late summer and early autumn (Fig. 2.9). By contrast, *P. nubifer* larval density was higher in spring and early summer than during late summer and early autumn (Fig. 2.2). Adult emergence of *P. nubifer* was low for the whole of spring and summer. These high light trap catches in late summer at North Lake may be partially

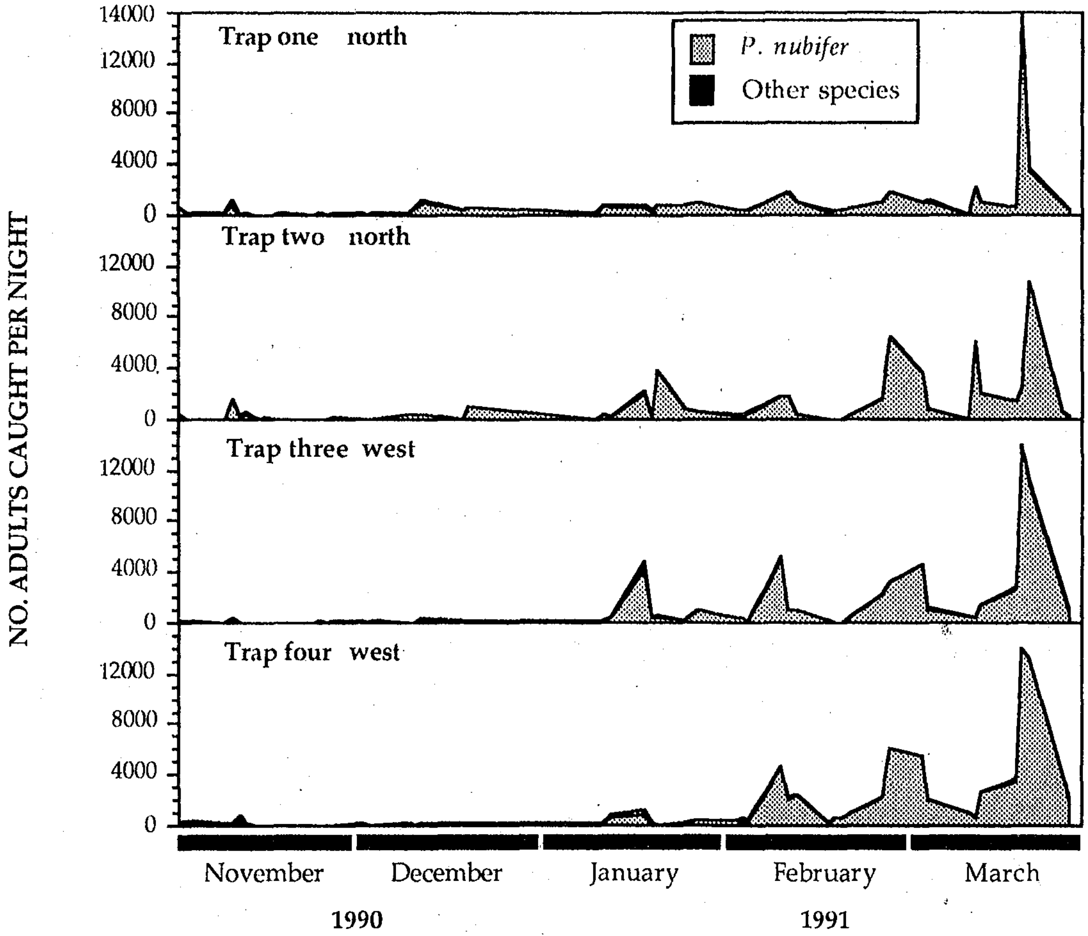

Figure 2.9. Changes in the number of adult chironomids caught by the four light traps stationed either to the north or west of North Lake, between 8th November 1990 and 21st March 1991.

explained by the very high larval densities of *P. nubifer* (up to 81 065 ± 9810 larvae/m^2) and rates of *P. nubifer* emergence (1537 ± 366 adults/m^2/night) recorded at Bibra Lake in the summer of 1990. It is therefore possible that adults from this lake also contributed to the light trap catches at North Lake. Bibra Lake is located approximately 200 metres south of North Lake. Southerly winds, which dominated during and after dusk, may have encouraged such northerly movement of the adult chironomids. If the adult flies originating at Bibra Lake did contribute to the light trap catches in 1990/91, then it is likely that Bibra Lake chironomids have contributed to nuisance problems encountered at North Lake in previous years. This suggests that midge control efforts at a single lake (whether short or long term) may not successfully reduce nuisance problems if nearby lakes are not also considered.

Forrestdale Lake

The species composition and cumulative abundance of larvae in the open water region of Forrestdale Lake is presented in Figure 2.10, while Figure 2.11 shows the relative abundance for the common species. Figure 2.12 presents the changes in chironomid larval species richness in the lake and Figure 2.13 presents the actual abundance of each species that occurred in the open water region of Forrestdale Lake.

<u>Species composition of larvae</u>
At Forrestdale Lake the density of *P. nubifer* contributed more than 50% to the total larval density on only eleven of the 64 occassions on which larval populations were sampled (Figs. 2.10 and 2.11). *C. occidentalis* made up over 30% of total abundance during most of the spring and summer of all four years, and often exceeded 50%. Similarly *C. alternans* and *D. conjunctus* together made up over 50% of total abundance for much of the spring and summer of 1988/89 (Fig. 2.10). *Tanytarsus fuscithorax* accounted for 40 to 100% of total abundance on four occasions (Fig. 2.10), on three of these conductivity was greater than 10 000 µS/cm. On one occasion in winter 1990 the density of this species was 57% of the total when conductivity was only 4400 µS/cm. However, this *T. fuscithorax* population may have been a remnant of the autumn population when it was the only species recorded and conductivity exceeded 19 000 µS/cm. *Procladius villosimanus* was the only other species to regularly contribute more than 20% to the total abundance of larvae and was most abundant immediately after the three Abate treatments. This species and *C. occidentalis* were the only two species to have been recorded on more sampling occasions than *P. nubifer*.

Species richness: Sixteen species were recorded during the four years of sampling at Forrestdale Lake. Three of these (*Paralimnophyes pullulus* and two *Cricotopus* spp.) were only recorded in October 1990 from within stands of the *Typha orientalis*. The maximum species richness in the open water region on any one sampling occasion was nine species and the minimum was zero, awhen the lake was almost dry (Fig. 2.12). Species richness

DENSITY OF CHIRONOMID LARVAE IN THE OPEN WATER REGION OF FORRESTDALE LAKE 1987 TO 1990

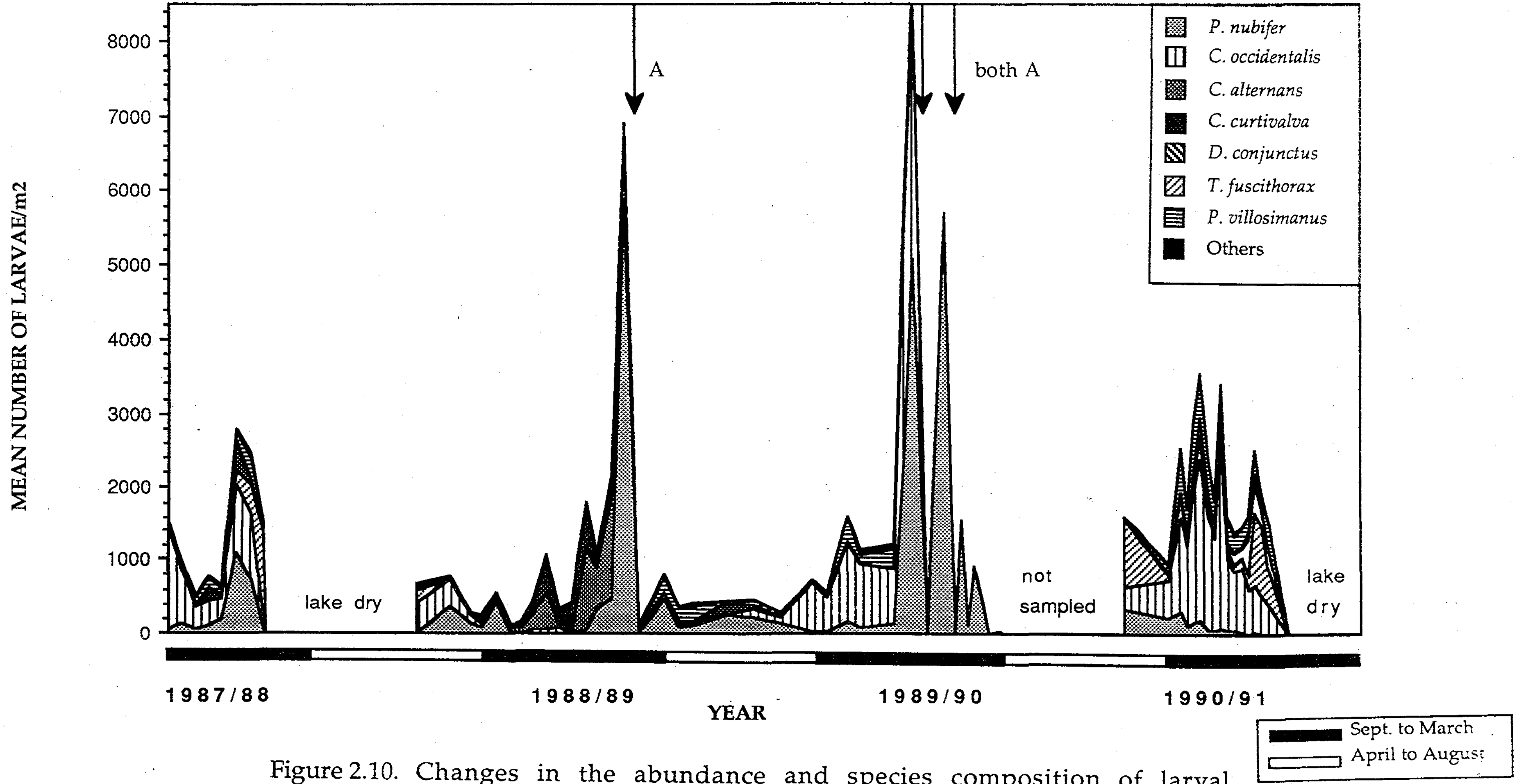

Figure 2.10. Changes in the abundance and species composition of larval Chironomidae in the littoral region of Forrestdale Lake, between October 1987 and March 1991. The dates on which Abate (A) was applied to the lake are indicated.

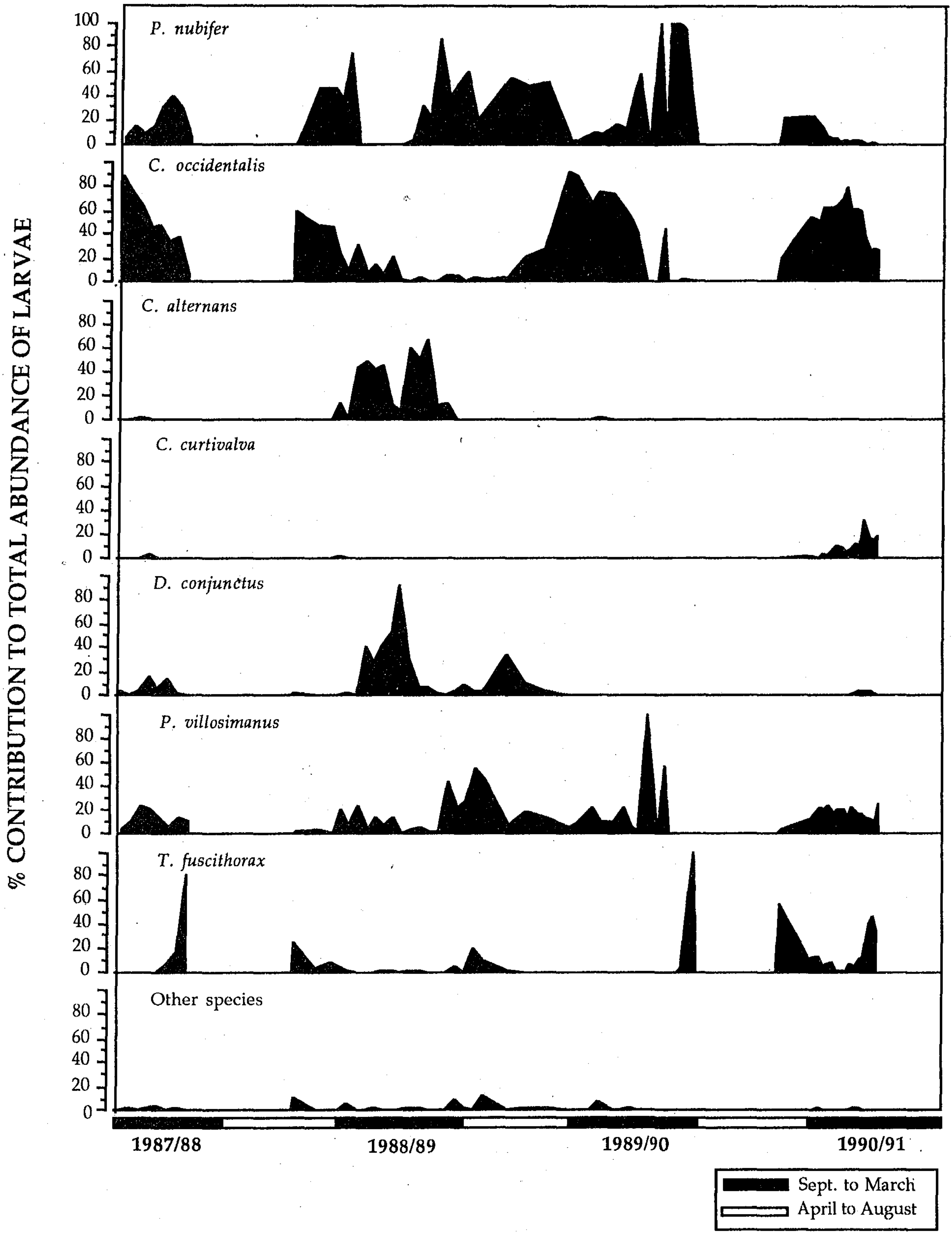

Figure 2.11. Changes in the percentage contribution of individual chironomid species to total larval density in the open water region of Forrestdale Lake between October 1987 and December 1990.

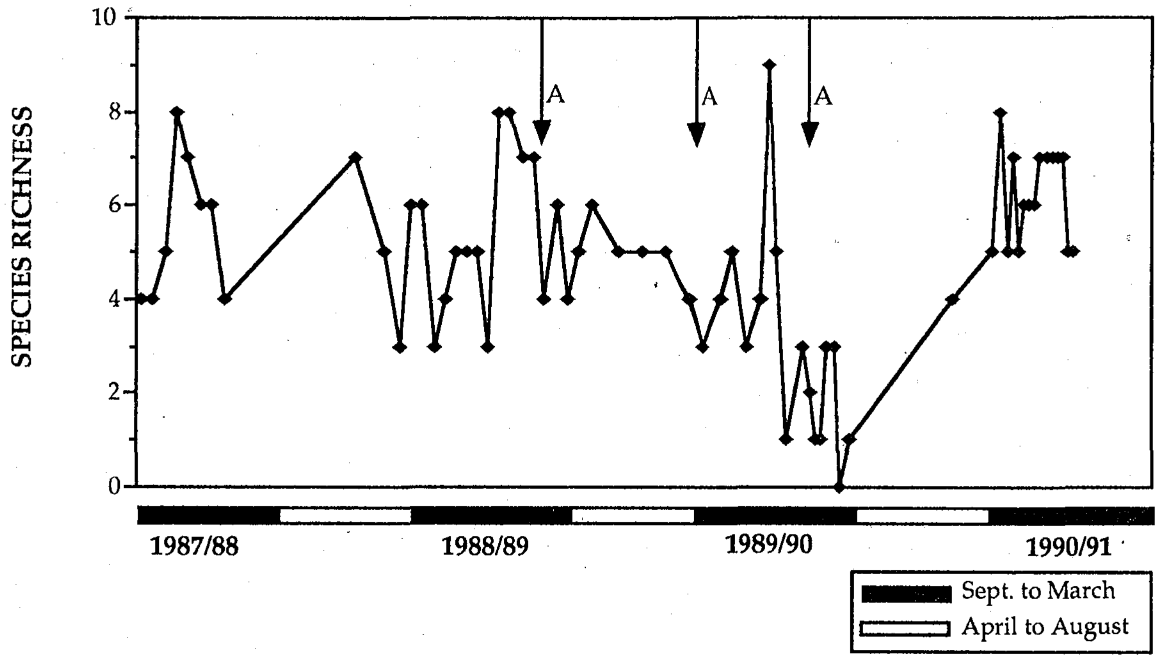

Figure 2.12. Changes in species richness of chironomid larvae at North Lake, between October 1987 and March 1991. Applications of Abate (A) are indicated.

tended to be highest in late spring or early summer before the first peak in *P. nubifer* abundance and then declined during summer and early autumn.

Abundance of larvae

Open water region: Midge nuisance problems were less severe during the spring and summer of 1990 than during the preceding three years. In the first three years a 5 to 33 fold increase in the density of *P. nubifer* over a three week period was recorded, resulting in densities of 1000 to 6000/m^2 (Figs. 2.10 and 2.13). These high densities of *P. nubifer* larvae led to severe nuisance problems with adult midges in the nearby urban area. In two of these years (1988/89 and 1989/90) Abate was applied to the lake, and populations of *P. nubifer* were reduced to less than 40/m^2 (Figs. 2.10). In 1989/90 a second increase from 0 to over 5500 larvae/m^2 was recorded five weeks after the first peak and a second Abate treatment was applied, which reduced *P. nubifer* density to zero. After this second treatment *P. nubifer* increased again to 900 larvae/m^2 but this rise was curtailed by increasing conductivity. In 1990/91 a rapid increase in *P. nubifer* density did not occur and numbers declined from just over 300/m^2 in early spring to less than 100/m^2 for the whole of late spring and summer until the lake dried out.

The mean densities of *P. nubifer* larvae for spring only (Sep. to Nov.), spring and summer combined and for summer only (Dec. to Mar.), excluding densities recorded on the first occasion after an application of Abate, were as follows;

	spring only		spring and summer		summer only	
1987/88 :	82 ± 35	(n=3)	303 ± 138	(n=8)	435 ± 204	(n=5)
1988/89 :	92 ± 63	(n=6)	507 ± 360	(n=12)	1118 ± 953	(n=6)
1989/90 :	117 ± 32	(n=5)	987 ± 432	(n=17)	2314 ± 844	(n=8)
1990/91 :	108 ± 27	(n=12)	90 ± 24	(n=15)	14.3 ± 9	(n=3)

(n = number of sampling dates)

These results show that while there were very few differences between the densities of *P. nubifer* in spring, substantial differences were recorded between densities recorded during the summer months of different years. The low density of *P. nubifer* larvae recorded during the summer of 1990 appears to be mainly due to the earlier drying out of the lake in late December/early January.

At various times, other species were also abundant in the open water region of this lake. Either *C. alternans* or *C. occidentalis* were generally more abundant than *P. nubifer* during spring and early summer, often exceeding 500/m^2 and occasionally 1000/m^2 (Fig. 2.10 and 2.13). These species either persisted until just before the lake dried, or were greatly reduced in number after pesticide applications. In 1990 *C. occidentalis* was more abundant

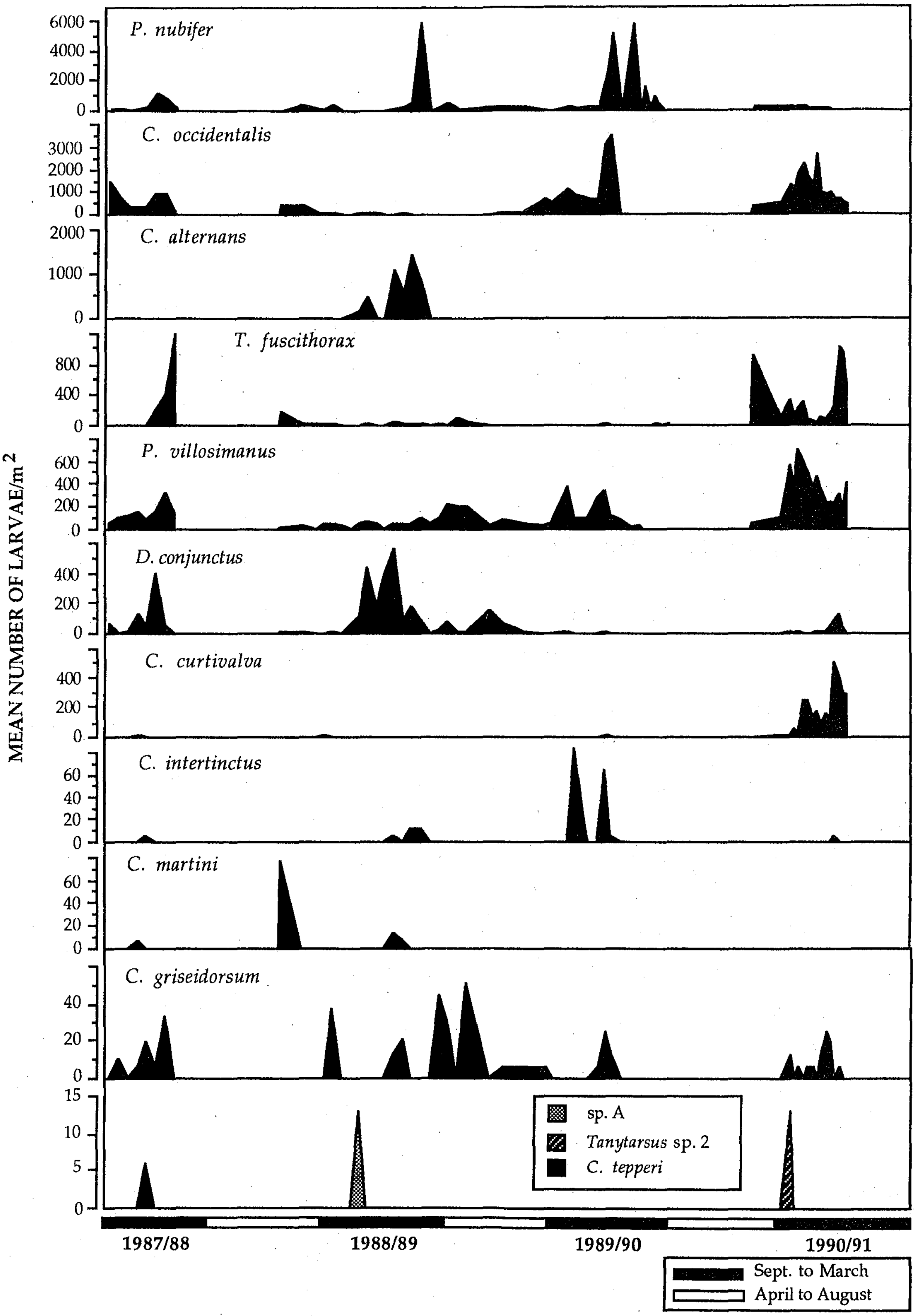

Figure 2.13. Changes in the density of larvae of individual species of chironomid at Forrestdale Lake, between October 1987 and December 1990.

than *P. nubifer* for the whole of spring and summer until the lake dried. *Tanytarsus fuscithorax* was the only other species to exceed 1000 larvae/m^2 (Fig. 2.13) and attained these densities when conductivity was, or had recently been, high.

Larval abundance within stands of the bullrush (Typha orientalis): During the three years in which samples were collected from the northern stands of *Typha orientalis* at Forrestdale Lake total larval density exceeded 1000/m^2 on only three occasions. The most abundant species was usually *C. alternans*, with some *C. occidentalis* and *D. conjunctus* present. *P. nubifer* did not exceed 100/m^2 in this region and by the time nuisance problems were at a maximum the *Typha* region was usually either too warm for larval survival or dry. This area did not appear to contribute to nuisance problems at the lake.

Adult Emergence

Emergence traps were set at Forrestdale Lake during the spring and summer 1988/89 and 1989/90. Adult chironomids were rarely caught by these traps. This may have been due to the generally low and patchy larval densities recorded for much of the time and the small number of traps set compared to the size of the lake. The use of a larger number of traps distributed over a wider area of the lake may be more appropriate for any future studies.

Lake Monger

Larval abundance

Prior to April 1989 a non-quantitative sampling technique was employed at Lake Monger in the central area only. This data showed that *C. occidentalis*, was the dominant species in the central region, and that this species increased during spring and summer and declined during autumn and winter (Davis *et al.* 1988 and 1989). *P. nubifer* is one of the main nuisance species at this lake but the larvae are most prevalent in the littoral area and were rarely recorded in the central samples. After April 1989 a standard corer was used to take quantitative samples and in January 1990 the littoral area was incorporated into the sampling program.

The dominant species inhabiting the littoral area of Lake Monger in the summer of 1989/90 was *P. nubifer* (Fig. 2.14). Over 17 000 larvae/m^2 were recorded during January 1990 and severe nuisance problems were reported. The density of *C. occidentalis* larvae varied between 30 and 1200 larvae/m^2 during this time. Low densities of both species were recorded during the winter. In the following spring and summer (1990/91) the density of *C. occidentalis* larvae was similar to previous years (30 to 900 larvae/m^2) while the density of *P. nubifer* did not exceed 30 larvae/m^2. No nuisance problems were reported from Lake Monger in 1990/91.

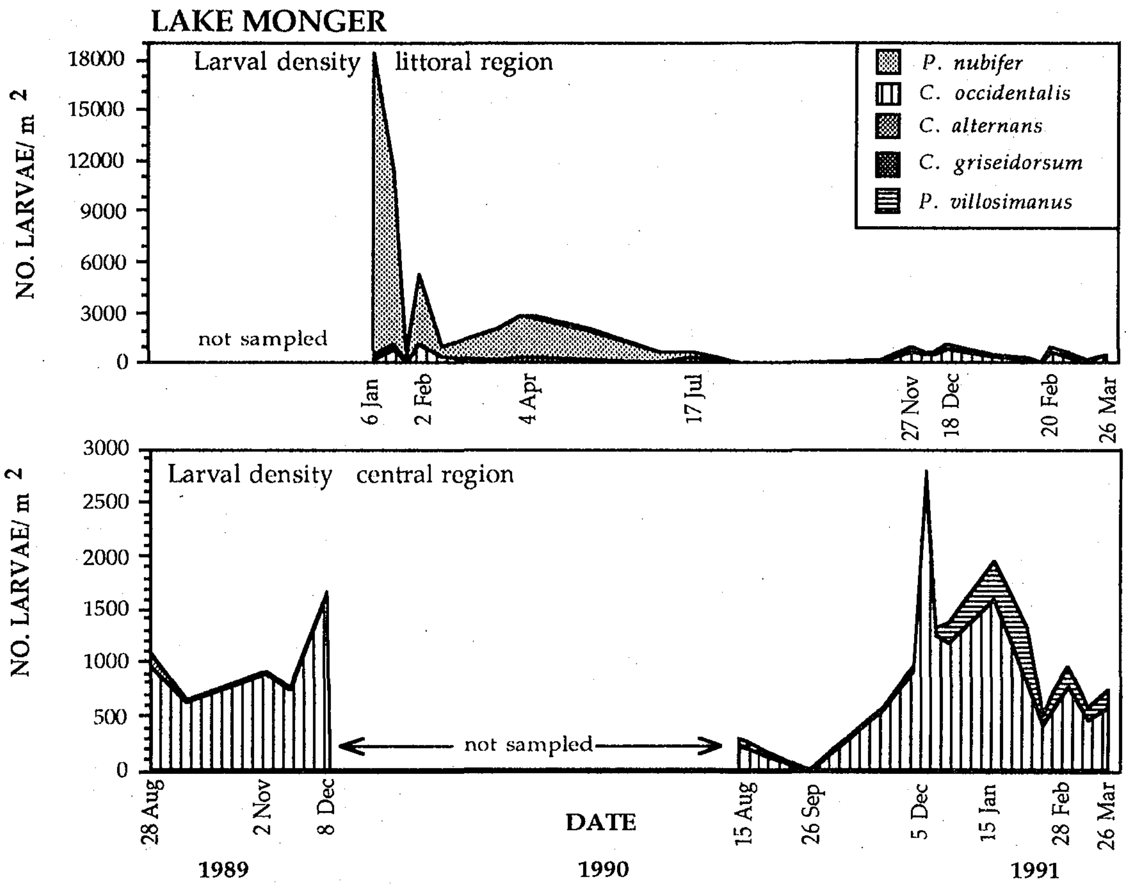

Figure 2.14. Changes in the density and species composition of larvae in the littoral and central regions of Lake Monger between April 1989 and March 1991.

The most abundant species in the central region of Lake Monger was *C. occidentalis* which varied between 500 and 2500 larvae/m^2 during the warmer months of both 1989/90 and 1990/91.

Light traps

In 1989/90 eleven light traps were installed on the perimeter of Lake Monger for the duel purpose of footpath illumination and control of adult chironomids. In 1990/91 an additional 12 light traps were installed. These are described and illustrated in the 1990 midge research report (Davis *et al.* 1990). The mean number of adult chironomids caught in 1989/90 ranged from 261 ± 155 to 128 000 ± 22 042 per trap per night. The adults caught by these traps were mainly *Polypedilum nubifer* which was also the most abundant larvae in the lake. These catches were estimated to represent up to 11% of adult *P. nubifer* emerging from the lake. It was concluded from this that such traps may be useful in an integrated chironomid control program at this lake.

In 1990/91 the mean light trap catch was much lower than during the previous year, ranging from 21 ± 5 to 1813 ± 4654 adults per trap per night, most of which were *Chironomus occidentalis*. The lower light trap catches in 1990/91 than in 1989/90 most likely reflect the lower abundance of larvae in 1990/91. *C. occidentalis* was also the most abundant species of larvae in the lake in 1990/91, however, the traps were emptied weekly to fortnightly during 1990/91 and so estimates of the ratio of light trap catches to emergence is not possible. To properly determine the effectiveness of these light traps it will be neccessary to set them overnight on the same night as adult emergence was measured.

Perth weather data

Weather records for Perth city are presented in Figure 2.15. Rainfall in Perth is highly seasonal, with most rain falling between late autumn and early spring. Of the four years in which chironomids were monitored, 1988 had the highest annual rainfall (909.4mm) which was 40mm above the average. Rainfall was over 100mm below the average on the three remaining years. Air temperature varied only marginally between years.

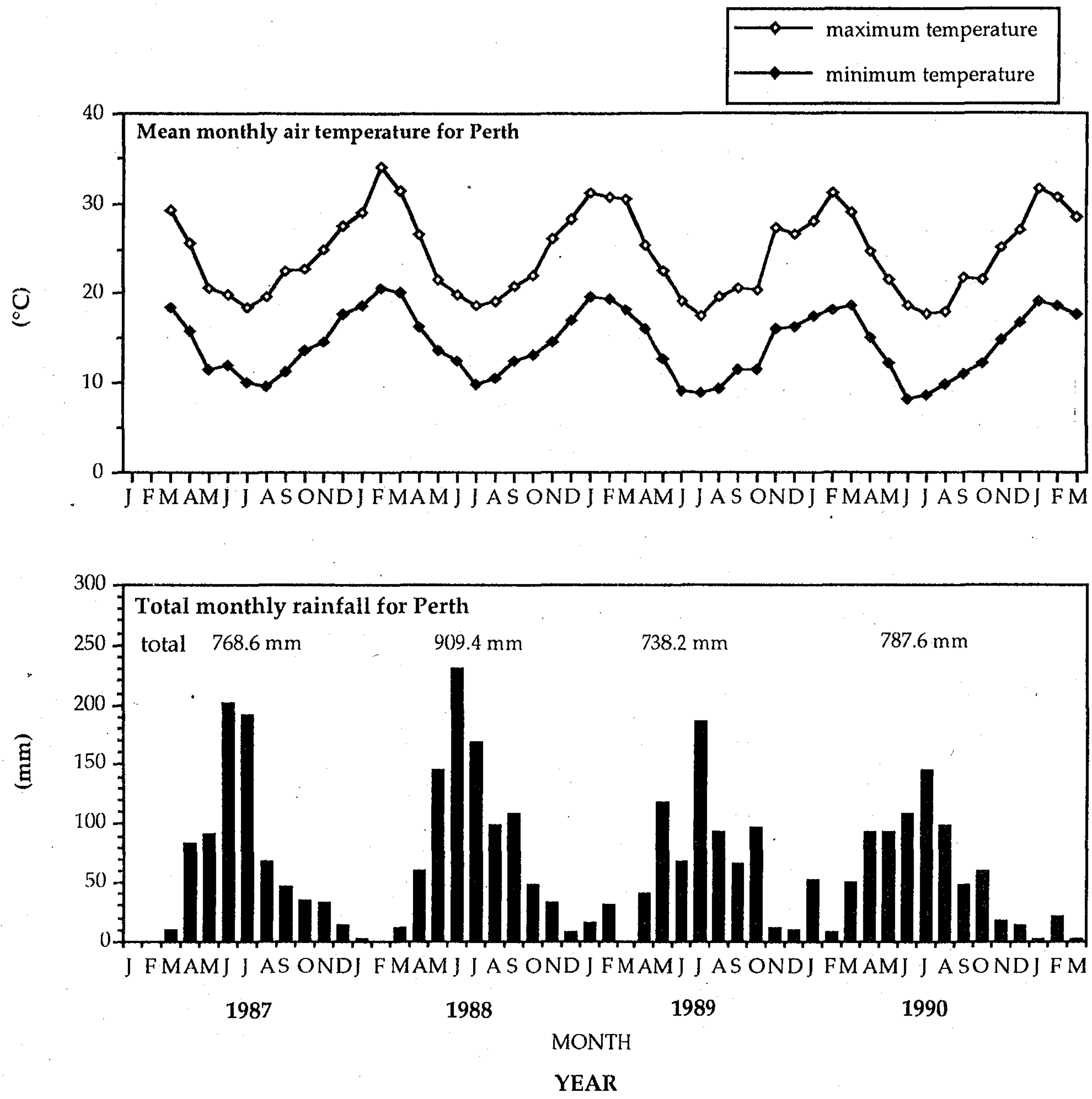

Figure 2.15 Average monthly minimum and maximum air temperature and total monthly rainfall measured in Perth City between January 1987 and March 1991.

Monitoring of the lake environment

These abbreviations are used in the following text; total phosphorus concentration [TP], total nitrogen concentration [TN], and chlorophyll-*a* concentration [chl-*a*].

North Lake

Changes in various environmental parameters measured at North Lake over the previous four to six years are given in Figures 2.16 to 2.19. These figures also provide the spring and summer mean values of selected variables. The figures for pH, conductivity, nutrients and chlorophyll-*a* concentrations also include the data of Davis and Rolls (1987) and Rolls (1989) for the period May 1985 to May 1987. This data, combined with that collected during the midge monitoring create a six year data set. Records prior to 1985 are sparse and of unknown reliability. Whereas the 1987 to 1991 values were calculated from weekly to fortnightly data, the mean spring and summer values for 1985/86 and 1986/87 were calculated from monthly data. Spring and summer is taken to include the period September to March, as this was the period when chironomid larvae are most abundant in the lake.

Lake Depth

The depth of North Lake has increased substantially in the past decade as a result of nearby suburban developments which have caused increased surface flow and a rise in the groundwater table (Bayley *et al.* 1989). In the years 1987 to 1989, the depth increased during winter and early spring to a maximum of 2.5 to 3.0 metres before falling to 1.5 to 2.2 metres by late summer (Fig. 2.16). In 1987 rainfall in the catchment of the Murdoch veterinary farm drain was the major source of water to the lake (Bayley *et al.* 1989). After the diversion of this drain in autumn 1990, the winter maximum was only 2.1 metres and summer minimum depth was less than 1.3m in 1990/91. The main consequences of annual changes in lake depth are changes in the concentration of dissolved chemicals such as salts and nutrients, altered buffering capacity of the lake to thermal changes, and changes to the site characteristics of the littoral region such as sediment type and organic content.

Conductivity

While obvious seasonal changes in conductivity occurred each year as a result of variations in lake depth (Fig. 2.16), there were no significant differences between conductivities in the spring and summer periods of 1987/88 to 1990/91 (P=2.15). Variations of conductivity are unlikely to have had any major effects upon invertebrates or algae within the range recorded (537 to 1157 µS/cm).

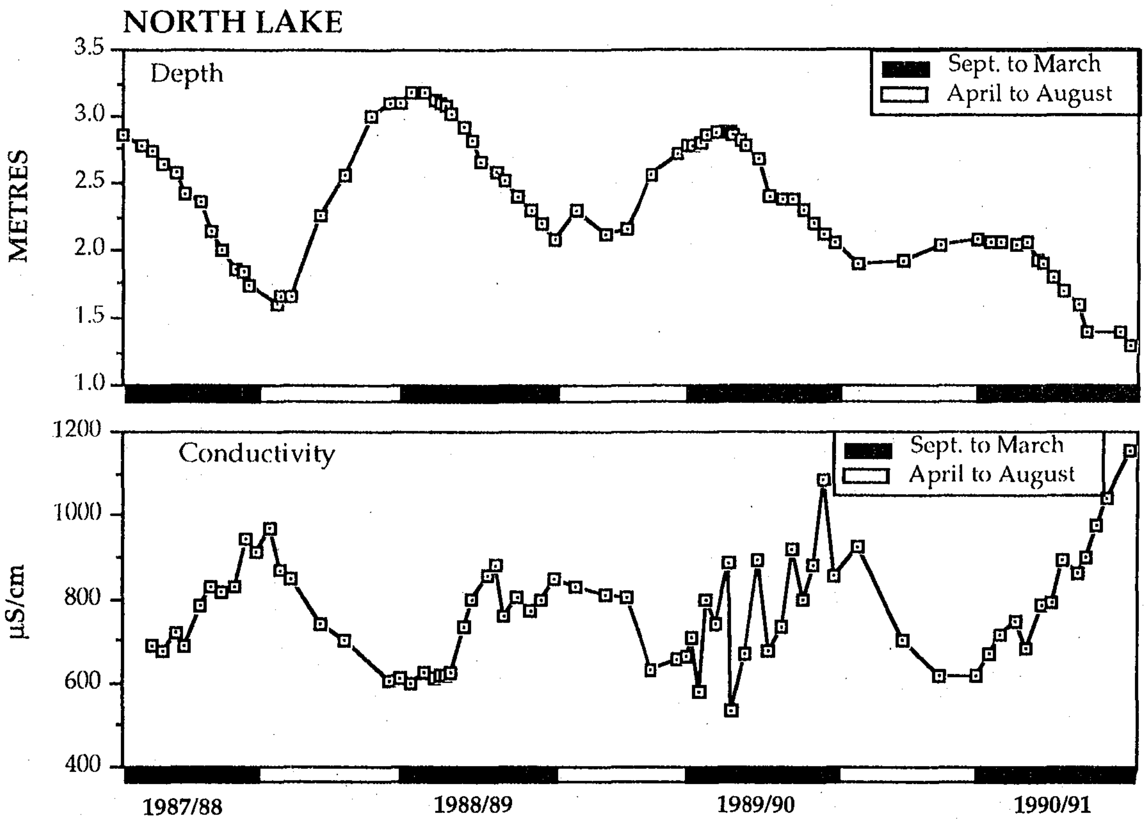

Figure 2.16. Changes in lake depth and conductivity for October 1987 to March 1991 in North Lake.

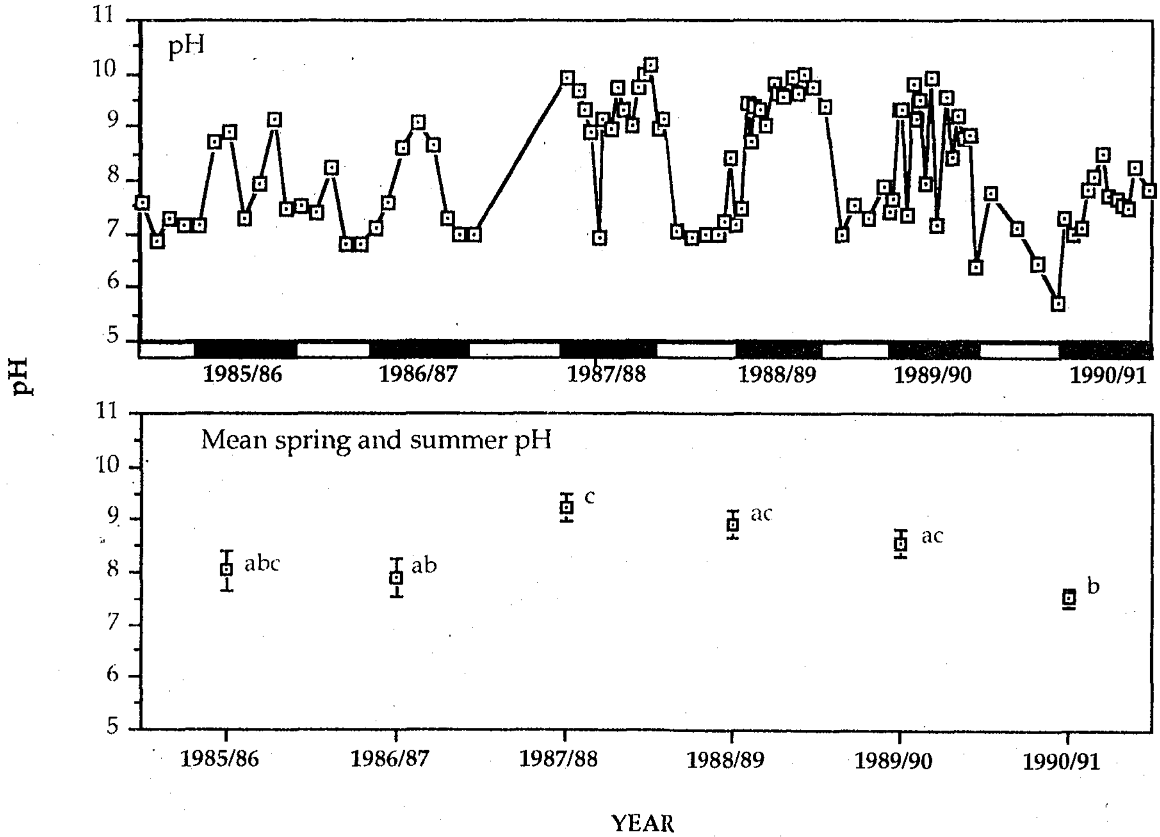

Figure 2.17. Changes in pH for May 1985 to March 1991, with spring and summer means (± standard error) at North Lake. Mean values followed by the same letter are not significantly different.

pH
In the six years 1985/86 to 1990/91 the pH of North Lake varied between 5.7 and 10.2 (Fig. 2.17). A major determinant of pH is the photosynthetic activity of algae, which uses up free carbon dioxide and leads to the removal H^+ ions from the water, thus raising pH. The increased photosynthetic activity during spring and summer is thus reflected by the higher pH during this period than during the cooler months. In 1990/91 the spring and summer pH was significantly lower than during the same months of the previous three years. The only other significant difference was the higher spring/summer pH of 1986/87 compared to 1987/88.

Water Temperature
The minimum water temperature (Fig. 2.18) in the littoral area varied between 9 and 20 °C in autumn and winter and 11 to 23 °C during spring and summer. Maximum temperature (Fig. 2.18) varied between 16 and 28 °C during the autumn/winter and 20 and 34 °C during spring/summer. Minimum and maximum water temperatures represent the extremes of temperature during the period between sampling dates (usually a week to a month). The midway point (median temperature) between these two extremes is probably a more representative measure of average temperatures during these periods (Fig. 2.18). The average median temperatures (Fig. 2.18) for each of the four years were as follows; 25 ± 0.5 °C (1987/88), 22.8 ± 0.7 °C (1988/89), 22.3 ± 0.7 °C (1989/90) and 22.5 ± 1.0 °C (1990/91). Neither minimum nor median temperatures for the spring/summer months were significantly different between the four years. Maximum spring and summer temperatures were significantly higher in 1987/88 than during the following two years.

Temperature is important to lake functioning because of its effect upon the growth, development and reproduction of algae, macrophytes, bacteria and macroinvertebrates. By affecting these biological components and rates of chemical reactions, temperature can also affect such processes as sediment nutrient release and oxygen depletion.

Nutrients and chlorophyll-*a*
In the autumn of 1990, the Murdoch University veterinary farm drain was diverted to an on-campus soak in order to meet the requirements of the Environmental Protection Authority. This drain had been found to contribute a large proportion of the total external nutrient load to North Lake (Bayley *et al.* 1989), and the diversion was seen as an important step towards the improvement of water quality at this lake. Only a single years data has been collected since the diversion of the drain so discussions as to the success of this action are still tentative and several further years of monitoring will be required before conclusions can be drawn. However, we have attempted to provide explanations for the observed changes in nutrients (including post-drain diversion) over the past six years based on available information.

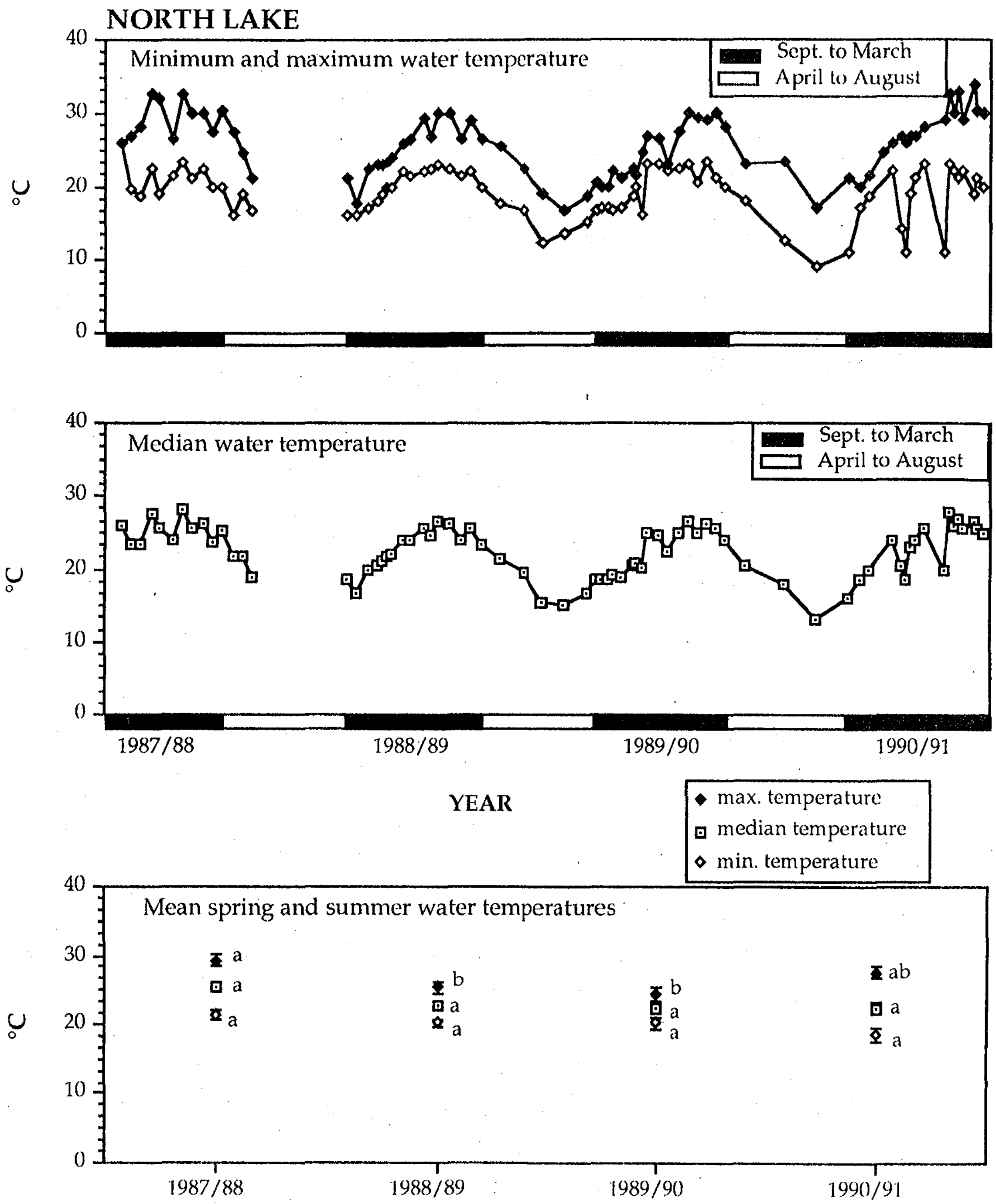

Figure 2.18. Changes in minimum, maximum and median water temperature, with spring and summer means (± standard errors), for October 1987 to March 1991, at North Lake. Mean values of the same variable followed by the same letters are not significantly different.

Phosphorus: Total phosphorus concentrations at North Lake (Fig. 2.19), ranged from 30 to 894 μg/l during the six years of monitoring (1985 - 1991). The highest [TP] (> 400 μg/L) were recorded in Spring 1987 and Autumn 1988. [TP] remained below 320 μg/L on all other occasions. Despite most of the phosphorus entering the lake via groundwater and drains in autumn and winter, [TP] tended to be lower in the cooler months than in spring and summer. This may be a result of seasonally varying sediment nutrient release (Keeney 1973), the continuous suspension of large amounts of nutrients in the algal cells of spring and summer algal blooms (Bayley *et al.* 1989) and the concentrating effect of lower summer water depths.

The main source of nutrients to North Lake in 1987/88 was the Murdoch University veterinary farm drain (Fig. 2.1) (Bayley *et al.* 1989). This drain contributed 73% of the external phosphorus inputs to North Lake in that year, the majority of which came from fertilizer applied to the farmland.

Between the spring and summer of 1985/86 and the same period of 1986/87 a significant increase in [TP] was recorded (from 167 ± 19 to 279 ± 11 μg/L). The possible reasons will be discussed later. [TP] then remained high (between 200 and 250 μg/L) for the following four spring and summer periods. However, only the 1986/87 [TP] was found to be significantly higher ($P<0.05$) than during 1985/86. In the spring and summer of 1990/91 [TP] decreased to 145 ± 16 μg/L following the diversion of the veterinary farm drain. This [TP] was significantly lower than in the four previous years.

Nitrogen: Total nitrogen concentration [TN] at North Lake (Fig. 2.19) ranged from 415 to 12672 μg/L between 1985 and 1991, but was usually between 1000 and 7000 μg/L. There were no apparent seasonal trends in [TN], and no significant differences ($P = 0.836$) were found between different years. The mean [TN] for the six spring and summer periods were 3776 ± 782 μg/l (1985/86), 2162 ± 542 μg/L (1986/87), 3990 ± 657 μg/L (1987/88), 3066 ± 310 μg/L (1988/89), 3911 ± 682 μg/L (1989/90) and 3012 ± 348 μg/L (1990/91). The veterinary farm drain was found to be the major source (60%) of external nitrogen input to North Lake in 1987/88 (Bayley *et al.* 1989). Although this drain was diverted in 1990, no significant reduction in [TN] was recorded in the water in the following spring and summer.

Chlorophyll-a: The concentration of chlorophyll-*a* in North Lake varied between 2 and 242 μg/L during the six years of monitoring (Fig. 2.20). The periods of highest [chl-*a*] were the winter, spring and summer months of 1988/89 and the spring and summer of 1989/90. While algal blooms were observed in all seasons, the actual timing varied between years and distinct winter blooms were only observed in 1986 and 1988.

The mean spring and summer [chl-*a*] for the six years 1985 to 1991 are; 37 ± 11 μg/L, 60 ± 19 μg/L, 75.1 ± 9.3 μg/L, 132.7 ± 16.8 μg/L, 135.1 ± 16.4 μg/L and 61.5 ± 15.1 μg/L

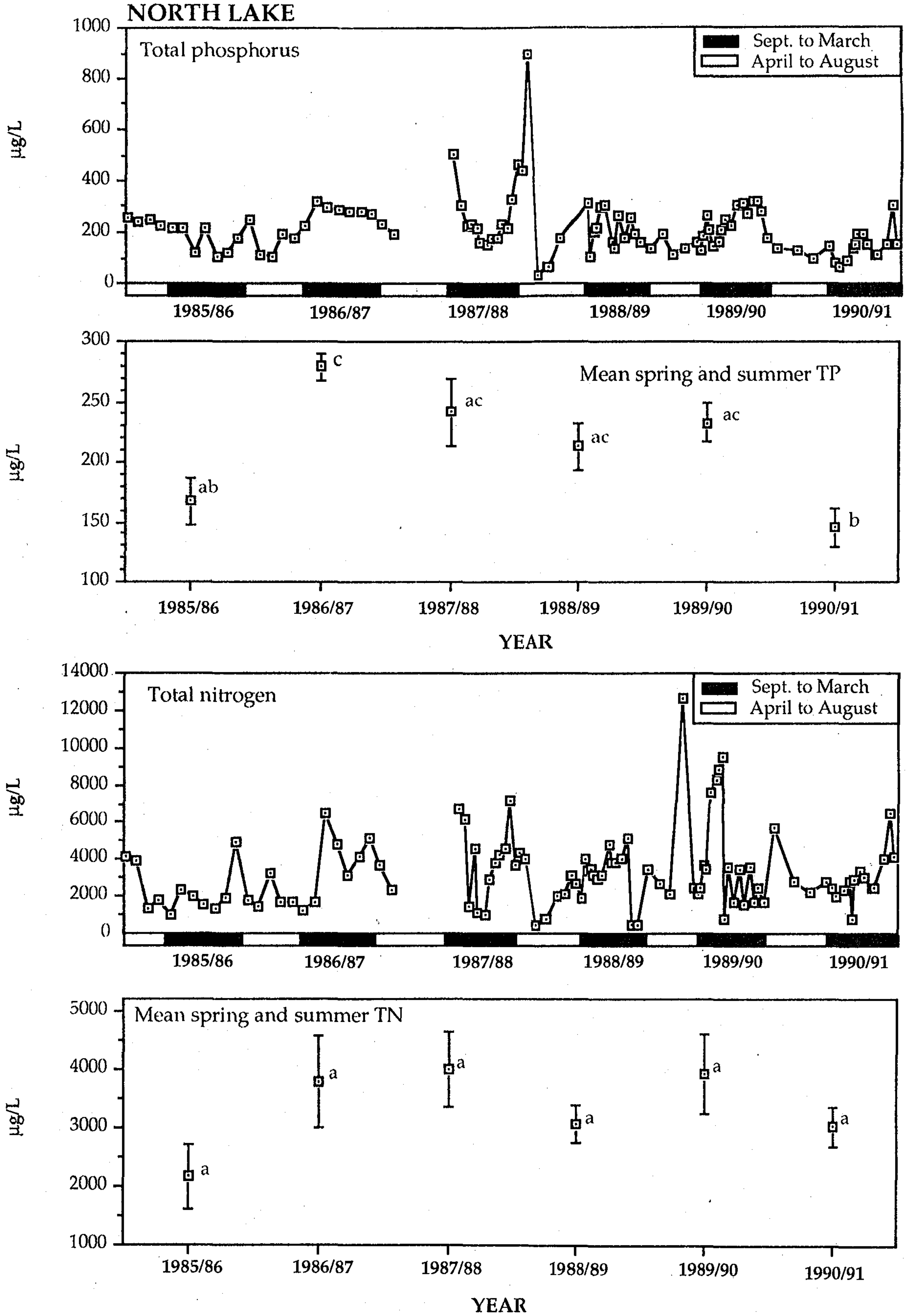

Figure 2.19. Changes in the concentrations of total phosphorus and total nitrogen, with means (± standard error) for the spring and summer periods between May 1985 and March 1991, at North Lake. Mean values followed by the same letters are not significantly different.

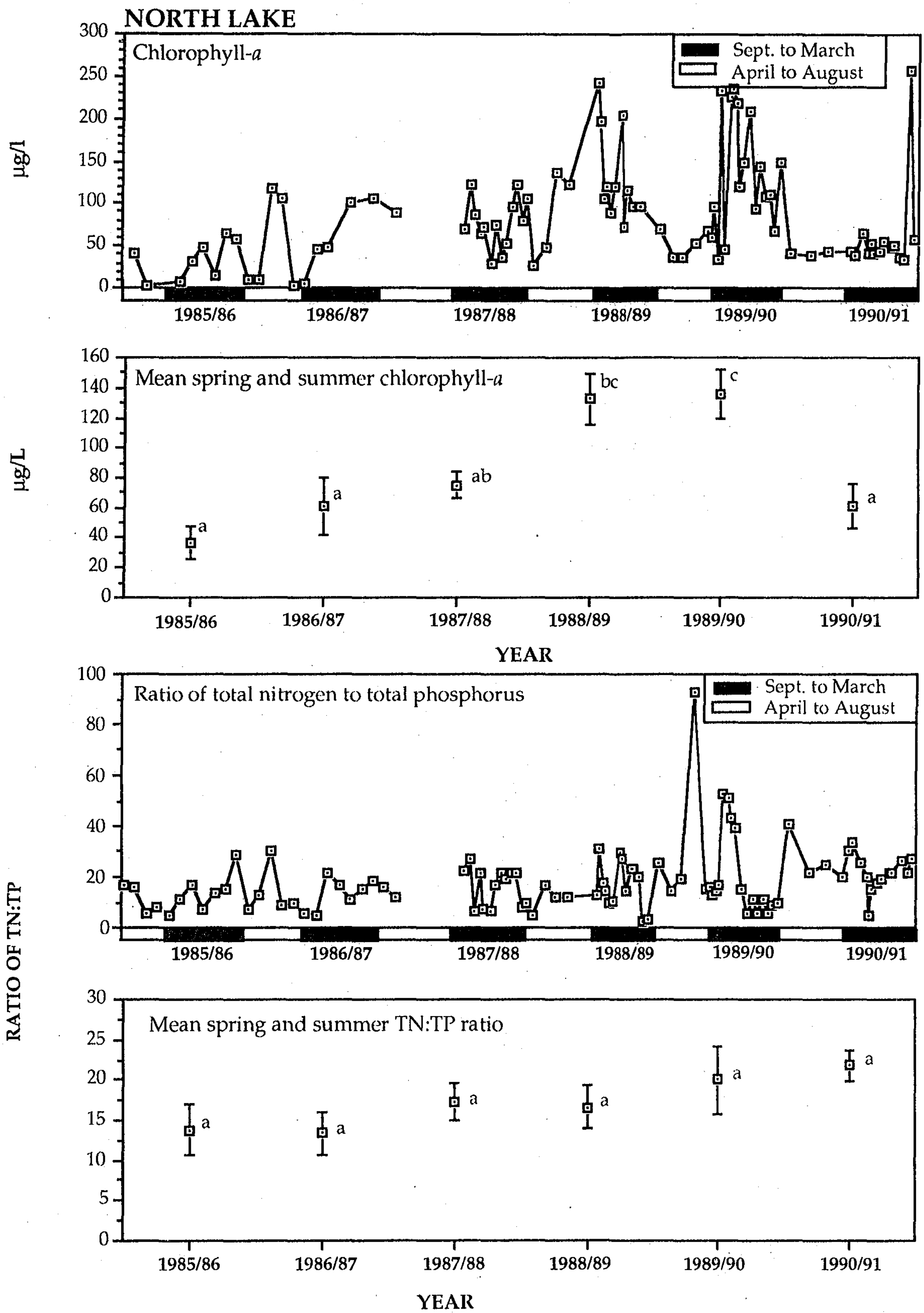

Figure 2.20. Changes in the concentration of chlorophyll-*a* and changes in the ratio of the concentration of total nitrogen to the concentration of total phosphorus, with spring and summer means (± standard error), between May 1985 and March 1991, at North Lake. Mean values followed by the same letter are not significantly different.

respectively. The spring and summer [chl-*a*] during 1988/89 was significantly higher ($P<0.05$) than during 1985/86 and the 1989/90 [chl-*a*] was significantly higher ($P<0.05$) than the previous three years (Fig. 2.20). The [chl-*a*] in 1990/91 was found to be significantly lower ($P<0.05$) than during the previous two years, but was not different to the first three years (1985/86 to 1987/88).

Despite the reduced [chl-*a*] in 1990/91, North Lake still exhibited the obvious signs of eutrophication such as odorous banks of decaying algae, as indicated by the peak in [chl-a] of 256 µg/L in February 1991 which may have been the result of a water sample being taken close to such a bank. The relationships between algal growth and changes in nutrient concentrations are discussed below.

Trophic status: The concentrations of nutrients and chlorophyll-*a* recorded at North Lake indicates that the lake lies within the eutrophic to hypereutrophic categories (even in 1990/91) of the trophic classification systems of the OECD (1982) and Wetzel (1983, after Vollenweider 1979). These classification systems are based on northern hemisphere data and have not been validated for Australian waters. However, they may still be useful indicators of the comparative health of a wetland. Nutrient and chlorophyll-*a* concentrations also exceed the ambient water quality standards for recreational use suggested by the Australian Environment Council (Australian Environment Council 1987)

Changes in nutrient concentrations at North Lake: The concentrations of nutrients in a lake are determined by the factors that influence the nutrient exchanges between the lake and the external environment and the exchanges between various sources and stores of nutrients within the lake.

Minor external sources of nutrients to a lake include rainfall and import by fauna and litter. A greater contribution of nutrients is via groundwater and the two drains (Fig. 2.1), the Kardinya storm water drain and the drain from the Murdoch University veterinary farm. In the period April 1987 to March 1988 the total input of nutrients to North Lake was 330 kg of phosphorus and 820 kg of nitrogen (Bayley *et al.* 1990). Nutrients are exported from the lake by migration of the groundwater and a small amount is lost by the movement of insects and birds. There are no surface flows of water that carry nutrients out of North Lake. Estimates from the 1988/89 and 1989/90 emergence trap data indicate that up to 2kg (dry weight) of adult chironomids/ha/night (or up to 180 kg/ha over three months) may emerge from the littoral area of North Lake. If phosphorus is only 0.5 percent of a midge body by weight, then up to 45 kg of phosphorus may be exported from North Lake by chironomid emergence during summer alone, and the emergence of other insects would add to this. However, some of this would be returned to the lake as eggs or dead midges.

Despite the already high nutrient concentrations recorded at North Lake in 1985/86, a further increase in phosphorus concentration was recorded in 1986/87 (Fig 2.19) and the concentration of this nutrient remained high in subsequent years until 1990/91. There are several factors that might account for the higher nutrient concentrations in 1986/87. Firstly, the total annual rainfall for Perth in 1985 was only 691 mm, well below the average of 869 mm, whereas the rainfall for 1986 was above average at 930 mm. Studies of the Peel Harvey estuary have suggested that nutrient loads in agricultural runoff can increase severalfold in higher rainfall years (Bayley *et al.* 1989). This probably applies to the catchment of North Lake. The amount of fertilizer applied to the veterinary farm between spring 1985 to winter 1986 was not higher than for the same period for 1984/85 (D. Brockway pers. comm.). However, the higher rainfall during winter 1986 may have leached more nutrients from the farm land into the drain in this year than during 1985. In addition, the residential suburb of North Lake (Fig. 2.1) was developed from 1983 onwards (Murdoch University 1986) and residents would have been establishing lawns and gardens over the following years. Some runoff from this suburb drains into the Kardinya drain and so increasing amounts of fertilizer and detergent-derived nutrients may have washed into the lake from 1985 onwards.

A further source of nutrients may have been the construction of a new mains sewer pipe along the eastern side of the Murdoch University campus during 1986 (Water Authority of Western Australia engineering section pers. comm.). The pipe construction involved lowering of the water table in the immediate vicinity of the pipe in order for it to be laid. Over a period of several months groundwater was pumped from the area of the sewer onto a site on the Murdoch farm within the drain catchment area. While uncontaminated groundwater is generally low in nutrients, it is possible that the groundwater from the vicinity of the farm contained above average nutrient concentrations. Thus, the pumping may have contributed to the flow and nutrient loading of the Murdoch farm drain in 1986.

As no data exist for the flows and nutrient loading of either the farm drain or the Kardinya drain prior to 1987/88, the explanations given above are mainly speculative. In addition, no reliable data are available prior to 1985/86 to support the hypothesis that nutrient concentrations in the spring and summer of 1985/86 were lower because of the lower rainfall in 1985. Other factors such as differences in internal nutrient exchange processes may also have influenced nutrient concentrations.

If the lower 1985/86 nutrient concentrations are accounted for by the above explanations rather than by interannual variations of in-lake processes, then the decrease in external nutrient loading, (as a result of the diversion of the veterinary farm drain) is a reasonable explanation of the decrease of in lake [TP] in 1990/91.

When compared to the previous four years [TP] decreased by approximately 35% in 1990/91 (Fig. 2.19). Since the drain contributed 73% of the total external nutrient load to

North Lake this decrease is less than expected (assuming that the amount of phosphorus that the drain contributed in 1987/88 was representative of the period 1987 to 1990). Marsden (1989) suggests that lake [TP] should decrease roughly in proportion to the reduction of external [TP] inputs, and that if this is not the case then internal loading (from the sediment) is implicated. Bayley *et al.* (1989) and McDougall and Ho (1991) found that internal loading at North Lake can be a major source of phosphorus in spring.

Internal transfer of nutrients from the sediments can increase in amount and importance following a major reduction in external nutrient inputs (Marsden 1989). However, the significantly lower spring and summer pH in 1990/91 (7.5 ± 0.19) compared to the previous three years (Fig. 2.17) may have inhibited sediment nutrient release. The studies of McDougall and Ho (1991) and Boers (1991) suggest that a decrease in pH of this magnitude can more than halve sediment nutrient release. The lower pH may have been due to the lower photosynthetic activity of the algae as measured by chlorophyll-*a* concentration.

Even if sediment nutrient release did increase after the drain diversion, a new equilibrium should be reached within a few years and sediment nutrient release should gradually decline (Marsden 1989). Reduction in internal loading occurs as nutrients within the sediments are converted to insoluble forms, move deeper into the sediments and thus become unavailable for release, without large external inputs to replace it. However, the sediment of some highly enriched waters can continue to release nutrients for many years and for this reason monitoring of nutrients at North Lake should be continued, to properly determine the effects of the diversion of the veterinary farm drain.

Neither total nitrogen concentration nor the ratio of TN:TP were significantly different in 1990/91 than in any of the previous five years. This may be due to the release of additional nitrogen from the sediments. However, the contribution of the sediments as a nitrogen source at North Lake has not been quantified. Alternatively, it may be due to nutrient input by the Kardinya drain, which had a higher nitrogen to phosphorus ratio than the Murdoch veterinary farm drain. Schindler (1978) suggests that changes in nitrogen input are often more rapidly corrected for by in-lake processes such as sediment release, absorption from the air and nitrogen fixation than changes in phosphorus input. In-lake nitrogen metabolism may need to be investigated further at North Lake.

Response of algae to changes in nutrient concentrations: The 1986/87 spring and summer concentrations of phosphorus at North Lake were considerably higher than those that recorded during 1985/86 (Fig. 2.19). It might have been expected that this increase in the availability of phosphorus would have resulted in an increase in the abundance of algae (as measured by chlorophyll-*a* concentration). However, an increase in [chl-*a*] was not recorded until 1988/89 and 1989/90 (Fig. 2.20). This delay in the response of algal abundance to increased [TP] must be explained before it can be assumed that the low 1990/91 [chl-*a*] is a response to the reduced [TP].

The relationships between nutrients and algal growth are complex and a change in the concentration of nutrients will not always be reflected in the abundance or species composition of algae (Marsden 1989). A common reason for this is that the nutrient that has been reduced in concentration may not have been the one that was limiting the growth of algae. Thus, the ratio of nitrogen to phosphorus concentration can be important. Many other factors such as temperature, pH, light availability, growth factors (organic chemicals in the water), micronutrients, pollutants and water turbulence influence algal growth and thereby the response of algae to changes in nutrient concentrations (Cooke *et al.* 1986).

Neither minimum nor median water temperature differed significantly ($P>0.07$) between the spring/summer periods of 1987/88 to 1990/91. Maximum spring and summer water temperature was significantly higher in 1987/88 than during the following two years which would have encouraged algal growth rather than inhibited it. It is also unlikely that nitrogen concentration or the ratio of nitrogen to phosphorus (TN:TP) contributed to the delayed increase in [chl-*a*] as these variables did not change significantly between years.

Based on the available data there appears to be no clear explanation for the delayed response of algal abundance to the increased availability in nutrients in 1986/87. It is possible that a number of factors were acting in combination to suppress algal growth for the spring/summer periods of 1986/87 and 1987/88.

The decreased concentration of phosphorus in the spring and summer following the drain diversion was accompanied by a lower [chl-*a*] than had been recorded in the two years prior to the diversion (Fig. 2.20). Whether the decrease in [chl-*a*] was a response to the reduced [TP] or to the same factors that caused [chl-*a*] to be low in 1986/87 and 1987/88 cannot be determined for certain. However, a review of the literature and the available data may help to determine the probability of such a response.

Many studies have demonstrated that reducing external phosphorus load can lead to significant increases in water quality (e.g. Edmondson 1970, Smith and Shapiro 1981 and Bailey-Watts 1982). However, several studies of northern hemisphere lakes suggest that the reduction in external phosphorus loading must be as high as 60 to 70% to significantly reduce the production of algae (OECD 1982; Maki *et al.* 1984; Forsberg 1985a and Seip *et al.* 1990). Such large reductions in phosphorus input were needed because the lakes were often already very eutrophic. Other studies have found no reductions in [chl-*a*] even after reductions in phosphorus inputs of similar magnitude (Ryding 1981).

Whether or not a change in algal growth occurs after a reduction in external phosphorus inputs can depend upon whether nitrogen or phosphorus is the main nutrient limiting algal growth (Marsden 1989). Nitrogen is generally considered to be limiting if the ratio of total nitrogen to total phosphorus (TN:TP) is less than a certain value. This value is not fixed, and can vary between lakes and for different species of dominant algae. Cooke *et al.* (1986) suggested that TN:TP ratios <13 generally indicate that nitrogen is potentially limiting, whereas ratios above 20 indicate phosphorus limitation. However, they qualify this by explaining that TN:TP ratios only indicate which nutrient will be used up first when growth is not limited by another factor such as light or temperature.

The ratio of total nitrogen to total phosphorus at North Lake varied between 10 and 30 for most of the six years of monitoring (Fig. 2.20) suggesting that either nitrogen or phosphorus may be the potentially limiting nutrient. Bayley *et al.* (1989) found that while nitrogen was the limiting factor in North Lake for much of 1987/88, phosphorus may have become limiting during late spring and summer.

If nitrogen is limiting then phosphorus concentrations may have to be reduced to levels that make phosphorus the limiting factor before water quality improves. Smith and Shapiro (1981) however, criticise the view that nitrogen limited lakes will not respond to reductions in external phosphorus loading. They provide evidence that even in lakes that are assumed to be nitrogen limited (TN:TP of 4-9) a significant decline in chlorophyll-*a* concentration can be found after reduction of external phosphorus inputs.

At North Lake it is possible that temperature may also have been limiting the growth of *Microcystis* (the major bloom forming species of algae). The average median water temperatures at North Lake are generally in the range of 20 to 25 °C during spring and 24 to 26°C during summer (Fig. 2.18), while average maximum temperatures are in the ranges 22 to 27°C for spring and 27 to 30 °C during summer. *Microcystis* species have an optimum temperature range for growth of 27 to 35 °C (Robarts and Zohary 1987), thus spring and summer temperatures at North Lake may be in the lower half of the optimum range.

In summary, the diversion of the Murdoch veterinary farm drain is a likely explanation of the decrease in [TP] and [chl-*a*] in 1990/91. However, this is only the first year of post drain diversion data and considering the variable results of overseas studies conclusions regarding the success of this source reduction cannot be made without further monitoring. In particular, the nitrogen and phosphorus concentration in the lake, the nutrient inputs to the lake, and sediment nutrient release should be monitored for at least a further five years. Despite the lack of firm conclusions regarding the effects of the drain diversion it is clear that a link exists between chironomid abundance and eutrophication. Therefore lake restoration is seen as the primary long term control strategy and any attempts to reduce nutrient loading of lakes should be commended.

Forrestdale Lake

Depth

Forrestdale Lake dried out completely on two occasions (February 1988 and December 1990) and on a third occasion (autumn 1990) lake depth fell to less than 14cm (Fig. 2.21). In the remaining year (1988/89) the minimum recorded depth was only 43cm. Maximum lake depth at Forrestdale Lake appears to be very dependent upon winter rainfall and the minimum depth of the previous summer. Minimum depth is dependent upon winter maximum depth and the rate of evapo-transpiration during spring and summer. The general rate of decrease in depth during spring and summer is approximately the same each year (four centimetres per week).

Conductivity

As the lake became very shallow during summer (<30 - 40cm), conductivity rose to over 10 000 µS/cm (5 ppt) and exceeded 20 000 µS/cm (10 ppt) (seawater is 75 000 µS/cm or 35 ppt) (Fig. 2.21). Larval monitoring suggests that populations of some species (e.g. *P. nubifer*) are limited by high conductivity (> 15 000 µS/cm) while one species (*T. fuscithorax*) is generally the most abundant species when conductivity is above 10 000 µS/cm .

pH

The pH of Forrestdale Lake varied between 7.1 and 9.9 during the study and was generally higher during the warmer months due to higher levels of photosynthetic activity (Fig. 2.21). In 1990/91 pH was above 8.5 during spring and summer despite the fairly low concentration of chlorophyll-*a* recorded. However, the submerged macrophyte growth was observed to be more abundant in 1990/91 than during previous years (this is a qualitative observation as no study of plant biomass was made) The photosynthetic activity of these plants is not accounted for by measurements of chlorophyll-*a* in the water column of the lake.

Water Temperature

Water temperature increased during each spring and summer as lake depth fell and ambient air temperatures increased (Fig. 2.22). The mean spring and summer minimum temperature was significantly lower ($P<0.05$) in 1990 than in 1988/89, while the mean spring and summer maximum temperature in 1990 was not significantly different to any other year. In 1988/89 maximum temperature was significantly lower ($P<0.05$) than during the same period of 1987/88 and 1989/90. No other differences in maximum and minimum water temperatures were apparent. Maximum and minimum temperatures represent extremes for each sampling interval and so median temperature may be a more representative measure of water temperatures. The median temperatures were usually 13 - 16 °C during the cooler months and 15 - 30 °C during spring and summer. Median temperature was significantly lower ($P<0.05$) in 1990 than during both 1987/88 and

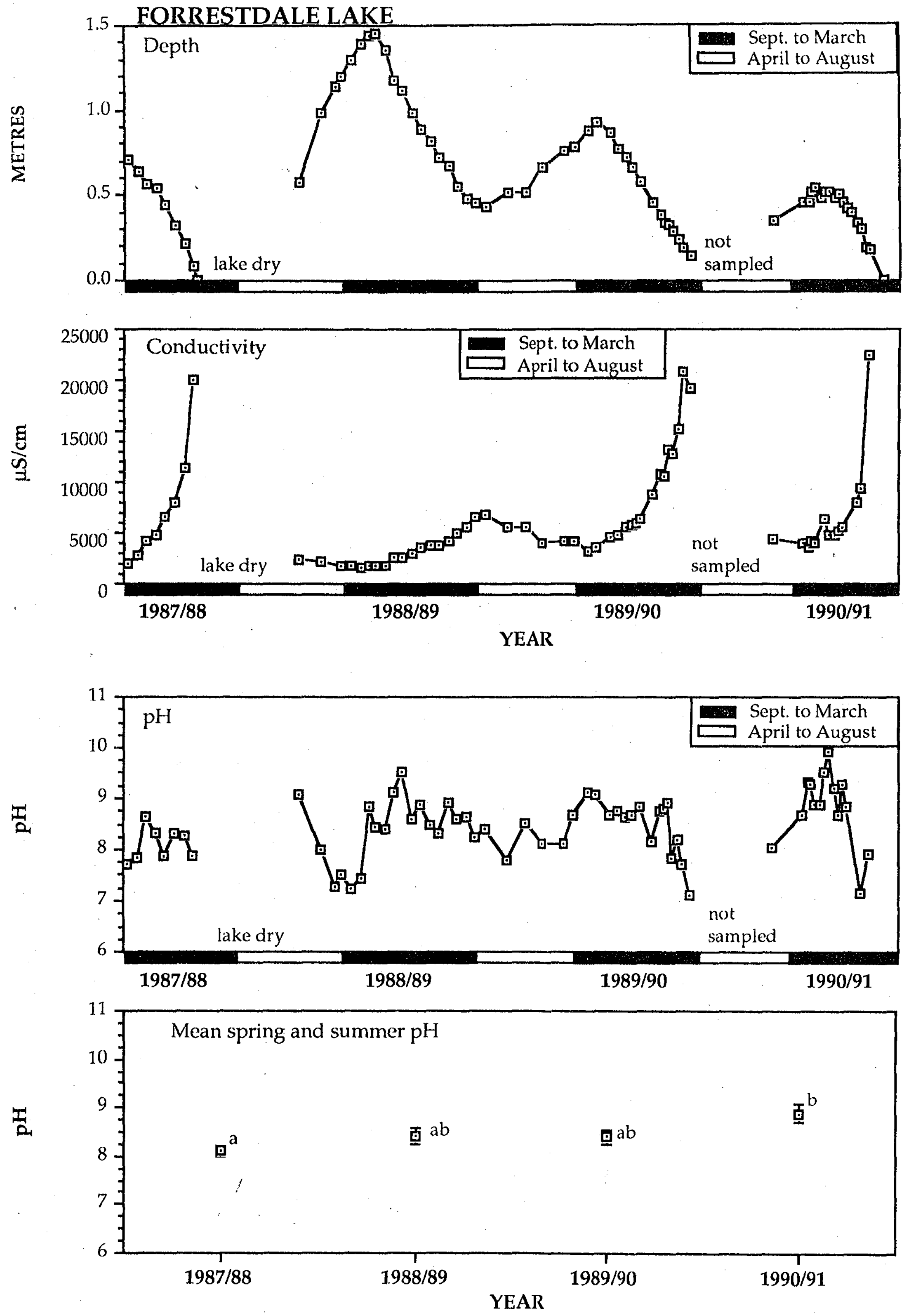

Figure 2.21. Changes in depth and conductivity and changes in pH, with means (± standard errors) for the spring and summer periods, between October 1987 and December 1990, at Forrestdale Lake. Mean values for pH followed by the same letter are not significantly different.

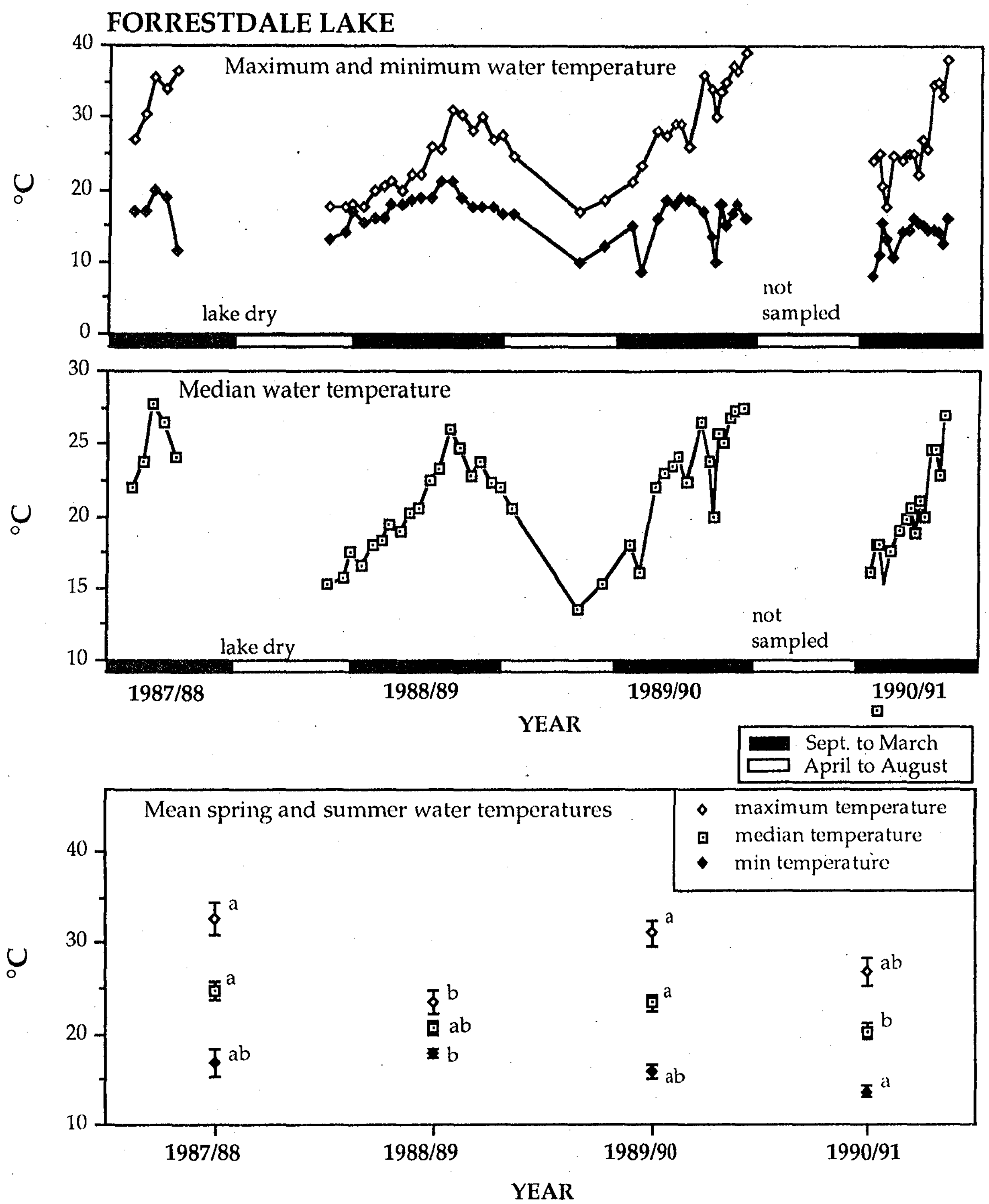

Figure 2.22. Changes in minimum, maximum and median water temperature, with spring and summer means (± standard error), between October 1987 and December 1990, at Forrestdale Lake. Mean values of the same variable followed by the same letter are not significantly different.

1989/90 (Fig. 2.22). The most likely reason for this lower water temperature during 1990 is that the lake dried in early summer (before the hottest part of the year) and so water temperature was only measured on three occasions during the summer of 1990. If the means of only the spring temperatures are compared, then neither the minimum nor median temperatures were significantly lower in 1990 than in previous years, and maximum temperature is significantly higher in 1990 than in 1988/89.

Nutrients and chlorophyll-*a*

Phosphorus: Mean spring and summer total phosphorus concentrations at Forrestdale Lake (Fig. 2.23) ranged from 329 ± 43 μg/L (1987/88) to 96 ± 8 μg/L (1990) and the overall range was 57 to 523 μg/L. Total phosphorus concentrations were found to be significantly lower ($P<0.05$) in spring and summer of 1990 than during the previous three years. There are very few summer data for 1990 because the lake dried. Therefore it may be more appropriate to compare mean phosphorus calculated from only the spring dates. The 1990 spring phosphorus concentrations were also significantly lower than for the previous three years.

Whether this lower [TP] during 1990 was due to reduced external loading or to in-lake processes is unclear since a nutrient budget has not been prepared for Forrestdale Lake and hence the contribution of external and internal loading of nutrients to the lake have not been quantified.

An alternative explanation for the lower [TP] is greater submerged macrophyte growth (mainly *Chara* and *Ruppia*) in 1990 than in previous years. These aquatic plants can absorb nutrients directly from the water column and so compete with phytoplankton (Wetzel 1983). The turbidity also appeared to be much lower in 1990 perhaps as a result of increased binding of the sediment by macrophyte growth. The reasons for the abundant macrophyte growth are not clear, although increased light penetration due to the lower water depth of the lake is a likely cause. While *Typha* can also absorb large amounts of phosphorus, the stands of *Typha* were not substantially larger in 1990 than in previous years, and these plants mainly absorb nutrients from the sediment (Wetzel 1983).

Nitrogen: Mean spring and summer total nitrogen concentrations (Fig. 2.23) were significantly lower in 1988/89 (2876 ± 421 μg/L) and 1989/90 (1790 ± 322 μg/L) than during 1987/88 (6876 ± 2067 μg/L) and 1990/91 (5759 ± 680 μg/L). A regular seasonal trend was not apparent, although [TN] increased immediately before the lake dried out in both 1987 and 1990.

Chlorophyll-a: The mean spring and summer chlorophyll-*a* concentrations for the three years 1987/88 to 1989/90 were 45 ± 17, 64.6 ± 13 and 105 ± 9.1 μg/L (Fig. 2.24). These three years were characterized by increasing [chl-*a*] during spring and summer and one or more peaks. Winter chlorophyll-*a* data are sparse and varied substantially.

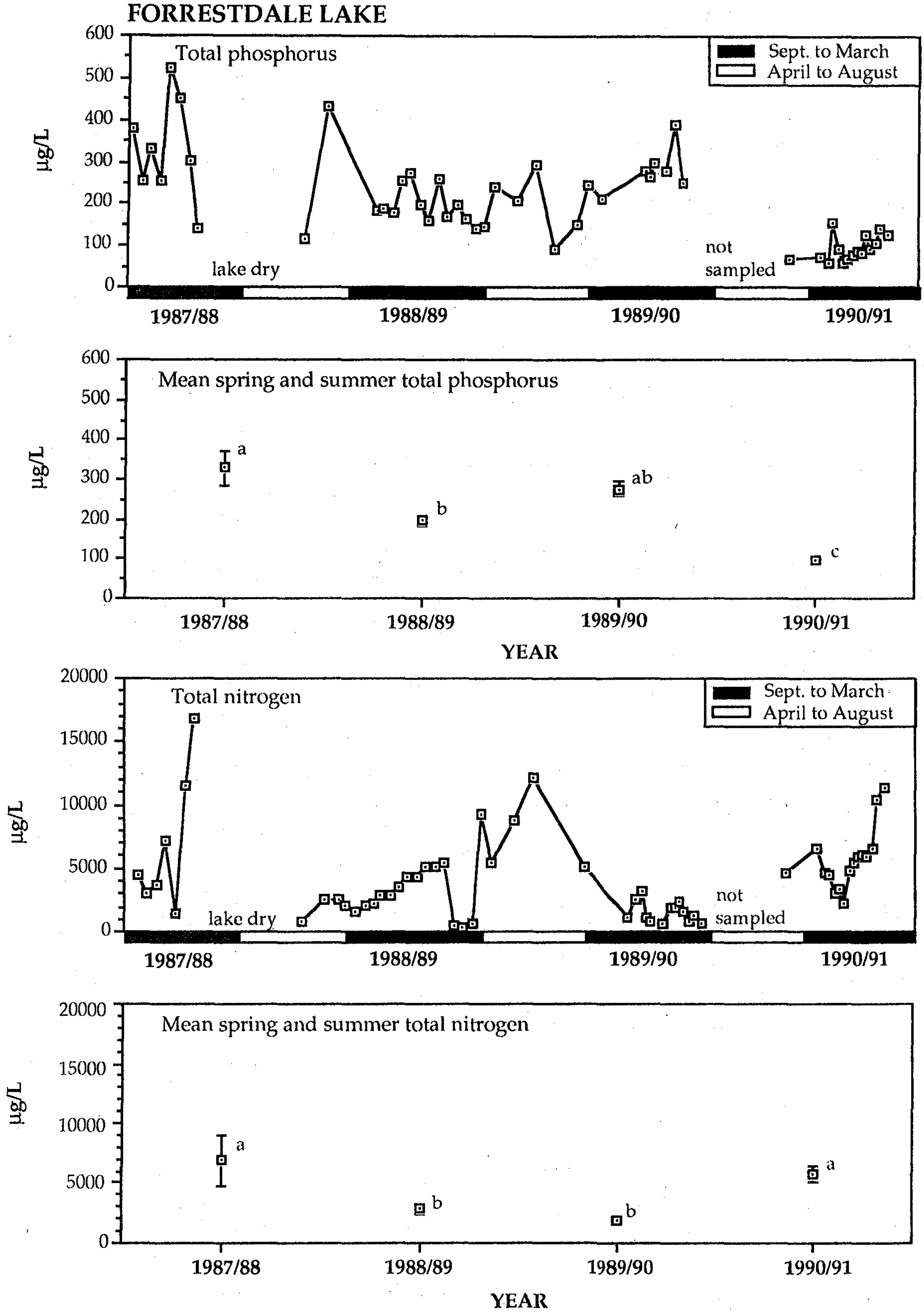

Figure 2.23. Changes in the concentrations of total phosphorus and total nitrogen, October 1987 to December 1990 at Forrestdale Lake, with means (± standard errors) for the spring and summer periods. Mean values followed by the same letter are not significantly different.

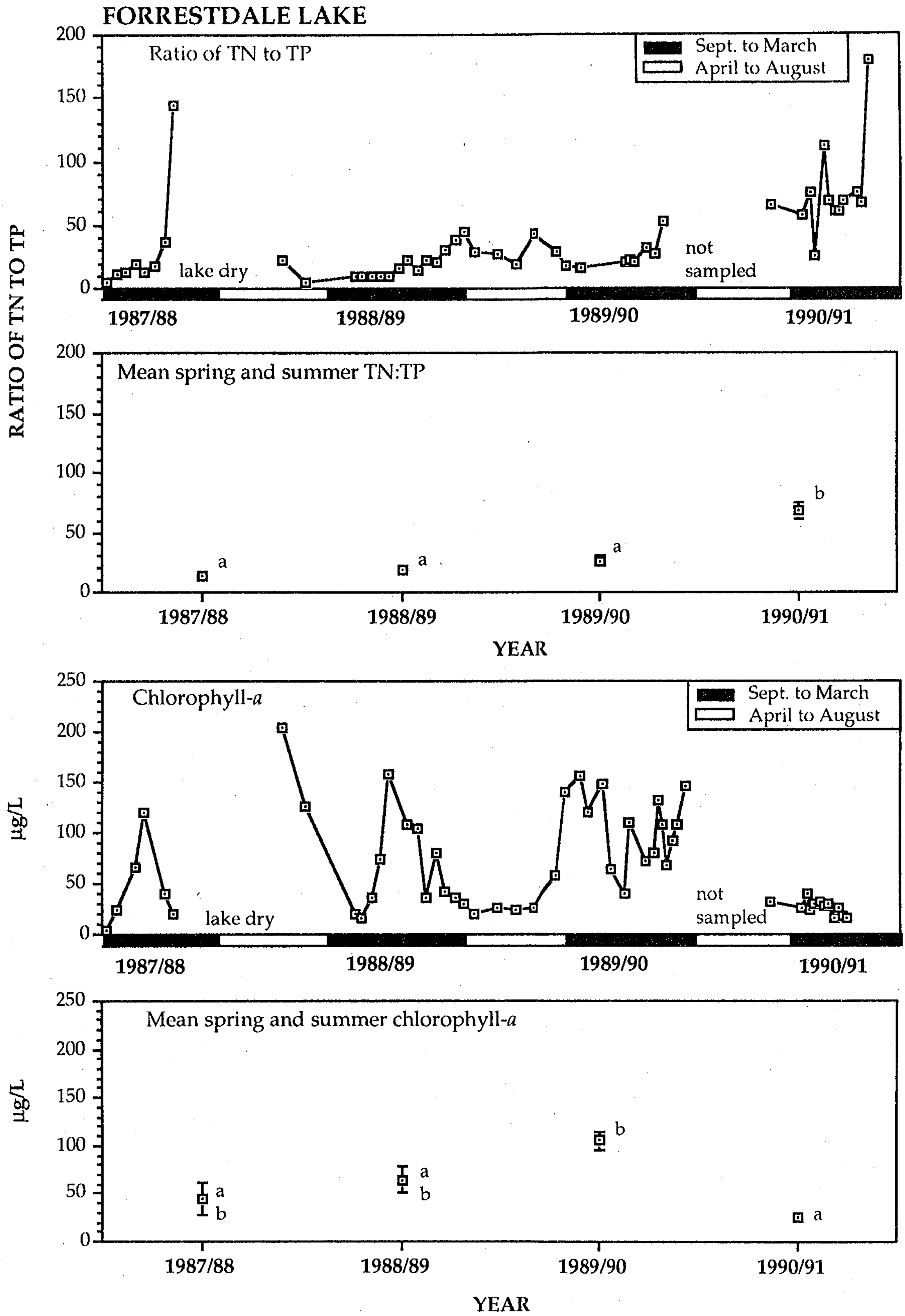

Figure 2.24. Changes in the ratio of the concentration of total nitrogen to total phosphorus and changes in the concentration of chlorophyll-a, between October 1987 and December 1990, at Forrestdale Lake, with means (± standard errors) for the spring and summer periods. Mean values followed by the same letter are not significantly different.

In the following year (1990/91) chorophyll-*a* remained below 50 µg/l (with a mean of 26 ± 2 µg/L). This is significantly lower ($P<0.01$) than during only one (1989/90) of the three previous spring and summer periods. However, the 1990/91 value includes spring data only and so should be compared only with the spring means of previous years, which are 14 ± 10 (1987), 24 ± 6 (1988) and 126 ± 17 (1989). Thus the spring [chl-*a*] of 1990 is only lower than the equivalent period of 1989.

Despite the often high concentrations of chlorophyll-*a*, dense blooms of blue-green algae (such as those that occur at Lake Monger and North Lake) were rarely observed, suggesting that other types of algae were also abundant.

Trophic status: Forrestdale Lake lies within the eutrophic to hypereutrophic categories of the classification systems of the OECD (1982) and Wetzel (1983, after Vollenweider 1979) based on nutrient and chlorophyll-*a* concentrations.

Lake Monger

Nutrient and chlorophyll-*a* concentrations at Lake Monger have been measured for five of the past six years by Davis and Rolls (1987), Rolls (1989), Lund (unpublished data) and some undertaken as part of the midge project. The spring and summer concentration of total phosphorus in 1990/91 (161 ± 38 µg/l) appeared to be lower than during previous years (averages of 333 ± 151 to 511 ± 30 µg/l). The concentration of phosphorus in the water in 1990/91 should still have been sufficient to result in major algal blooms. However, no blooms were observed in 1990 and this was reflected in the lower [chl-*a*] (41 ± 8.2 µg/l) in 1990/91 than in previous years (for which averages of 78 ± 19 to 550 ± 134 µg/l were calculated). Peak [chl-*a*], which can also used as a trophic indicator (Wetzel 1983), was also much lower in 1990/91 (57 µg/l compared to 146 to 1044 µg/l recorded in previous years). The [TP] was not sufficiently lower in 1990/91 to solely account for the lower algal abundance. Therefore, other factors such as temperature or grazing by zooplankton must have contributed to the lower [chl-*a*] at Lake Monger in this year. Temperature data are not available but grazing zooplankton were abundant at Lake Monger throughout the summer of 1990/91, whereas in previous years they have almost disappeared from the lake during summer. Total nitrogen concentration was not lower in 1990/91 than in previous years.

Relationships between environmental variables and chironomid populations

Two approaches to relating environmental factors to changes in insect populations can be taken. Firstly, environmental variables may be used to predict the timing of population changes within a year. With chironomids this would involve predicting when a major increase in larval density and thus nuisance problems will occur. Secondly, the factors which govern the abundance of insects between years can be identified. Both of these approaches will be discussed in this section.

Forrestdale Lake

At Forrestdale Lake water temperature has been found to be the most consistent predictor of when a major increase in the density of *P. nubifer* is likely to occur. In the 1990 midge research report (Davis *et al.* 1990) it was suggested that maximum temperature was the measure of temperature that provided the most consistent prediction of when a rise in *P. nubifer* density would occur. However, median temperature has since been found to be preferable as it allows the first year of chironomid and temperature data to be considered and provides a narrower prediction time.

In 1987 to 1989 a major increase in the density of *P. nubifer* (several hundred percent over 2 - 3 weeks) was recorded four to six weeks after median temperature had exceeded 22 °C (Fig. 2.25). This major increase in *P. nubifer* larval density caused the most severe nuisance problems in each year. The month during which median temperature crosses this critical level varied according to the depth of the lake. The deeper the water column, the less influence rising summer air temperatures have on water temperature at the sediment/water interface. Thus, the timing of the problem also varies with lake depth. In addition, the earlier that water temperature crosses the critical level, the longer will be the duration of nuisance problems. Water temperature also partly depends upon air temperature but this is very consistent between years (Fig. 2.15).

In 1987 median temperature exceeded 22°C in late November, resulting in a rapid rise in the density of *P. nubifer* over a six week period and high levels of nuisance by early January (Fig. 2.25). However, in February rising conductivity associated with the shallow lake depth appeared to supress chironomid populations and the lake subsequently dried out.

By contrast, lake depth was considerably higher (>1.4 metres) in 1988 (Fig. 2.25) and so median temperature did not exceed 22°C until January (Fig. 2.25). The resulting high *P. nubifer* density and severe nuisance problems in February were reduced by an application of Abate to the lake. Following this, water temperature declined (Fig. 2.25) and *P. nubifer* remained at low densities. In 1989, lake depth was again shallower (0.9 metres maximum) and so water temperature rose above the critical level by November (Fig. 2.25). The resulting high *P. nubifer* densities in December was reduced by a treatment with Abate. However, water temperatures were still high enough to allow a second major increase in the abundance of *P. nubifer* in January and a second treatment was required. A third increase in *P. nubifer* density was recorded in February, however this was foreshortened by rising conductivity.

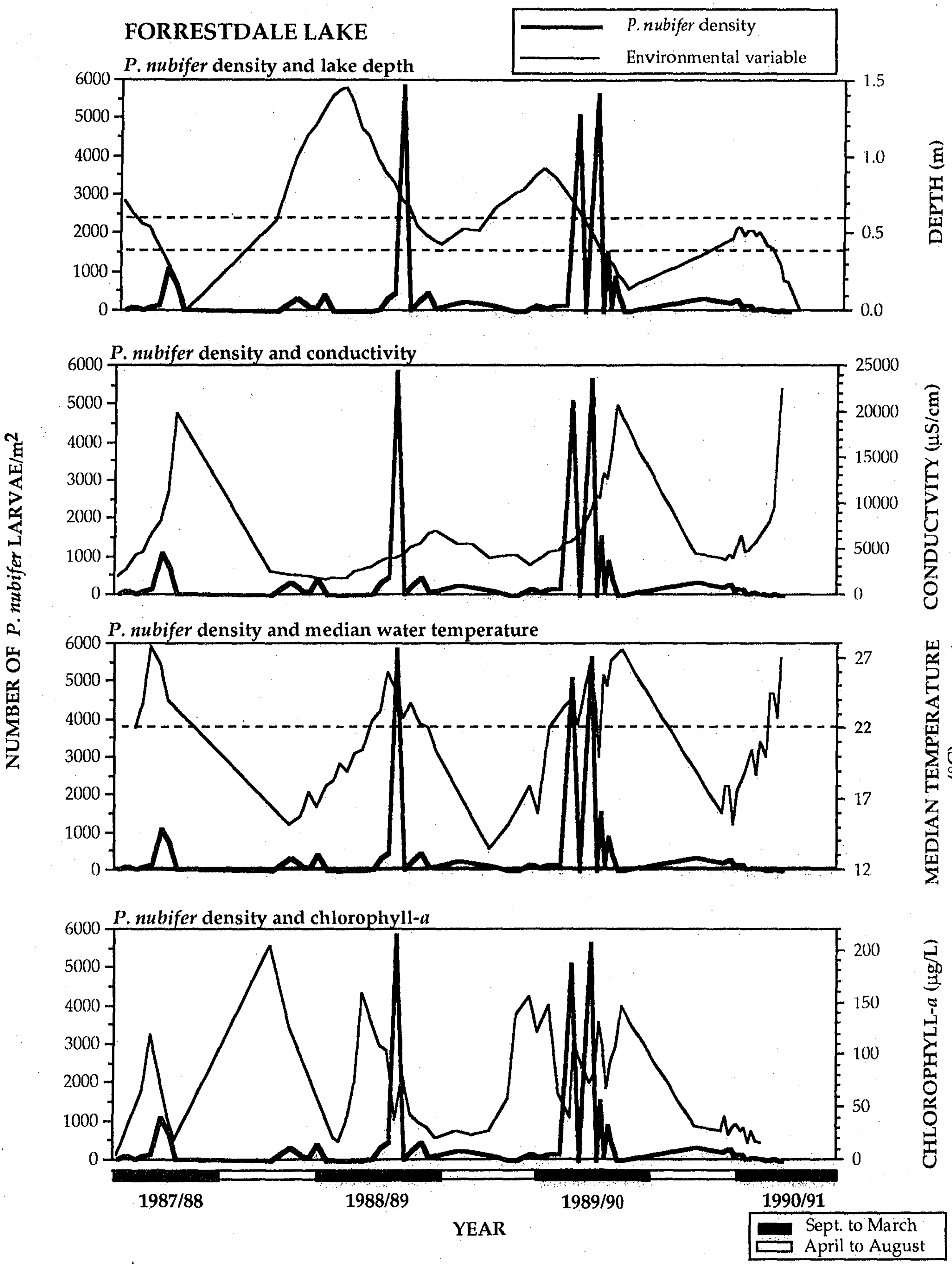

Figure 2.25. Changes in the density of *Polypedilum nubifer* with changes in lake depth, conductivity, median water temperature and concentration of chlorophyll-*a* at Forrestdale Lake, October 1987 to December 1990.

In the final year of monitoring (1990) median temperature exceeded 22°C in mid-November (Fig. 2.25). However, within six weeks of this, conductivity was sufficiently high (>15 000 µS/cm) to prevent a rise in *P. nubifer* altogether and the lake subsequently dried.

In addition to the rise in conductivity when the lake becomes shallow, it also possible that oxygen concentrations decrease due to the high temperatures and large amounts of decomposing animal and plant biomass. This lower oxygen concentration may also limit growth of chironomids when the lake is shallow.

Thus, the timing and duration of nuisance problems is dependant upon water temperature and conductivity and these are partially dependant upon lake depth. While this suggests that lake depth may be manipulated to control chironomid populations, this approach may not be compatible with the conservation aims of the Forrestdale Lake nature reserve or with Typha control strategies.

The effect of algal abundance upon interannual variation in larval abundance is difficult to establish. Lake depth, influencing conductivity, water temperature and possibly oxygen concentration, appears to be the dominant factor influencing the abundance of chironomids at Forrestdale Lake. These factors probably overide any more subtle effects of algal abundance. Thus, although the lower chlorophyll-*a* in 1987 (than in 1988 and 1989) was associated with lower abundance of *P. nubifer* (Fig. 2.25), the latter was most likely due to the suppression of larval abundance by deteriorating water quality as the lake dried. In 1990 only spring data are available and chlorophyll-*a* was not lower than in two of the three previous years (Fig. 2.25). Both chlorophyll-*a* and chironomid abundance may have increased during summer if the lake had been deeper.

In summary, at Forrestdale Lake the timing and duration of nuisance problems appears to be determined primarily by water temperature and lake depth/conductivity. If median water temperature has risen above 22°C then a rapid rise in *P. nubifer* density can be expected four to six weeks later, unless lake depth is already below 40cm, in which case the lake may dry before nuisance problems arise. If lake depth is already below 60cm, then a problem may still occur but should be short-lived.

North Lake

The prediction of nuisance problems at North Lake is not as straight forward as at Forrestdale Lake. The lack of well defined increases in larval density and the loose relationship between the abundance of larvae and the perceived level of nuisance means that there is no obvious event to predict.

The most consistent predictor of larval density at North Lake is maximum temperature, which generally increases above 20°C three to seven weeks prior to the density of larvae exceeding 8000 larvae/m^2 (Fig. 2.26). This is a very wide time period and so is of little practical use for predicting nuisance problems and making decisions about the timing of pesticide applications. For the present, monitoring of larval populations directly is probably still the best method for judging when nuisance problems will occur at this lake. Even this approach has problems because of the loose relationship between larval density and nuisance problems at North Lake, which is complicated by the potential contribution of Bibra Lake chironomids to nuisance problems at North Lake.

The abundance of *P. nubifer* varies considerably between years. In 1988/89 and 1989/90 the density of *P. nubifer* larvae increased to over 15 000 larvae/m^2 on more than one occasion, while in 1987/88 and 1990/91 the density of this species remained below 4000 larvae/m^2 for most of spring and summer. The three environmental parameters most likely to account for such interannual variation are water temperature, abundance of algae and lake depth.

The only significant differences in water temperature (during the spring and summer) between any of the four years was the higher maximum temperature in 1987/88 (Fig. 2.18). Thus, water temperature is unlikely to have contributed significantly to interannual variation in chironomid densities.

The interannual variation in larval *P. nubifer* abundance is more readily related to chlorophyll-*a* concentration. In the two years in which high densities of *P. nubifer* were recorded the mean spring and summer [chl-*a*] was over 130 µg/l, whereas in the years with lower densities of *P. nubifer* larvae mean spring and summer [chl-*a*] was less then 80 µg/l (Fig. 2.27). A significant correlation (r^2 = 0.97) was calculated between mean summer *P. nubifer* density and mean spring and summer chlorophyll-*a* concentration (Fig. 2.27). However, too much emphasis should not be placed on a correlation based on only four years data, especially since pesticides may have altered the natural response of chironomid populations to algal abundance. Correlations on a finer scale than spring/summer means are not possible due to the presence of winter algal blooms and the effects of pesticides.

The abundance of *P. nubifer* larvae may also have been effected by the lower depth of North Lake in 1990/91. The shallower depth caused the shoreline to recede exposing the sandy littoral sediment which has previously supported high densities of *P. nubifer*. While the sediment in the littoral region of North Lake in 1990/91 was still fairly sandy, much of it was covered with the fine mud which was restricted to the deeper region in previous years. This may have had a negative effect on the abundance of *P. nubifer*, which tends to prefer cleaner sandy regions.

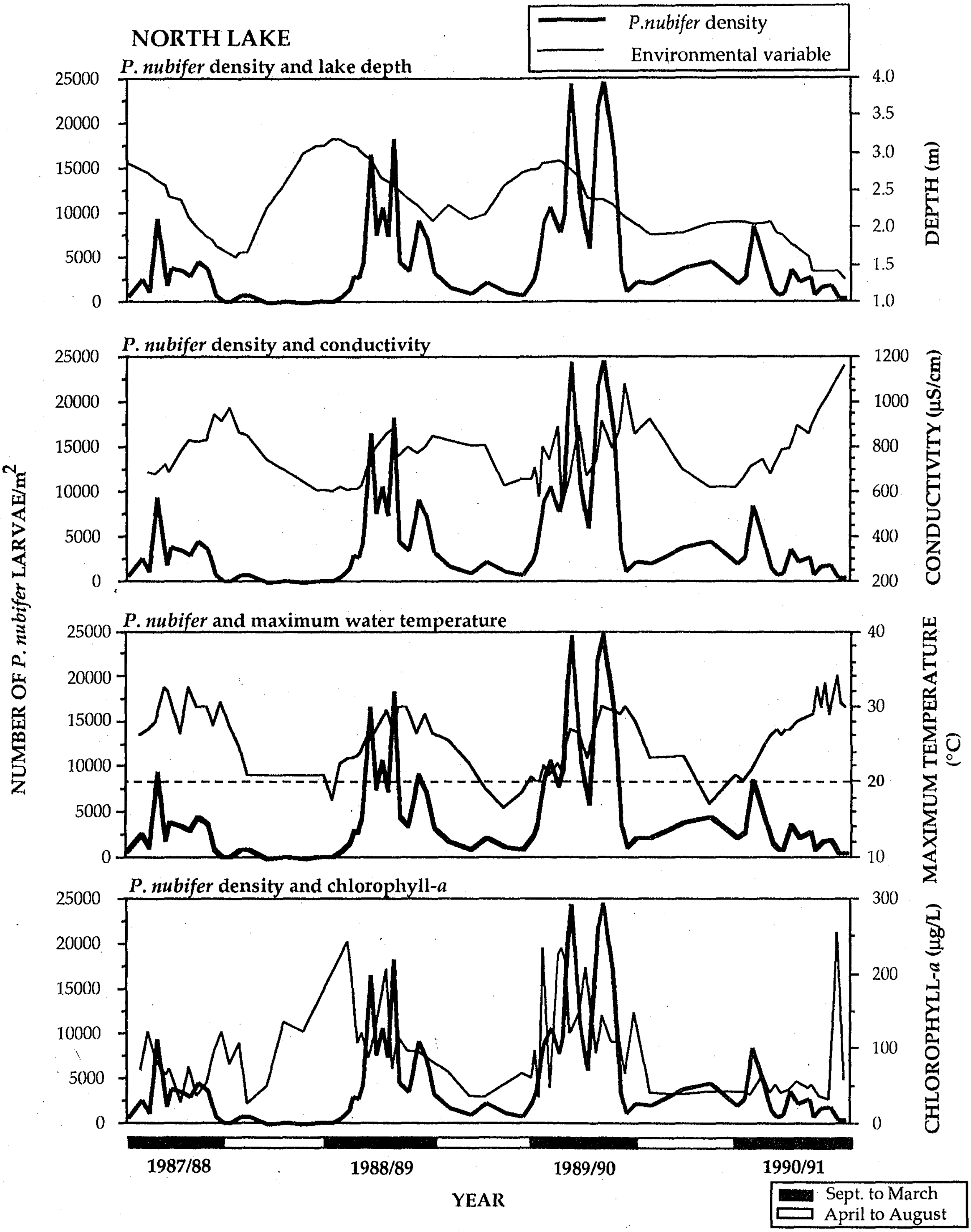

Figure 2.26. Changes in the density of *Polypedilum nubifer* with changes in the lake depth, conductivity, maximum water temperature and concentration of chlorophyll-*a* at North Lake, October 1987 to March 1991.

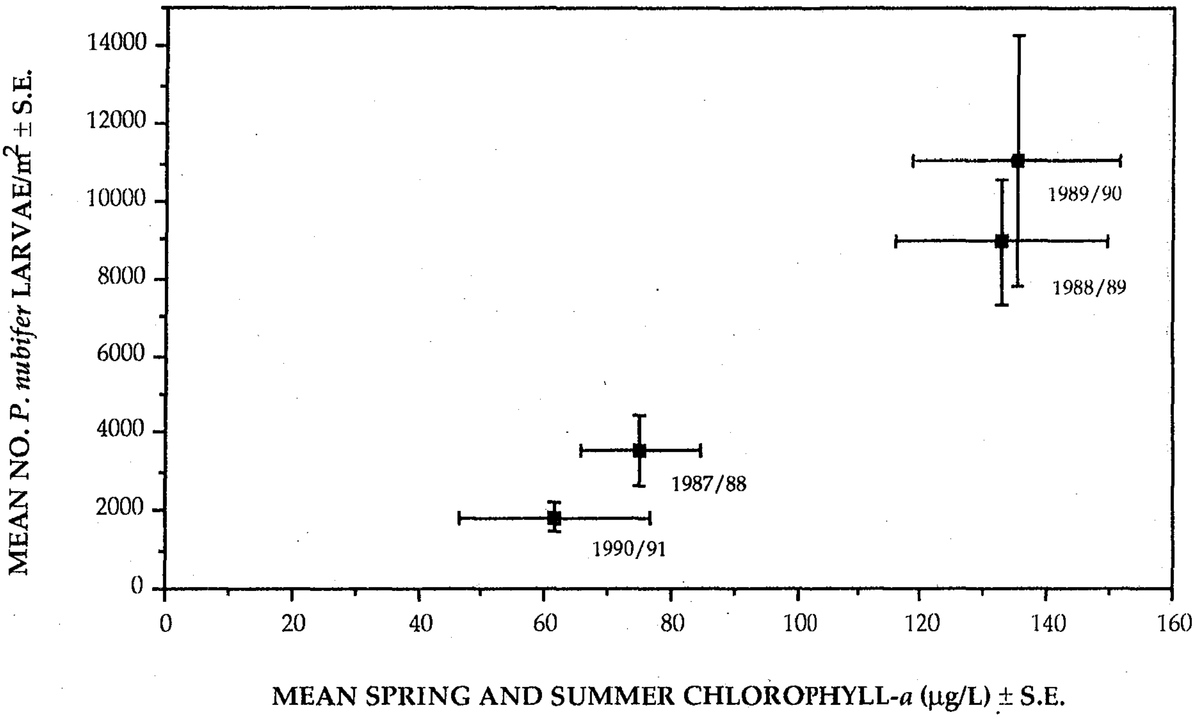

Figure 2.27. Mean spring and summer concentration of chlorophyll-*a* vs. mean summer density of *Polypedilum nubifer* larvae in the littoral region of North Lake, for the years 1987/88 to 1990/91.

Other Lakes monitored during the midge project

To what extent the relationships between chironomids and environmental variables observed at North Lake and Forrestdale Lake are representative of other Perth wetlands is not clear. The general effects of increased temperature and abundance of food on chironomids will be universal, but may be modified by or overidden by other factors peculiar to each lake. Critical temperatures and nuisance threshold densities of larvae (or emergence of adults) need to be determined individually for each lake.

Monitoring at Lake Monger also provided some evidence for the importance of eutrophication to the abundance of chironomids. At Lake Monger the density of *P. nubifer* larvae was substantially lower in 1990/91 than in the previous year, and nuisance problems were lower than during the previous four years. The most likely explanation for this is the lower algal abundance in 1990/91 compared to previous years.

The first year of the midge research involved the study of chironomid populations and environmental parameters at a variety of lakes in Perth (Davis *et al.* 1988). The results of this monitoring showed that of the six wetlands studied, those that were most nutrient enriched generally had the highest abundance of chironomids and the highest incidence of nuisance problems. Booragoon Lake, which had very high nutrient concentrations but very low chlorophyll-*a* concentration, was an exception to this. The reason postulated for the low chlorophyll-*a* at this lake was the highly tannin stained water which may have suppressed algal growth by light limitation.

Additional evidence for a positive relationship between chironomid abundance and degree of lake eutrophication comes from data collected on 40 Perth wetlands as part of the 'wetlands classification project'. These data suggest that lakes with the highest total phosphorus concentration are likely to have higher abundance of chironomid larvae than lakes with less phosphorus (Fig. 2.28). The exceptions to this were lakes that were either highly tannin stained (e.g. Carabooda, Spectacles, Kogolup, Gin Gin and Bartram), highly polluted (Neerabup), or were seasonal lakes which were almost dry when sampled (Forrestdale Lake) (unpublished data).

There are numerous other published studies of relationships between algal abundance and the abundance of chironomids. In general, these studies substantiate the hypothesis that eutrophication enhances the production of chironomids. These studies are of two main types; those that monitor changes in a lake within a year or over several years (temporal studies), and those that involve the comparison of several lakes or several areas within a large lake (spatial studies).

In the former category the majority of studies show that chironomids are most abundant (or have a maximum period of growth and emergence) when food is most available

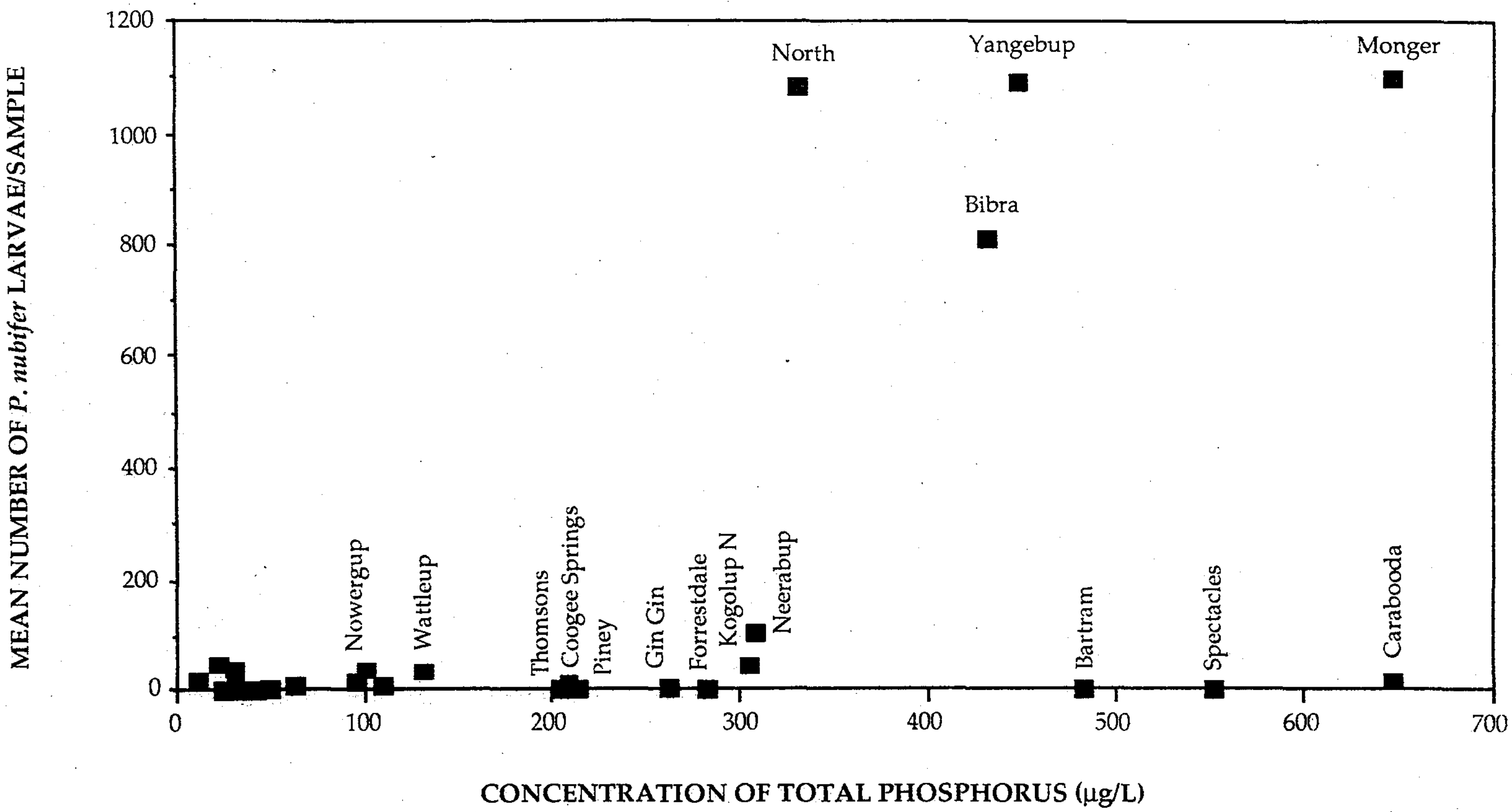

Figure 2.28. Concentration of total phosphorus vs. mean abundance of *Polypedilum nubifer* larvae in Perth wetland in summer 1989. Data collected as part of the AWRAC/EPA/Water Authority wetlands classificaton project.

(Rasmussen 1984 and Jonasson 1972). However, such periods did not always occur when temperature was highest, suggesting that food availability may have been more important than temperature in some lakes. These studies were carried out in cold temperate lakes which usually have well defined periods of food availability and low temperatures year round. In the most eutrophic Perth wetlands algal abundance is less seasonal and chironomids tend to be most abundant in spring and summer when temperature is highest. Few studies have examined interannual variation of chironomid populations within a lake.

Of the studies that involve inter-lake comparisons, the work of Davies (1980) and Welch *et al.* (1988) show the clearest relationships between chironomid and algal abundances. These studies involved the analysis of data from the Canadian experimental lakes area and demonstrated that phytoplankton abundance explained 74% of variation in the numbers of chironomids emerging and 96% of variation in emergent biomass. In another large Canadian lake, chironomid biomass was found to be 2.8 times higher at one end of the lake than at the other end. Depth, oxygen and temperature were similar at both sites but the site with higher chironomid production had significantly higher algal abundance (Dermott *et al.* 1977). In contrast to these studies, Elmore *et al.* (1984) found that oxygen concentration limited chironomid production in a shallow, highly eutrophic Californian lake while a less eutrophic lake had higher chironomid densities, thus demonstrating that factors other than food availability can limit chironomid populations.

Proving and quantifying relationships between environmental variables and chironomid populations is difficult and for the most part we must rely on observations and trends from studies such as those described above. However, experimental studies such as that described in Chapter Three can provide more conclusive evidence.

Overall, the evidence points to a strong positive relationship between the abundance of chironomids and high levels of phytoplankton abundance resulting from nutrient enrichment. It is thus imperative that reductions in nutrient loading to wetlands form an integral part of a general wetland restoration program. Other aspects of lake restoration, such as replanting of fringing vegetation, should also help to reduce nuisance midge problems by reducing water temperatures within the littoral area, encouraging a more diverse invertebrate community and suppressing algal growth by increasing the tannin content of the water.

Chapter Three

The Relationship Between Algal Blooms and Chironomid Production in Perth Wetlands

An experiment carried out by Debbie Kirby

INTRODUCTION

Initial research into chironomid problems in Perth wetlands suggested that the density of larval chironomids increases in response to nutrient enrichment and algal blooms (Davis *et al.* 1988) . The most eutrophic wetlands (eg. North Lake, Bibra Lake and Lake Monger) supported very high densities of larval chironomids and experienced severe problems with adult midges. Although the link between high nutrients, algal blooms and midge problems was apparent, further work was required to verify this relationship. This was undertaken by Debby Kirby during 1990/91 as an Honours project at Murdoch University. The results of this project are summarised here and described in full in a separate Honours thesis which is available from the Murdoch University Library.

BACKGROUND

Eutrophication is a process in which increased nutrient input into lakes causes them to become highly productive, in terms of the abundance of organic material. The increase in production is often at the expense of diversity as many organisms are unable to tolerate the decline in water quality. Although eutrophication can occur naturally as a wetland ages, the process is accelerated as a result of anthropogenic activity in the form of industry, agriculture and horticulture (Vollenweider 1989).

Most nutrients enter a wetland through surface runoff or groundwater but some may enter with rainfall. In wetlands, nutrients can be found in three components, in the water column, sediments or biomass. The majority of nutrients are bound to the sediments, a small amount is dissolved in the water column and the remainder is bound up in the biota. Nutrient availability can regulate primary production. The availability of nutrients is controlled, both externally and internally, through runoff, sediment release and the release upon death of the biota. In general, as nutrients in a wetland increase in concentration, primary production will also rise (Lugo 1982). High levels of primary production are often visible as algal blooms, floating mats of algae and macrophytes and submerged macrophyte and algal agglomerations (Vollenweider 1989).

The most important nutrients with regard to eutrophication are phosphorus and nitrogen as both are essential for cell growth and reproduction (Pick and Lean 1987; Seale *et al.* 1987). Dissolved phosphorus, usually in the form of soluble reactive phosphorus, is the major source of this nutrient for phytoplankton growth and is taken up by cells via diffusion (Reynolds 1984). Soluble reactive phosphorus is released by the sediment and is usually found in surface runoff.

Nitrogen is available to algae in various forms including nitrate, nitrite, ammonium ions, urea and other free amino acids and peptides. Some blue-green algae are able to utilize atmospheric nitrogen via the mechanism of nitrogen fixation. Nitrogen is rarely limiting to algal growth because low nitrogen conditions will favour the occurrence of species that can produce their own (Edmondson and Lehman 1981).

Increased algal abundance can result in the greater abundance and growth rates of some macroinvertebrates, and secondary production in general, as a consequence of the abundant food supply (Hershey *et al.* 1988). Previous studies of the fertilization of entire lakes have indicated that a relationship exists between chironomid larval density and phytoplankton concentration (Aagaard 1982; Fairchild *et al.* 1989 and Welch *et al.* 1988). These experiments were all performed in natural lakes that already contained chironomid populations. The major objective of this study was to investigate the effect of nutrient-enhanced phytoplankton growth on chironomid density in experimental ponds.

METHODS

Experimental design and pond establishment

Six experimental ponds were created within plastic lined water tanks (1.8m in diameter and 90cm in depth) in an open area on the northeastern side of the Murdoch University campus (Fig.3.1 C). The base of each tank was covered with white, washed sand to a depth of approximately 14 cm. The ponds were then filled with tap water to a total depth (including sand) of 80 cm. The use of washed sand and tap water ensured that all ponds were initially free of chironomid larvae and low in nutrients. The ponds were then left to age for 4 weeks to allow evaporation of chlorine and other compounds present in tap water. The water level in all tanks was maintained at the initial level throughout the study by a weekly addition of small amounts of tap water.

Following aging, three ponds were chosen at random for treatment (i.e. the addition of nutrients and algae) while the remaining three acted as controls. The nutrients and trace elements added were in the form of the soluble commercial plant food, Aquasol® . This particular fertilizer was chosen as it contained a total nitrogen to total phosphorus ratio of 5:75 which is low enough to promote the growth of Cyanophyta rather than Chlorophyta

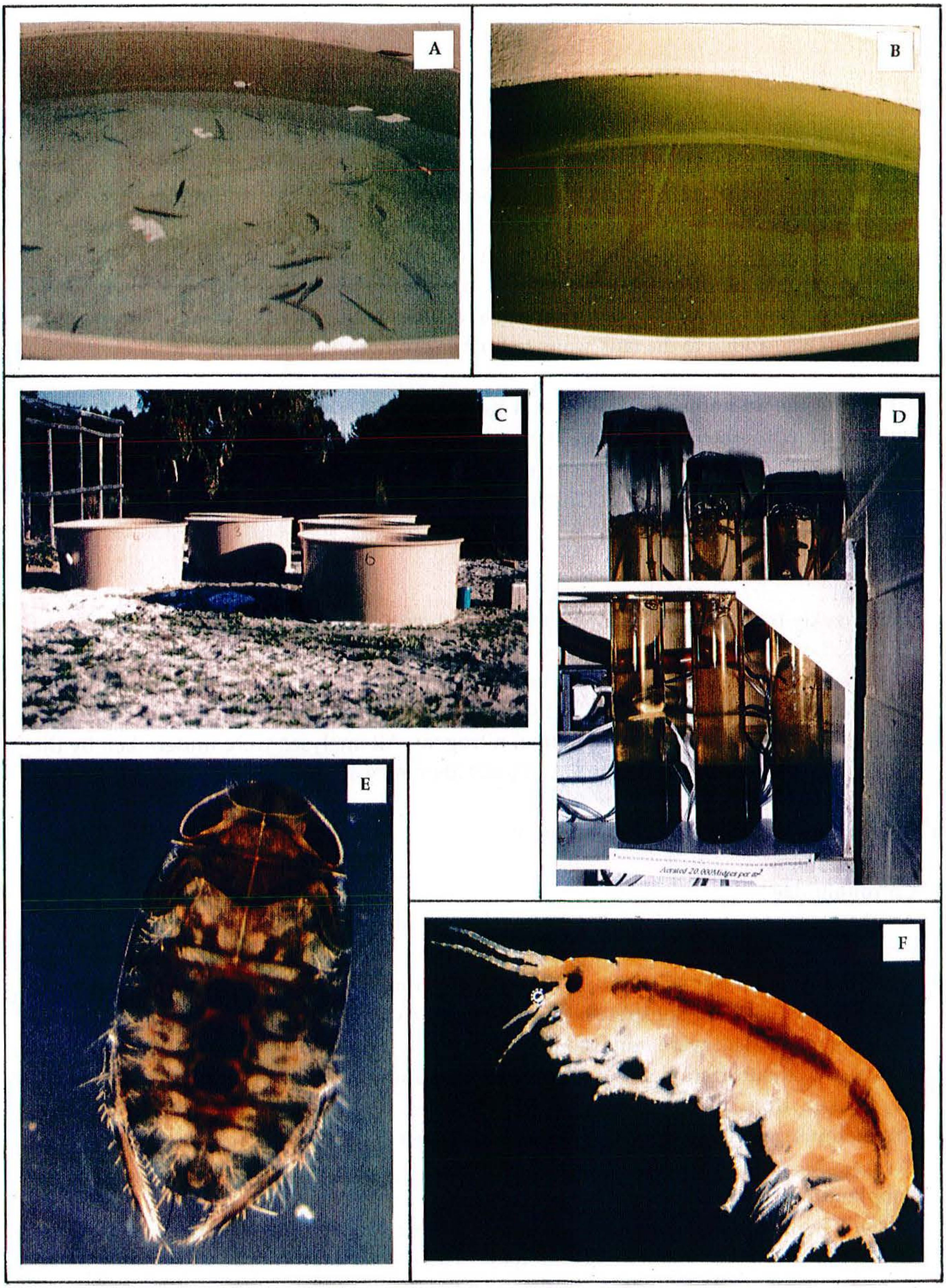

Figure 3.1 Photographs showing: the amount of algal growth in the control ponds and enriched ponds, respectively (A&B); the fiberglass tanks used in the study to determine the relationship between algal blooms and chironomid abundance (C); columns used to examine the effects of chironomids on nutrient release from sediment (D); two of the macroinvertebrates used in the bioassays to determine the toxicity of Sumilarv (pyriproxyfen) to non-target organisms, the juvenile corixid, *Micronecta robusta* and the amphipod, *Austrochiltonia subtenuis* (E&F).

(McQueen and Lean, 1987). Blue-green algae were desired because they are the common bloom-forming algae in wetlands where problems with nuisance midges have been experienced. Sufficient Aquasol® (12.6g per pond) was added to the three treatment ponds to give an initial total phosphorus concentration of 300 ug/l. This level of phosphorus was decided upon as recent studies of Perth wetlands have indicated that similar concentrations of total phosphorus are usually associated with blue-green algal blooms and nuisance midge plagues at wetlands (Fig. 2.28). An iron solution (Borowitzka 1988) was also added to the tank water to further promote algal growth.

Following the addition of nutrients and trace elements, the three enriched tanks were inoculated with algal cells by adding one litre of unfiltered water from North Lake. Aquasol ® (12.6g per addition) was then added to these ponds on a weekly basis until self sustaining blooms were present. This occurred after approximately 12 weeks.

Physico-chemical variables

On a weekly basis for the duration of the experiment, temperature, pH, conductivity and dissolved oxygen concentrations were measured in all tanks. Water samples were taken from each tank, and filtered prior to analyses of soluble reactive phosphorus, ammonia, nitrate-nitrite and chlorophyll-*a*. A 200 ml water sample was collected and filtered for the analysis of total phosphorus and total nitrogen. All analyses were undertaken by the Centre for Water Research Nutrient Analysis Laboratory at Murdoch University.

Invertebrate sampling and identification

Chironomidae

A substrate sampler consisting of a plastic screw top jar (diameter=7 cm) attached to a wooden handle was used to sample the density of Chironomidae in each pond. The jar was inserted into the substrate to a depth of approximately 3cm and dragged for a distance of 70 cm. The substrate collected in the jar was taken to the laboratory where all chiromonids present were removed and preserved. The sand was then returned to the ponds so that substrate depth remained constant throughout the study. Two samples were taken from each pond every week for the duration of the experiment. The chironomids collected were identified to species level .

Zooplankton

A plankton net was used to sample zooplankton in each pond on a weekly basis. The net was pulled across the diameter of each tank. The zooplankton caught were counted and identified to genus. Density was calculated on the basis of number per litre.

Statistical analyses

Differences in chironomid density and concentrations of total phosphorus, chlorophyll-*a* and dissolved oxygen, in the enriched and control ponds, were compared over time by repeated measures analysis of variance (ANOVA). Prior to the analyses all data were tested for heteroscedasticity by Cochrans C test and those variables showing significant heterogeneity were log transformed [$\log_{10}(n+1)$]. Where Cochrans C test showed that variance was still heterogenous after transformation the results of the ANOVA were only considered significant at the 0.01 level of probability (Loneragan *et al.* 1989).

RESULTS

Physico-chemical Variables

Temperature: A gradual increase in maximum and minimum temperatures occurred in all ponds over time (Fig. 3.2a,b). There was little variation in temperature between the control and the enriched ponds.

Conductivity: All six ponds acted similarly with regard to conductivity (Fig. 3.3) with a gradual increase evident over the course of the study.

pH: Weekly readings revealed major differences in pH between the control and enriched ponds (Fig. 3.3). After the four week settling period and the addition of nutrients to the enriched ponds, the pH levels recorded were considerably higher than those of the control ponds. Whilst the pH in the control ponds did not exceed 8, a maximum of 9.7 was recorded in the enriched ponds.

Dissolved oxygen: There was no evidence of stratification of dissolved oxygen occurring within the ponds at any time throughout the study. The concentration of dissolved oxygen in the control and enriched ponds displayed the same trends over time (Fig. 3. 3). However dissolved oxygen concentrations in the enriched ponds were significantly higher ($P < 0.01$) than in the control ponds in the period after the addition of nutrients. A difference of almost 8 μg/l was recorded during week 8.

Nutrients: Total phosphorus concentrations were significantly lower ($P < 0.01$) in the control ponds than in the enriched ponds (Figs 3.4a,b) after the addition of nutrients. The highest concentration of total phosphorus recorded in the control ponds was 41μg/L. In the enriched ponds, total phosphorus concentrations were never below 100μg/L after the initial addition of nutrients, and were recorded as high as 1288μg/L.

Low concentrations of soluble reactive phosphorus compared to total phosphorus indicate

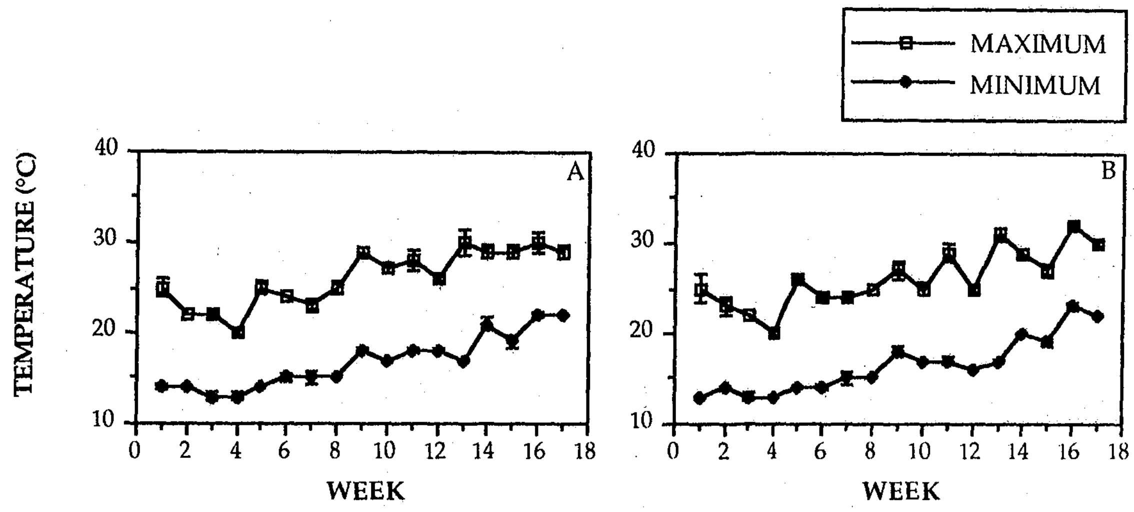

Figure 3.2 Changes in the minimum and maximum water temperatures recorded in the control (A) and enriched (B) ponds.

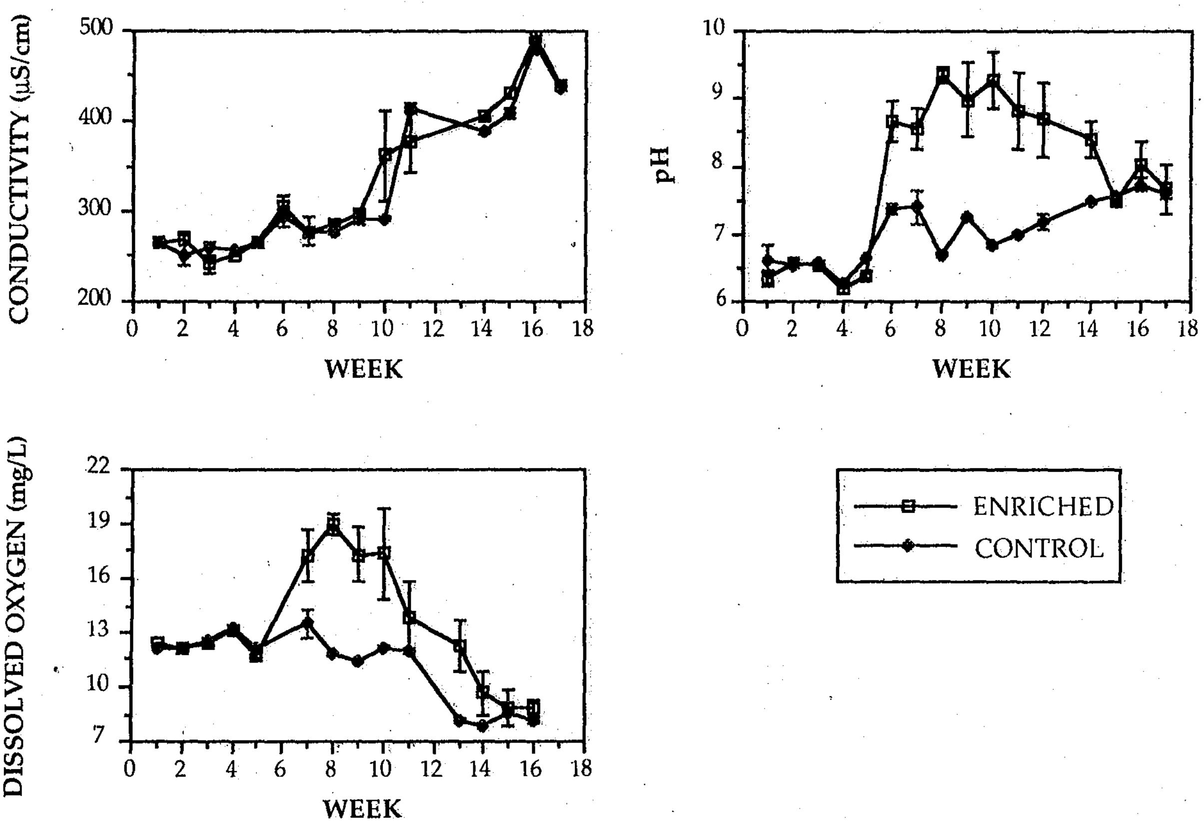

Figure 3.3 Changes in conductivity, pH and the concentration of dissolved oxygen in the control and treated ponds during the experiment.

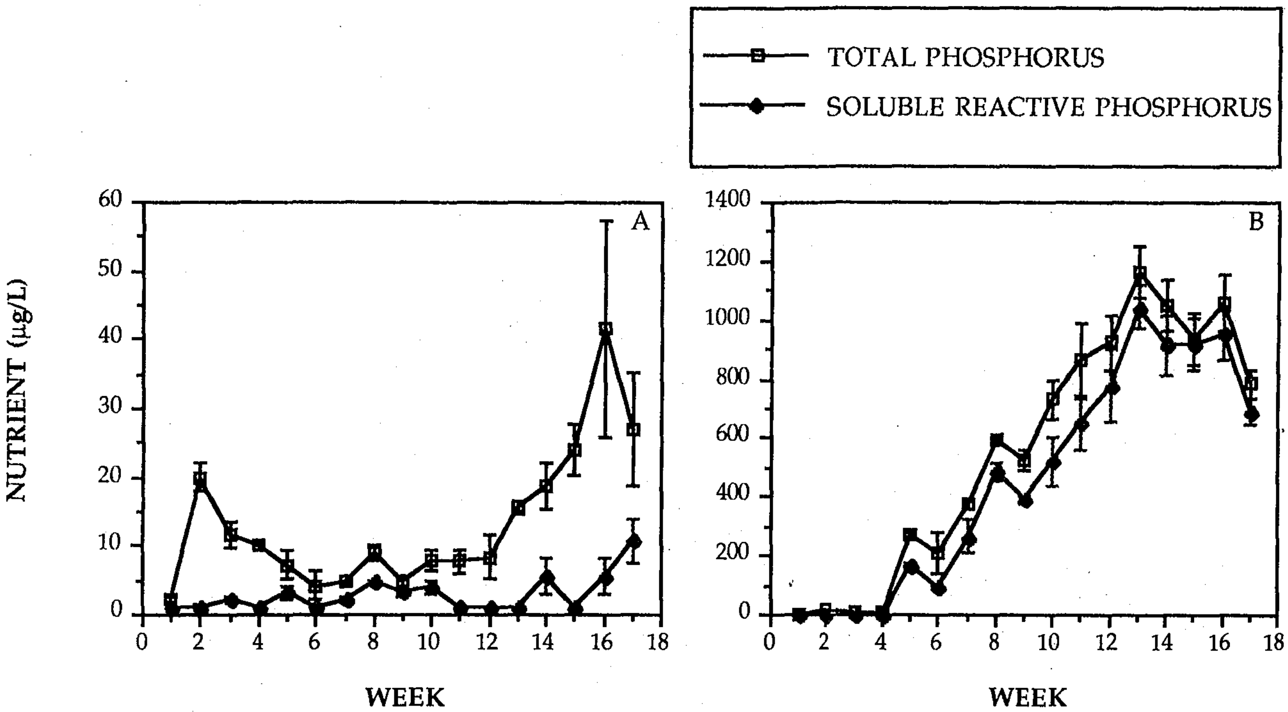

Figure 3.4 Changes in the concentration of total phosphorus and soluble reactive phosphorus (mean ± s.e.) in the control (A) and enriched ponds (B).

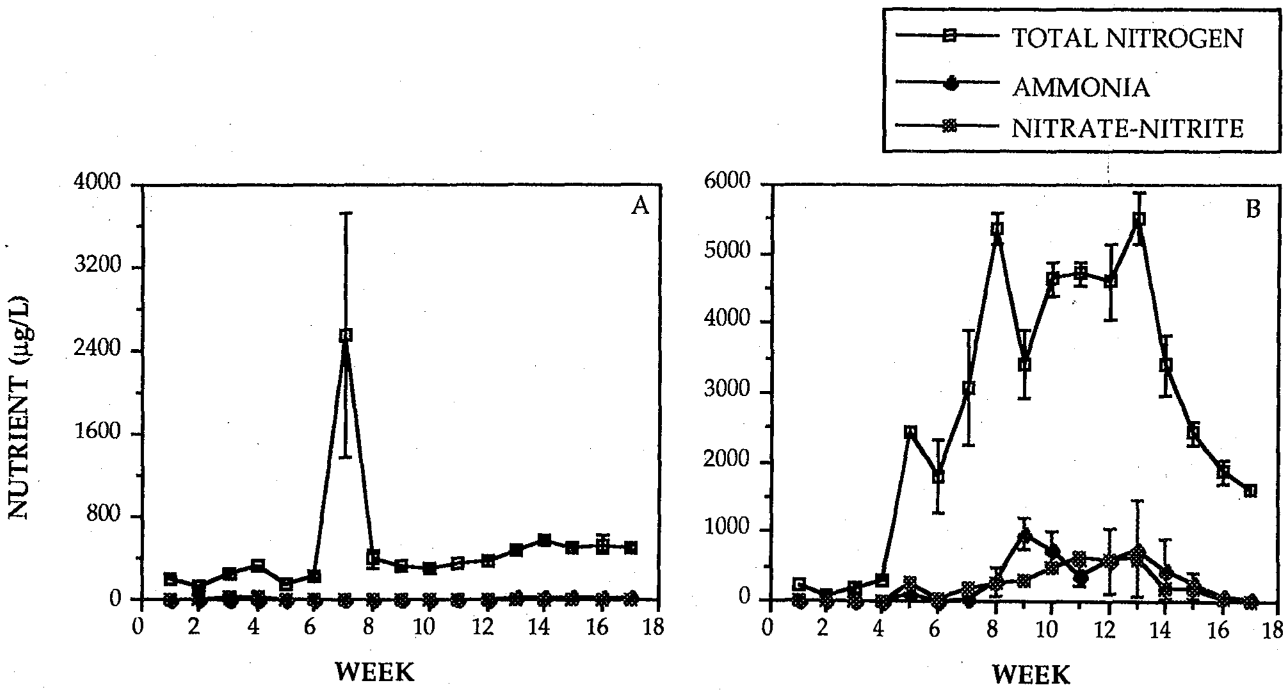

Figure 3.5 Changes in the concentration of total nitrogen, ammonia and nitrate-nitrite (mean ± s.e.) in the control (A) and enriched ponds (B).

that much of the phosphorus in the control ponds was in the form of organic material (Figs 3.4a). By contrast, most of the total phosphorus present in the enriched ponds was in the form of soluble reactive phosphorus (Fig. 3.4b).

The concentration of total nitrogen was much higher than the sum of the inorganic compounds in both of the pond types (Figs. 3.5a,b), indicating that a large proportion of the total nitrogen present in the ponds was in the form of organic nitrogen. The mean concentration of nitrogen was generally much lower in the control ponds than in the enriched ones. The maximum total nitrogen recorded in the control and in enriched ponds were 2,500 μg/L and 5,400 μg/L, respectively.

Algae: The concentration of chlorophyll-*a* was significantly higher ($P< 0.01$) in the enriched ponds than in the control ponds (Figs 3.6a,b) after the addition of nutrients. The maximum concentration of chlorophyll-*a* recorded in the enriched ponds was 169 ± 115 μg/L. By contrast, the maximum concentration in the control ponds was 8 ± 6 μg/L. Differences between the amount of algal growth in the control and enriched ponds are illustrated in Figure 3.1A and B.

Although the amount algal growth in the control ponds was low, increases in concentration of chlorophyll-*a* were positively associated with increases in the concentration of phosphorus in those ponds (Fig 3.7a). This relationship was not apparent in the enriched ponds (Fig. 3.7b) where chlorophyll-*a* concentration declined after week 11 despite further increases in total phosphorus.

Algal species recorded in the tanks included the green unicellular genera *Ankistrodesmus* and *Scenedesmus* and the green filamentous alga *Cladophora*. The main bloom-forming genera which occur in many Perth wetlands, *Microcystis* and *Anabaena*, were not recorded in the ponds during this study.

<u>Colonization of the experimental ponds</u>

Chironomidae

The density of larval chironomids was significantly higher ($P<0.01$) in the enriched ponds than in the control ponds (Fig 3.8a) after the four week settling period and the addition of nutrients. The maximum density of chironomids recorded in the enriched ponds was 11200 ± 2852 larvae/m^2 compared with 287 ± 197 larvae/m^2 in the control ponds.

An overall increase in concentration of chlorophyll-*a* in the control ponds was accompanied by a corresponding rise in chironomid density (Fig 3.9a). In the enriched ponds, chlorophyll-*a* concentrations increased from 0 to 168μg/L over a six week period after the initial nutrient addition. As algal abundance increased, so did the density of chironomids. Chironomid density continued to remain high despite a decline in the

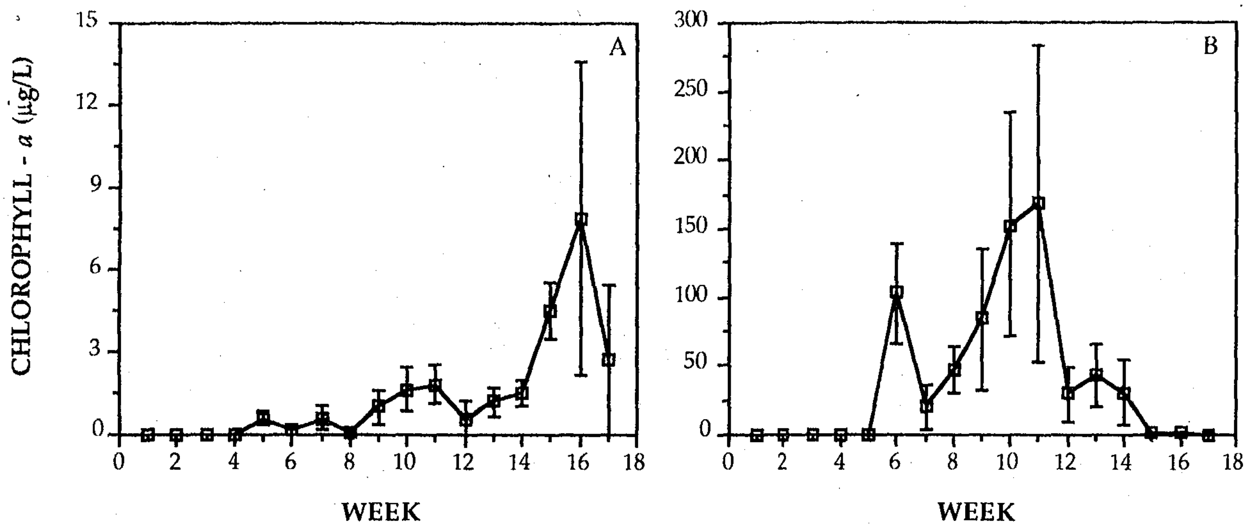

Figure 3.6 Changes in the concentration of chlorophyll-*a* (mean ± s.e.) in the control (A) and enriched (B) ponds.

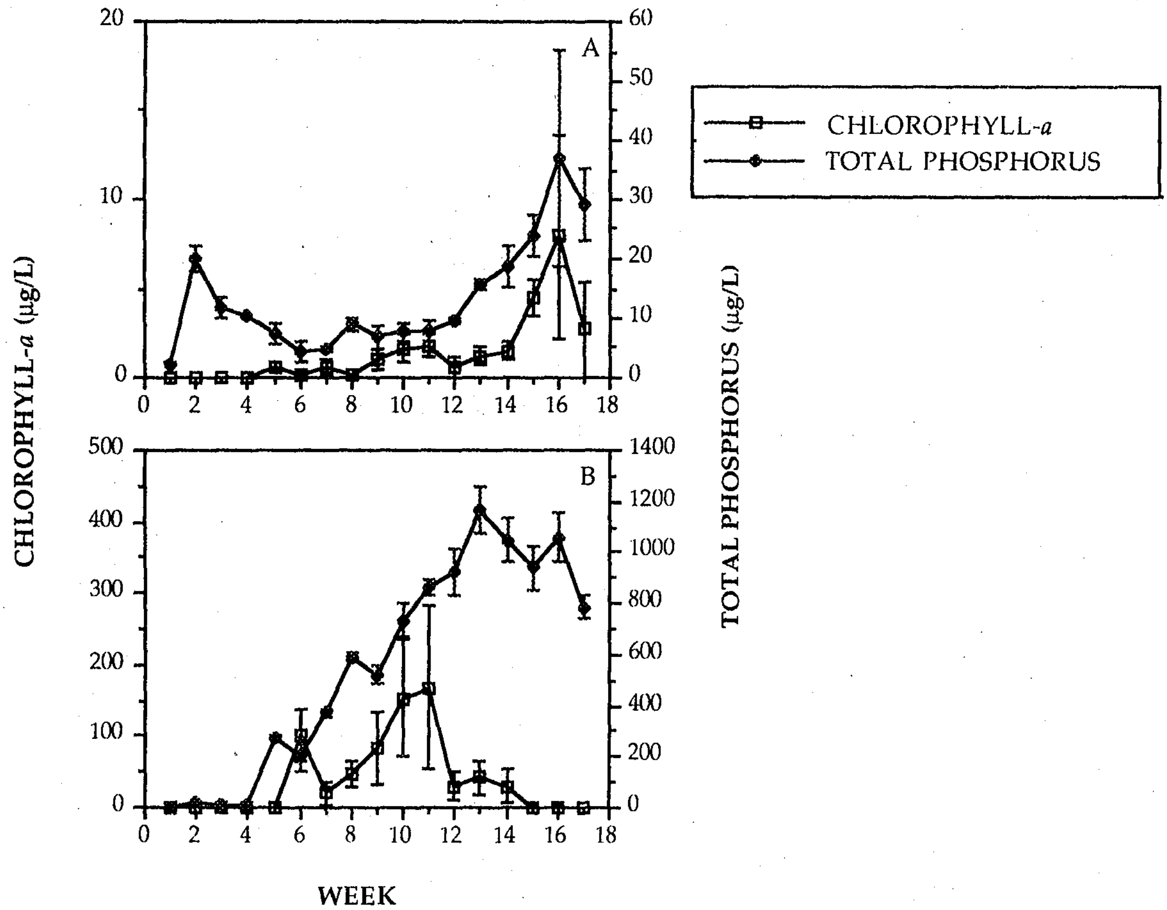

Figure 3.7 Changes in the concentration of chlorophyll-*a* and total phosphorus (mean ± s.e.) in the control (A) and enriched (B) ponds.

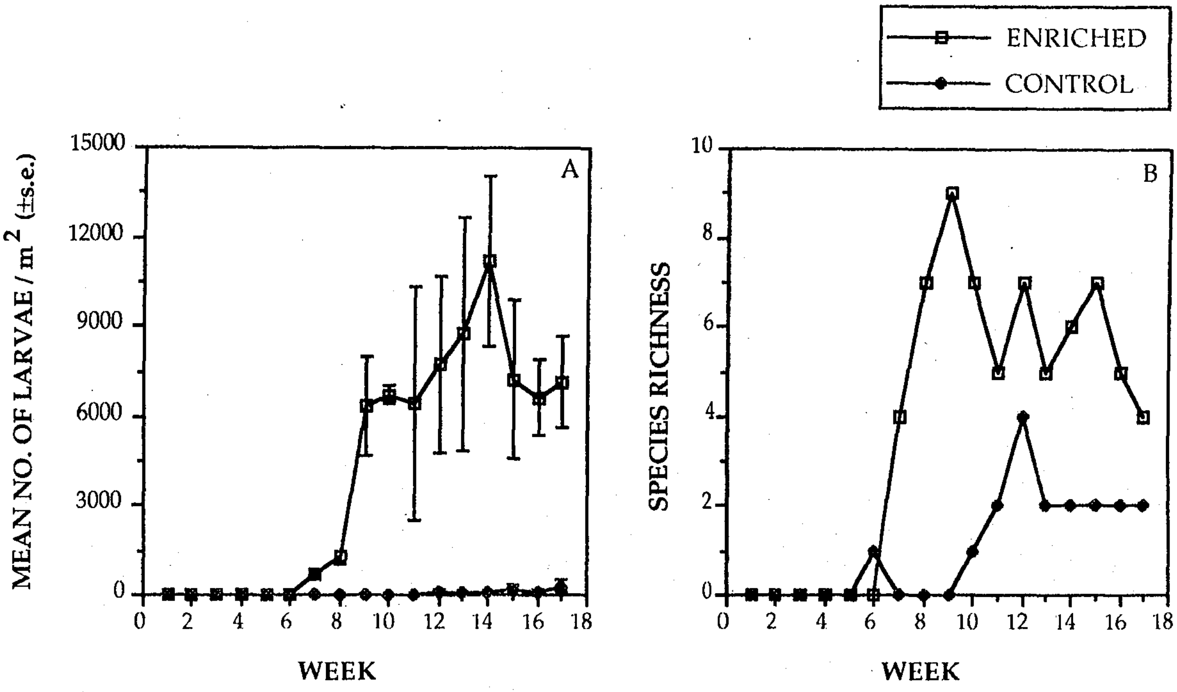

Figure 3.8 Changes in the density of chironomid larvae (A) and in the number of chironomid species (B) present in the control and enriched ponds during the experiment.

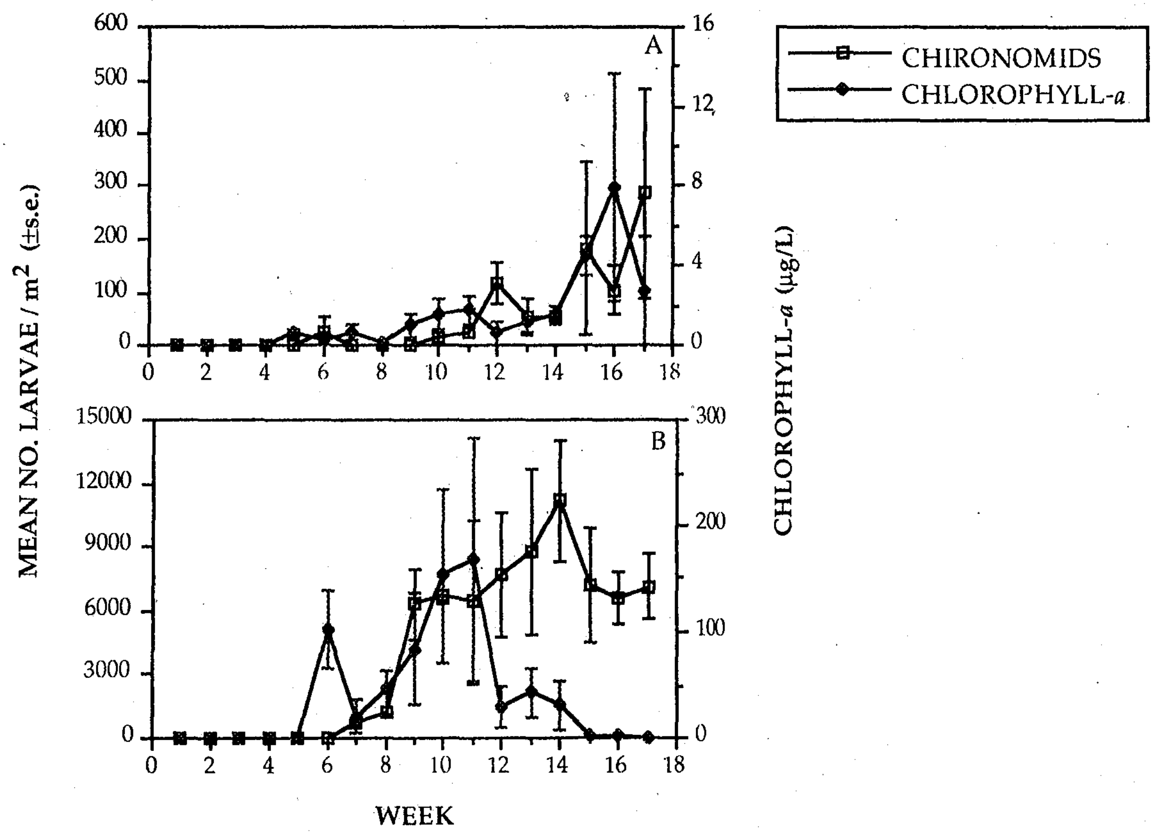

Figure 3.9 Changes in the density of chironomid larvae and the concentration of chlorophyll-*a* in the control (A) and enriched (B) ponds. Note the different Y-axis.

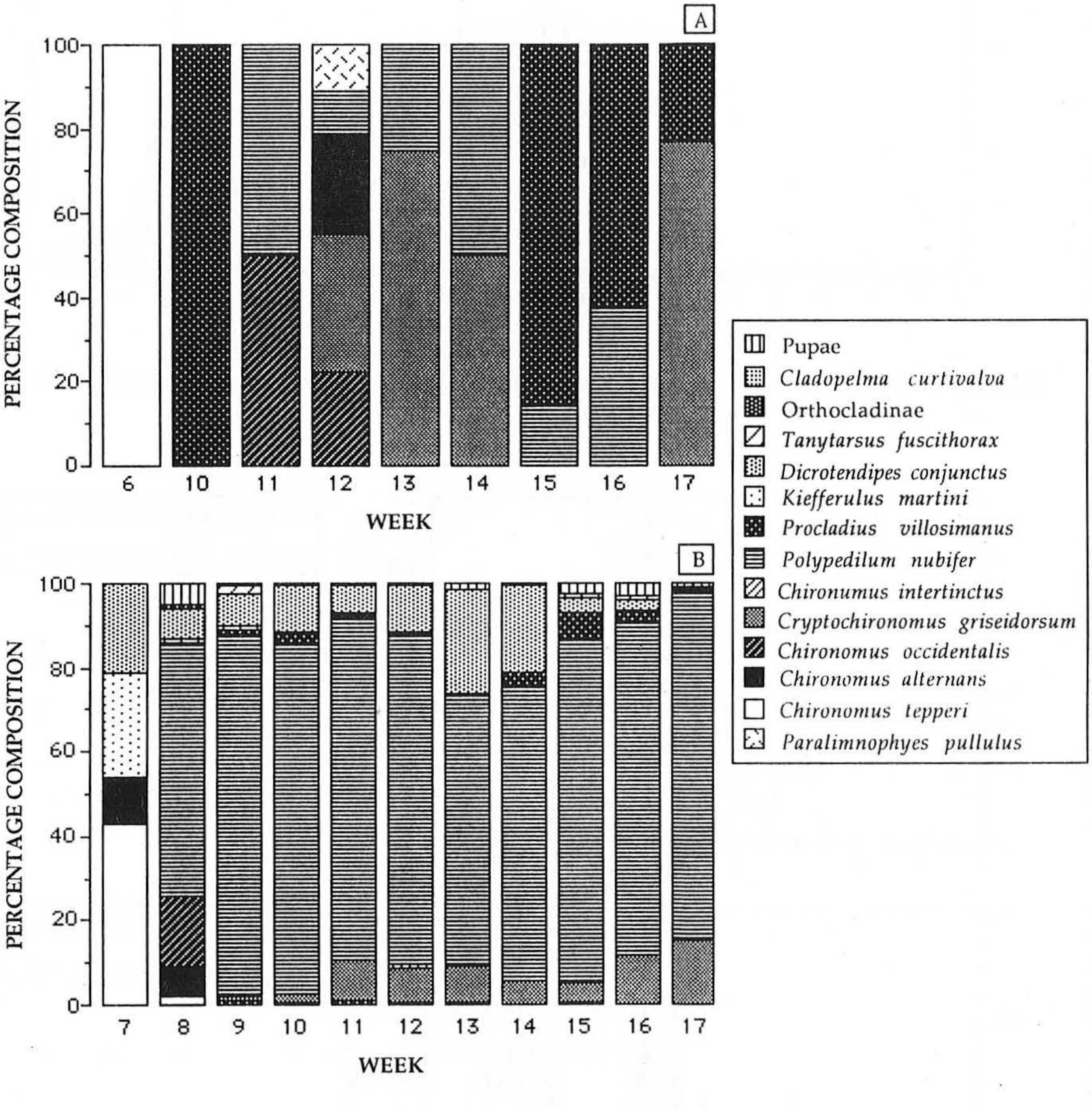

Figure 3.10 The composition of the chironomid species in the control (A) and enriched ponds (B) during the experiment.

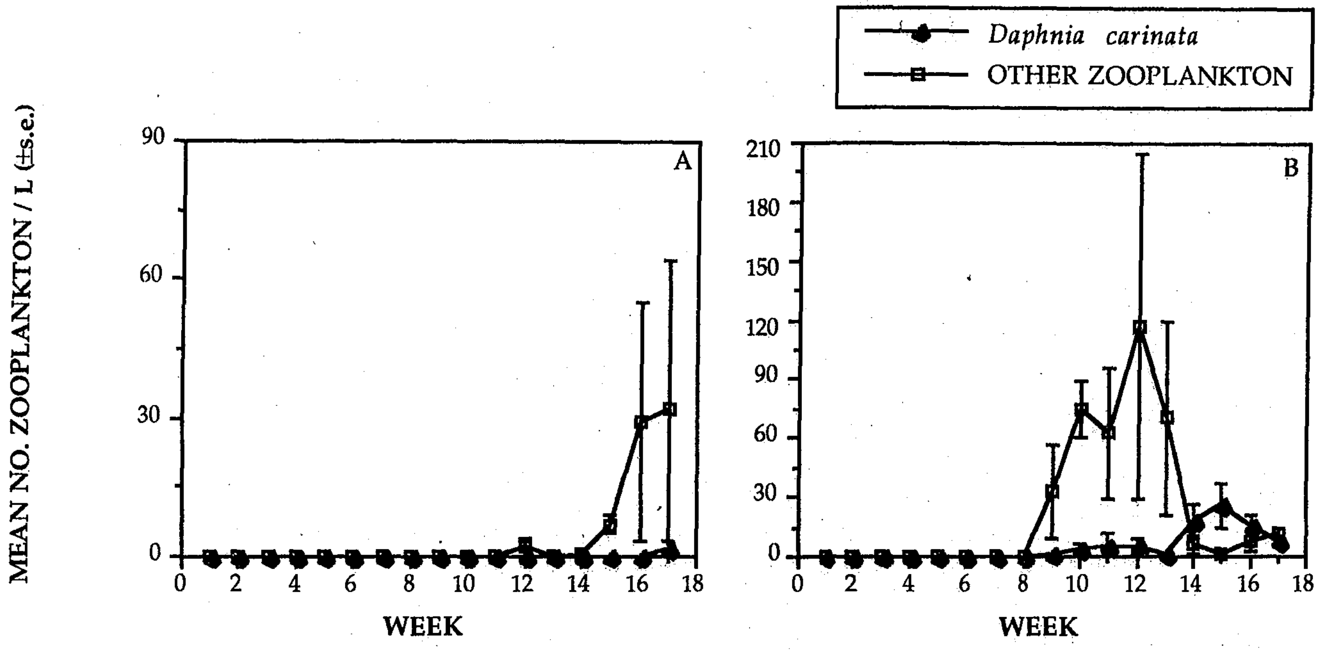

Figure 3.11 Changes in the density of zooplankton in the control (A) and enriched (B) ponds during the experiment.

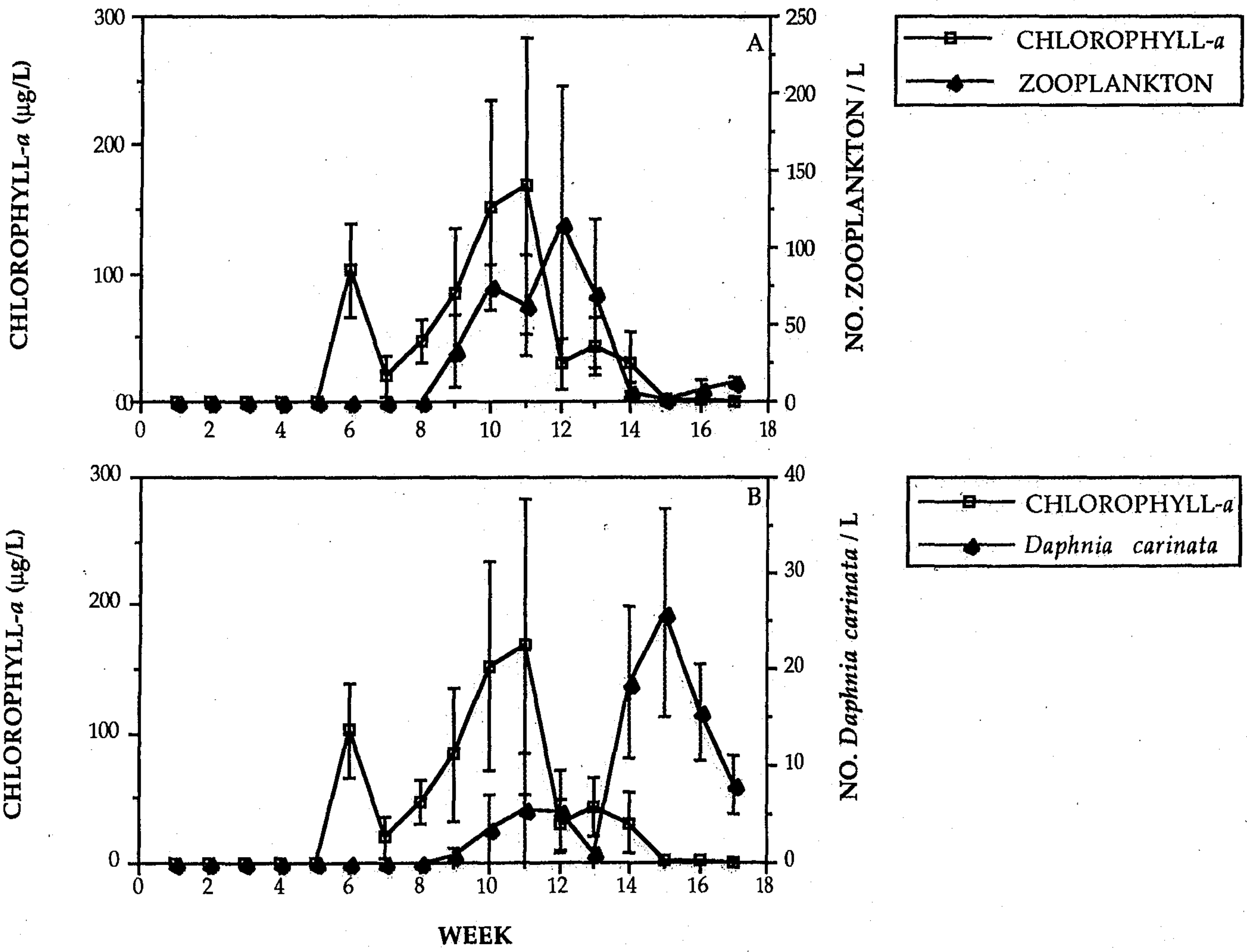

Figure 3.12 Changes in the concentration of chlorophyll-*a* with changes in the densities of zooplankton other than the cladoceran, *Daphnia carinata* (A) and *D. carinata* (B) in the enriched ponds.

concentration of chlorophyll-*a* after week 11 probably as a result of the abundant supply of decaying algae in the ponds (Fig 3.9b).

A greater number of chironomid species were recorded in the enriched ponds than in the controls (Fig. 3.8b) and, with the exception of the chironomid *Paralimnophyes pullulus*, all of the species recorded in the controls were common to the enriched ponds. The chironomids, *Chironomus tepperi, Kiefferulus martini Chironomus alternans* and *Dicrotendipes conjunctus* initially colonized the enriched ponds but as the experiment progressed, *P. nubifer* became increasingly dominant with the relative abundance of this species accounting for as much as 80% of the chironomids present in the samples (Fig. 3.10a). No single species dominated the larval composition in the control ponds, throughout the experiment (Fig. 3.10b)

Zooplankton

The density of the cladocerans, *Daphnia carinata*, were scored separately from other zooplankton because they formed the largest, most visible species. Overall, the density of zooplankton (rotifers, copepods and other cladocerans) was higher in the enriched ponds than in the controls (Fig 3.11a,b). Increases in zooplankton abundance in the enriched ponds appeared to be associated with a marked decrease in chlorophyll-*a* (Fig. 3.12a). The decline in the abundance of algae was followed by a decrease in the density of other zooplankton and a subsequent increase in the abundance of *D. carinata* (Fig. 3.12b).

DISCUSSION

Physico-chemical variables

Water temperature and conductivity were similar in the control and enriched ponds. Temperature in all of the ponds rose as the study progressed as a result of the increase in ambient air temperatures associated with the change of season from spring to summer. Despite regular additions of tap water, conductivity also increased gradually in all ponds due to evaporative processes . Those variables that are affected by nutrient concentration and algal growth differed markedly between pond types. pH and dissolved oxygen were much higher in the enriched ponds than the controls probably due to the presence of algal blooms. pH tends to increase because algal photosynthesis results in a decrease in the concentration of hydrogen ions in solution.

Development of Algal Blooms

Many researchers have found that high levels of nutrients, particularly phosphorus, in the water column, result in increased algal biomass (Fairchild *et al.* 1989; Pick and Lean 1987; Seale *et al.* 1987 and Tilman *et al.* 1982). A similar observation was recorded in this study where a mean chlorophyll-*a* concentration of 100 μg/l was recorded in the enriched ponds just one week after the addition of a small amount of algal material and nutrients. Although no algae or nutrients were added to the control ponds, very low concentration of nutrients were present and algal abundance in these ponds increased gradually over time. Nutrients may have become available from the breakdown of organic matter such as leaf litter from nearby vegetation. The decline in chlorophyll-*a* concentration in the enriched ponds in week 12 may have been the result of intense grazing by zooplankton.

Blue-green algae did not form blooms in any of the ponds. This was despite the favourable total nitrogen to phosphorus ratios in the enriched ponds. One explanation for this result is that water temperatures were not high enough. Blue-green algae are more abundant than green algae in warmer water (McQueen and Lean 1987). The optimum temperature for growth of *Microcystis* is 27-35 $^{\circ}C$ (Robarts and Zohary 1987). Although temperatures within the lower half of this range were recorded, a longer period of temperatures above 30$^{\circ}C$ may have been required.

The Response of Invertebrates to Algal Blooms

Chironomidae

The results of this study suggest that the density of larval chironomids will increase in response to algal blooms. Other researchers have also found a relationship between primary production and density of chironomids (Aagaard 1982; Fairchild and Lowe 1984; Fairchild *et al.* 1989; Welch *et al.* 1988 and Winterbourn 1990). The greatest increase was observed by Fairchild and Lowe (1984) who found that there were approximately 16 times as many chironomids in fertilized than non fertilized areas. By contrast, an average of 700 times more chironomids were recorded in the enriched ponds than in the controls of this study.

The relationship between algal abundance and chironomid density is complex. The algae provides a food source which can support large populations and also provides protection from predators by reducing visibility (Fairchild *et al.* 1989). Chironomids are thought to feed on both live and decaying algae. In this study some chironomids were observed actively grazing on the filamentous algae attached to the pond walls and live algal cells were also found in the gut contents of some larvae . However, some larvae are known to gain much of their nutrient requirement from the bacteria which colonize both live and decaying algal cells rather than the algal cells themselves (Johannsson and Beaver 1983). Thus the algae may be either a direct or indirect food source for the chironomids. Regardless of this, the larvae produced when a food source is abundant will be larger, healthier and more likely to survive to the adult stage than if the food source is limiting (Welch *et al.* 1988).

The low species richness recorded in the control ponds was probably due to their oligotrophic nature. The ponds contained very little nutrient and there was a much lower concentration of algae in these ponds than the enriched ponds. The limited food supply may have inhibited the establishment of all but a few chironomids in these ponds.

All ponds were initially colonized by a group of chironomid species whose numbers subsequently declined within a short period. Previous research has also recorded the presence of pioneer or fugitive species which rapidly invade new water bodies and build up high population densities. However they are generally poor competitors whose population size declines rapidly with the appearance of other chironomid species (Cantrell and McLachlan 1977). The colonizing species in the enriched ponds were *C. tepperi*, *K. martini*, *C. alternans* and *D. conjuctus* All of these species have been previously recorded in temporary waters, with *C. tepperi* almost exclusively found in new water bodies (Edward 1964). Overall, the most abundant species collected was *P. nubifer*, which is the main pest species associated with Perth wetlands (Davis *et al.* 1990). This species may have been dominant in the enriched ponds for a number of reasons. Firstly, *P. nubifer* larvae are thought to feed by filtering suspended phytoplankton and this was abundant. In addition, the larva of this species has a very short life cycle during the warmer months (three to four weeks) and this can result in the production of six to seven generations of *P. nubifer* per year, leading to large population increases. Finally, *P. nubifer* larvae often leave their tubes to swim actively in the water column and this may allow them to search for food more efficiently than other species (Edward 1964).

The major objective of this study was to investigate the relationship between algal blooms and midge densities given that the relationship between nutrient enrichment and algal blooms is already well established. For this reason nutrients and algae were added to three treatment tanks. However with hindsight we realise that all tanks, both treatment and controls, should have been inoculated with algae because we could not be sure that a small number of early instar chironomids, zooplankton eggs and adults were not also present with the algae. We assume that their effect, if any, would have been small and the temporal changes in species composition recorded in the ponds suggested that colonization was occurring from waterbodies elsewhere. Adult midges were also frequently seen in the vicinity of the ponds and egg sacs were observed in the enriched ponds. Further research is needed to determine whether females preferentially lay eggs in eutrophic waters or whether egg and larval survival is greater in enriched compared to non enriched waters.

Zooplankton

The high densities of zooplankton recorded in the enriched ponds compared to the controls indicates that these organisms also display a positive response to algal blooms. These results support those of previous researchers (Aagaard 1982; Fairchild *et al.* 1989) who have documented an increase in total zooplankton biomass with increased primary production.

Although other factors may have been responsible (eg. nitrogen limitation), chlorophyll-*a* concentrations in the enriched ponds appeared to decrease in response to the increased abundance of zooplankton and greater grazing pressure. A subsequent decline in non-daphnid zooplankton numbers was observed and this was accompanied by increase in the abundance of the cladoceran, *Daphnia carinata* which may have maintained the high grazing levels.

Daphnid species have been found to reduce the concentration of phytoplankton in a water body (Lynch and Shapiro 1981; Porter 1976) and have some preference towards *Scenedesmus* (Lampert and Taylor 1985), the most abundant algal genera in the enriched ponds. A decline in phytoplankton abundance in enclosed water bodies, known as the 'clear water' phase, is often associated with the increased abundance of daphnid cladocerans. This decrease may initiate succession towards an abundance of large filamentous algae because these cannot be grazed by large cladocerans (Vanni and Temte 1990). At the conclusion of this study the filamentous *Cladophora* was the dominant algal species in the enriched ponds.

CONCLUSIONS

The aim of this study was to determine whether nutrient-enhanced phytoplankton growth is associated with, and hence a possible cause of, high densities of chironomid midges. The results obtained from the experimental ponds indicate that a positive relationship exists between chlorophyll-*a* concentration and abundance of chironomids. The enriched ponds contained a significantly higher abundance of both algae and chironomids.

Nutrient enrichment or eutrophication also appeared to influence the abundance of zooplankton with with greater densities being present in the enriched ponds than the control ponds.

The results of this study have far-reaching implications for the management of wetlands which have previously experienced problems with massive swarms of midges. The relationship found between phytoplankton abundance and chironomid density indicates that if the development of large algal blooms can be prevented, then the incidence of nuisance swarms of midges may be reduced.

The observations of zooplankton density made during this project indicated these organisms may be important in reducing algal abundance, which in turn may result in a decrease in chironomid densities. Further information is now needed to determine the feasibility of using zooplankton as a means of controlling algal blooms in wetlands where nutrient impacts are difficult to reduce or control. However, blue-green algae may not be the preferred food source of zooplankton, particularly *Daphnia* species and this may limit the effectiveness of this means of control. Further studies are required to determine whether the larval chironomids are eating living algal cells for their nutritional value or whether they are detritivores that take in the cells as a means of ingesting any bacterial film associated with them.

Chapter Four

The Influence of Chironomids on the Release of Nutrients from Wetland Sediments

An experiment carried out by Jan Otto van Erpers Roijaards

INTRODUCTION

A constant exchange of nutrients occurs between lake sediments and the overlying water column. This exchange is mediated by a range of chemical reactions whose equilibria are partly dependant upon a range of prevailing environmental conditions. Several studies have shown that sediment dwelling invertebrates, including chironomids, molluscs and oligochaetes (worms) can also affect the exchange of nutrients between the sediment and the overlying water column (Fukuhara and Sakamoto 1987, Gallep 1978 and Tatrai 1988).

Benthic invertebrates may influence nutrient exchange through the physical disturbance of the upper sediment layers when burrowing, feeding and pupating (known as bioturbation) or as a result of the excretion of nutrient rich waste products. In addition, the construction of tubes in the sediment increases the surface area of the sediment/water interface at which chemical and microbiological processes occur, thus affecting the exchange of nutrients. This burrowing activity can also increase the depth of the oxidised layer of the sediment, which would encourage the adsorption of phosphorus by sediments. However, the use of oxygen by invertebrates for normal physiological processes may have the opposite effect.

Additionally, emergence of invertebrates will result in some loss of nutrients from the whole lake system if the insects move away from the lake before dying. The dry weight of *P. nubifer* adults is 0.0036 ± 0.0001g/10 adults (calculated from 10 samples of 5 males and 5 females (with eggs) each, collected in light traps set at North Lake). This multiplies up to rates of emergence of over 5 kg dry weight/ha/night (assuming an emergence rate of 1400 adults/m^2, as was recorded at Bibra Lake in February 1991).

The production of phytoplankton is partly dependant upon the nutrient content of the water. Chironomids utilize the phytoplankton as a food source and so by increasing sediment nutrient release they may be affecting their own food supply.

The question of whether this process may operate in the sediments of Perth wetlands was addressed by an experiment carried out by Jan Otto van Erpers Roijaards (a student from The Netherlands on a work experience program at Murdoch University) with the assistance of the Murdoch research group.

METHODS

To investigate the effects of chironomid densities upon the release of nutrients from the sediment under laboratory conditions, twelve microcosms (5.5cm x 55cm glass cylinders) containing lake sediment and water were set up. These are illustrated in Figure 3.1D.

The sediment was collected from the littoral area of Bibra Lake as three distinct layers, a loose surface layer with high organic content, and two lower coarse sand layers. These separate layers were transported to the laboratory and frozen at -20°C to kill the existing invertebrates. After thawing each layer was homogenized and samples of each were placed into each column so as to approximately recreate the original strata.

Water from Bibra Lake, was filtered through a 53µm seive and then autoclaved (sterilized) at 109°C for 45 minutes. The water was then filtered again and stored in a black drum at 4°C to prevent algal growth. Half a litre of water was added to each column, taking care not to disturb the sediment structure. The columns were aerated using a bubbler 10cm below the water surface. The columns were then left for five days in a constant temperature room (16 ± 0.5°C) in the dark to stabilise, before the chironomid larvae were added.

Late instar *P. nubifer* larvae were collected from the same area of Bibra Lake as the sediment and water. The larvae were acclimated to test conditions (16 ± 0.5°C) by placing them in an aquarium with room temperature lake water.

Four treatments were employed (0, 6, 12 or 24 larvae per column, corresponding to 0, 5000, 10000 or 20000 larvae/m^2), with three replicates of each. The experiment was carried out under dark conditions to reduce light and thus the growth of algae. The tops of the columns were covered by wire mesh to prevent the escape of emerging adults. The columns were checked daily and dead larvae, floating pupae and emerged adults were replaced by live larvae.

Measurements of pH and dissolved oxygen concentration were taken from each column (approximately 10cm below the water surface) two days and one day prior to the addition of larvae and on six further occasions over a three week period after treatment. On each of these days a 15ml water sample was taken from approximately 10cm below the water surface using a syringe and tube extension. The sample was stored at -4.0°C until analysis for nutrient concentration. After the water sample was removed an equivalent amount of stored lake water was added to each column to restore the original volume.

The samples were analysed for soluble reactive phosphorus (SRP) using an automated phenate method (Murphy and Riley 1962) and for ammonium using the methods of APHA (1985).

The data were analysed using repeated measures analysis of variance to examine the overall effect of treatment level. A univariate analysis of variance (ANOVA) and aposteriori Scheffe tests were then employed to investigate differences between specific treatments, using averaged post treatment data.

RESULTS

After the last water samples were taken the larvae were recovered and counted. Low survival rates between 16 and 22% were recorded, however the time of death could not be determined. The average of the initial and final density of live midge larvae in each treatment were calculated to be zero, 3300, 6100, and 12200 larvae/m^2 and these densities were used as an estimate of the average density of larvae for the duration of the experiment. The treatments will henceforth be referred to as treatments A, B, C and D respectively.

Similar concentrations of dissolved oxygen and pH were recorded for all treatments (Fig. 4.1). Oxygen concentration was more variable before treatment but later stabilized to between 8.5 and 10.5 mg/l. pH was also more variable before addition of larvae but stabilized to between 8.5 and 8.9.

The concentration of soluble reactive phosphorus (SRP) increased in all treatments over the first six days after addition of larvae (Fig. 4.2), although to a greater extent in the treated columns than in the controls. The mean post-treatment SRP concentration for treatments A,B,C and D were 1045 ± 35, 1088 ± 61, 1319 ± 40 and 1371 ± 31 µg/L respectively.

Statistical analysis of this data indicated that there was a significant treatment by time effect ($p<0.05$), indicating that the density of larvae affected the pattern of change in soluble reactive phosphorus concentrations over time. In addition, a pre-treatment vs. post-treatment contrast indicated that the differences between treatments were larger after the addition of larvae than they were before.

To determine which treatments were different, the post-treatment data for each treatment were combined and analysed by a oneway ANOVA. Significant differences ($P<0.01$) were found between the post-treatment SRP concentrations of treatments A vs. D and B vs. D. Treatments A vs. C were significantly different at a lower confidence level ($P<0.05$). No significant differences were found between treatments A and B, C and D or between B and C. Thus, the only significant differences in post-treatment phosphorus concentrations

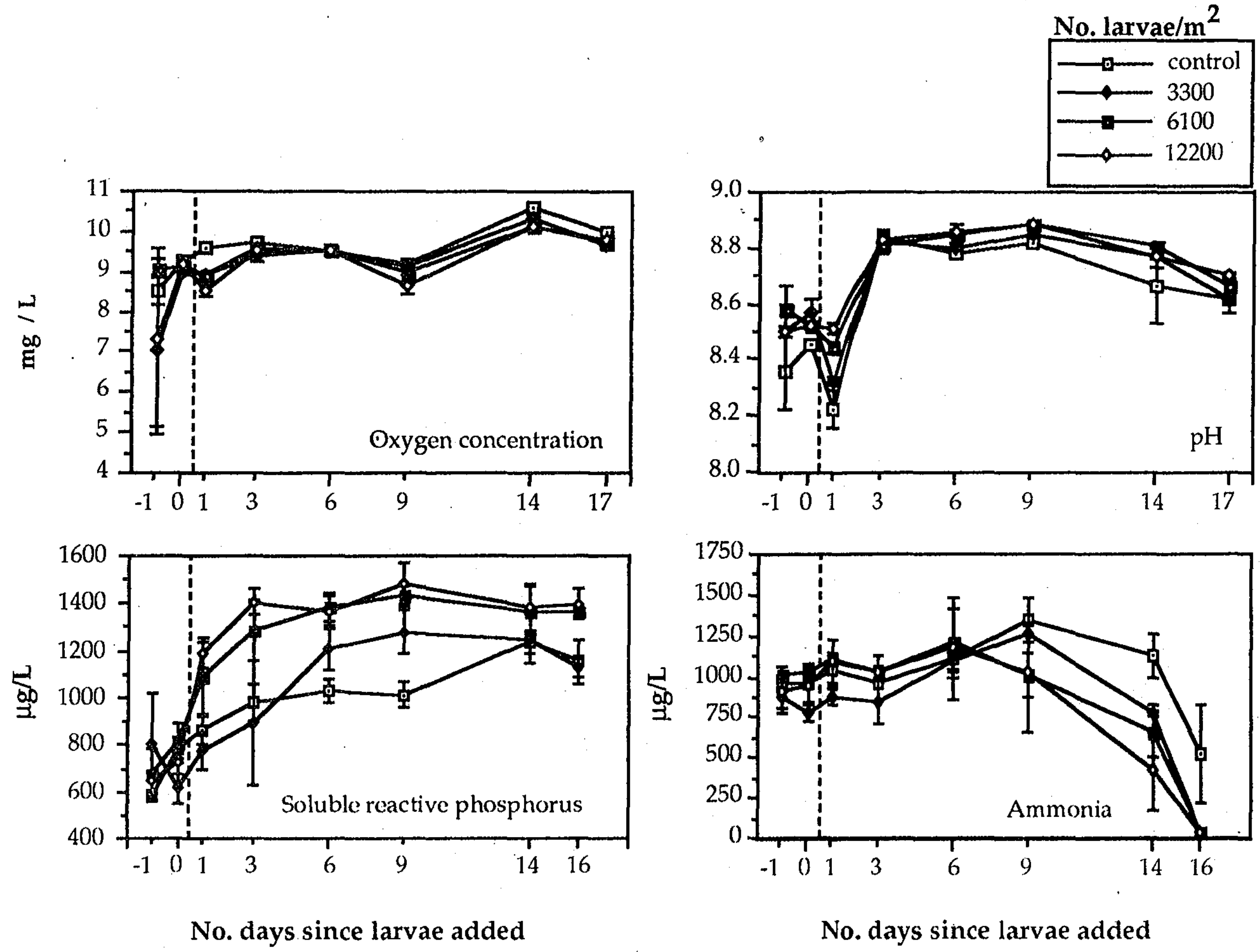

Figure 4.1. Changes in oxygen concentration, pH and concentrations of soluble reactive phosphorus [SRP] and ammonia during the column nutrient release experiment. Dashed line represents time of treatment (addition of larvae).

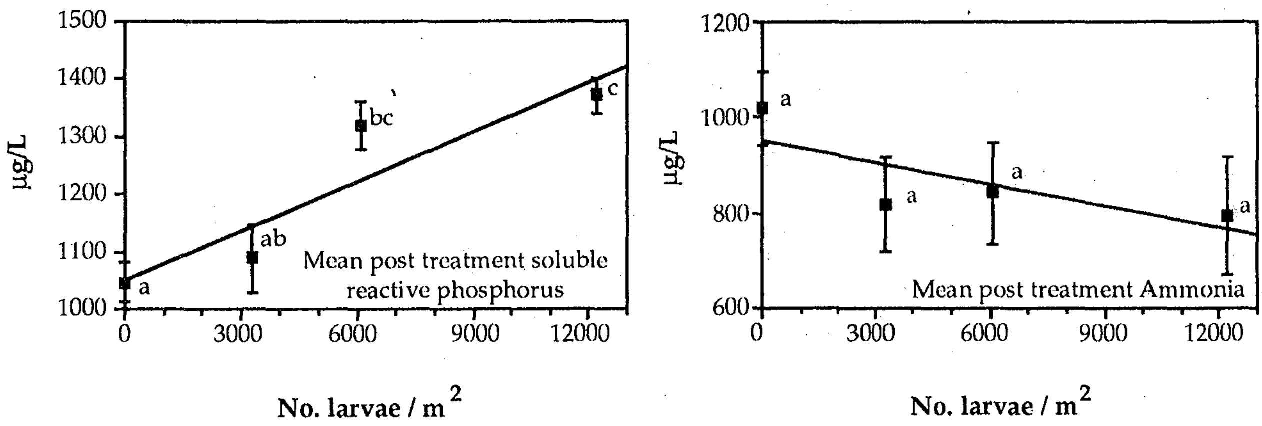

Figure 4.2. Density of larvae in columns versus mean post-treatment concentrations of soluble reactive phosphorus [SRP] and ammonia. Mean values followed by the same letter are not significantly different.

were between the control and both 6100 and 12200/m^2 and between 3300/m^2 and 12200/m^2.

Ammonium concentration (Fig. 4.2) remained fairly stable for the first ten days in both treated and control columns before dropping to very low levels on day 19 (31 - 34 µg/L) in the columns with larvae. A lesser decline in ammonium concentration (to 527 ± 304 µg/L) was recorded for the control columns. A significant treatment by time effect ($P<0.05$) was found for ammonium concentration, indicating that treatment level had some effect on the changes in ammonium concentration. The direction of the effect is not clear, and the differences between treatments were not significantly larger after addition of larvae than before ($P=0.176$). No significant differences were found between any of the treatments when the post treatment data were subjected to univariate ANOVA.

DISCUSSION

The results indicate that increasing the density of larvae of *P. nubifer* in the sediment of the columns had a positive effect on phosphorus concentration in the water column. The exact relationship cannot be determined because of the low survival rate by the end of the experiment. However, if the mean density of larvae during the experiment is assumed to be the density mid-way between the original density and the final density, then some approximate calculations can be made. A correlation coefficient of 0.84 is then obtained between post-treatment SRP concentration and mean larval density. That is, 84% of variation in SRP concentration after treatment is explained by the density of larvae. Despite the problems with survival of the larvae, the results suggest that *P. nubifer* larvae can contribute to transfer of phosphorus from the sediment to the water column.

This finding supports that of Gallep (1978) and Fukuhara and Sakamoto (1987), both of these studies found that increasing the abundance of chironomid larvae caused an increase in sediment phosphorus release (most of which was in the form of SRP). Gallep (1978) showed that most of the SRP release is explained by excretion of waste products by the chironomid larvae and noted that the effect was ten times greater when temperature was increased from 10 to 20°C. This experiment was carried out at 16°C, which is approximately the median water temperature at the sediment water interface during winter at North Lake (see Chapter Two). Summer median temperatures at North Lake have been between 25 and 30°C and larval densities have reached between 27000 and 30000/m^2, thus the results of this experiment may represent an underestimation of the effect of chironomids on phosphorus release at times when larvae are most abundant. Low dissolved oxygen concentration at the sediment water interface generally encourage phosphorus release, however, Tatrai (1988) found that oxygenated sediment can also be an important nutrient source in lakes when chironomids occur at densities above 1000 larvae/m^2. Bioturbation in four Wisconsin Lakes was found to have a greater effect on phosphorus release than any other physical and chemical factor tested (Holdren and Armstrong 1980).

Phosphorus exchange between sediment and water is also dependant upon such factors as sediment metal content (particularly iron, calcium and aluminium), pH, dissolved oxygen concentration and microbiological activity (Forsburg 1985a), all of which may modify the contribution of chironomids to phosphorus release.

Nitrogen dynamics in wetlands is somewhat more complex than for phosphorus. Statistical analysis showed that while density of larvae did affect the pattern of change in ammonia concentration over time, there was no difference in post-treatment ammonia concentrations between treatments. Denitrification (the conversion of ammonia or nitrates to elemental nitrogen, N_2), and nitrification (the conversion of ammonia to nitrates) are important in nitrogen cycling and may confuse studies of nitrogen release where only ammonia is analysed. Henrickson *et al.* (1983) demonstrated that the rate of both of these processes can be enhanced by burrowing invertebrates. More meaningful results may have been obtained if the concentration of nitrates had also been monitored.

The transfer of the results of this single experiment to the field situation is limited because the many physical, biological and chemical processes influencing the cycling of nutrients in wetlands could not be incorporated into the experiment. However, the results of this and other studies suggest that chironomid larvae may contribute substantially to nutrient exchange between the sediment and water in the wetlands of Perth.

Chapter Five

Studies on the Emergence and Movement of Adult Chironomidae

A) DIEL EMERGENCE OF CHIRONOMIDS

INTRODUCTION

The time of day or night that chironomids emerge is highly species specific, ranging from continuous emergence to one or more peak periods. The latter is more common and the peak periods can be either at dusk, midnight, dawn or mid-day depending upon the species and time of year (Morgan and Waddell 1961; Iwakuma and Yasuno 1983 and Jackson 1988). Patterns such as these that repeat regularly over a 24 hour period are known as diel rhythms. The time of emergence is often determined by the response of the pupae to one or more environmental cues such as water temperature or light intensity, but may also be partly controlled by internal biological rhythms.

The biological advantages of diel patterns of emergence to the chironomid are severalfold. Firstly it enables large numbers of males and females to congregate together at the same time and place. The males generally form large highly visible mating swarms which the females are able to see from a distance. Females then fly into the swarm to mate and having done so leave to lay eggs in the lake. Synchronous emergence allows all individuals a much better chance of finding a mate and possibly gives a broader choice of mate leading to increased genetic exchange.

Another advantage of emerging at a particular time of day would be to avoid unfavourable environmental conditions. Whereas warm climate species tend to emerge at dusk or dawn when the threat of dehydration is low, arctic and subarctic species generally emerge during the middle of the day, when the temperature is warm enough to allow maximum activity (Oliver 1971).

Finally, mass emergence at dusk when visibility is low may reduce predation by both aquatic and terrestrial predators (Morgan and Waddell 1961), although predation may still occur. At North Lake aquatic insects (corixids) and birds (waterfowl and swallows) have been observed to prey on emerging and emerged chironomids at dusk.

The temporal pattern of emergence of adult midges has several implications for their control. Fogging of adults with synthetic pyrethroids has occasionally been used in Perth as a control measure and the timing of such operations may be crucial to their effectiveness. In addition, the accuracy of sampling chironomids by emergence traps and light traps is dependant upon setting and collecting the traps at the correct times, to catch the majority of emerging adults of all species during a 24 hour period.

In order to investigate the diel patterns of emergence of chironomid species inhabiting Perth wetlands, emergence was monitored over 24 hour periods on several occasions. Ambient physico-chemical conditions were also monitored to determine whether the chironomids were responding to any environmental cues.

METHODS

Eight emergence traps were set on the northern shore of North Lake on six occasions between October 1990 and February 1991. The traps were the same type as were used for the Sumilarv field experiment and the field monitoring (see Chapter Seven for a description of the traps). Eight traps were set at 1200hrs (midday) and the trapped insects were removed every two hours for 24 hours. On three occasions the traps were collected at half hourly intervals in the period around sunset.

Whenever the traps were emptied environmental variables were also measured. These variables included the amount of incidental solar radiation above water, the maximum and minimum water temperature, and the concentration of dissolved oxygen in the water.

RESULTS AND DISCUSSION

Only two species (*Tanytarsus bispinosus* and *Polypedilum nubifer*) occurred in large enough numbers for their patterns of emergence to be determined. A typical pattern of emergence for these species over a 24 hour period is illustrated in Figure 5.1. Emergence patterns during the period around dusk are illustrated in Figure 5.2.

The emergence of both species followed a unimodal periodic pattern of emergence, that is, one well defined peak of emergence occurred during a 24 hour period (Fig. 5.1). This peak generally started 30 to 60 minutes after sunset (as defined by the Perth Observatory) (Fig. 5.2). However, light was still detectable up to half an hour after the official sunset, and so solar radiation is preferable as a meaningful and quantifiable measurement. Maximum emergence of *T. bispinosus* occurred in the first half hour period after solar radiation was no longer detectable (zero $\mu E/m^2/s$) (Fig. 5.2). For example if solar radiation was no longer detectable at 2000 hours then the main peak in emergence was between 2000 and 2030 hours. Very few adults emerged prior to or following this period. The maximum period of emergence was the same for both males and females.

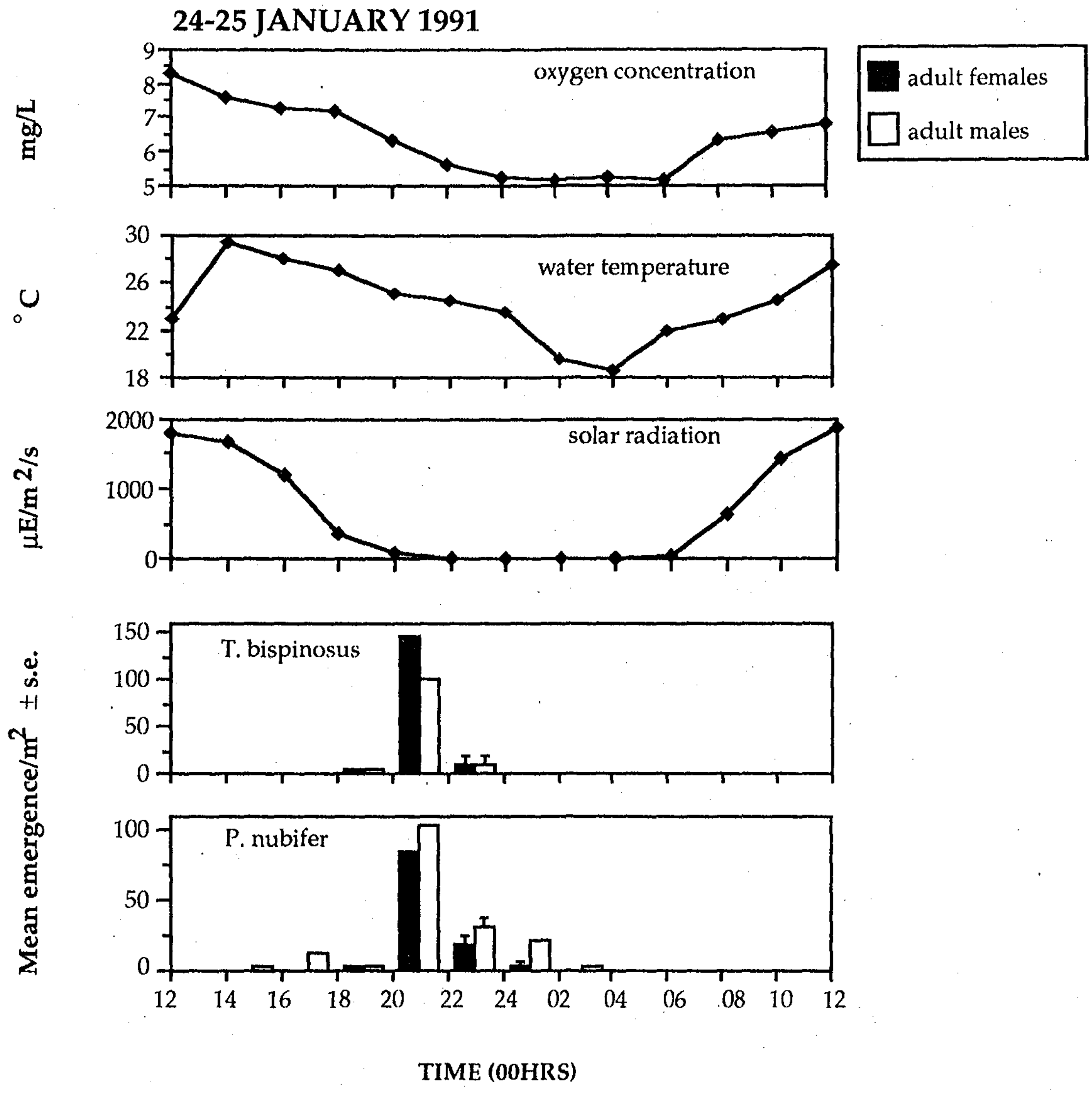

Figure 5.1. Changes in the rate of adult emergence of two species of chironomidae from the littoral region of North Lake measured for two hourly intervals over over a 24 hour period.

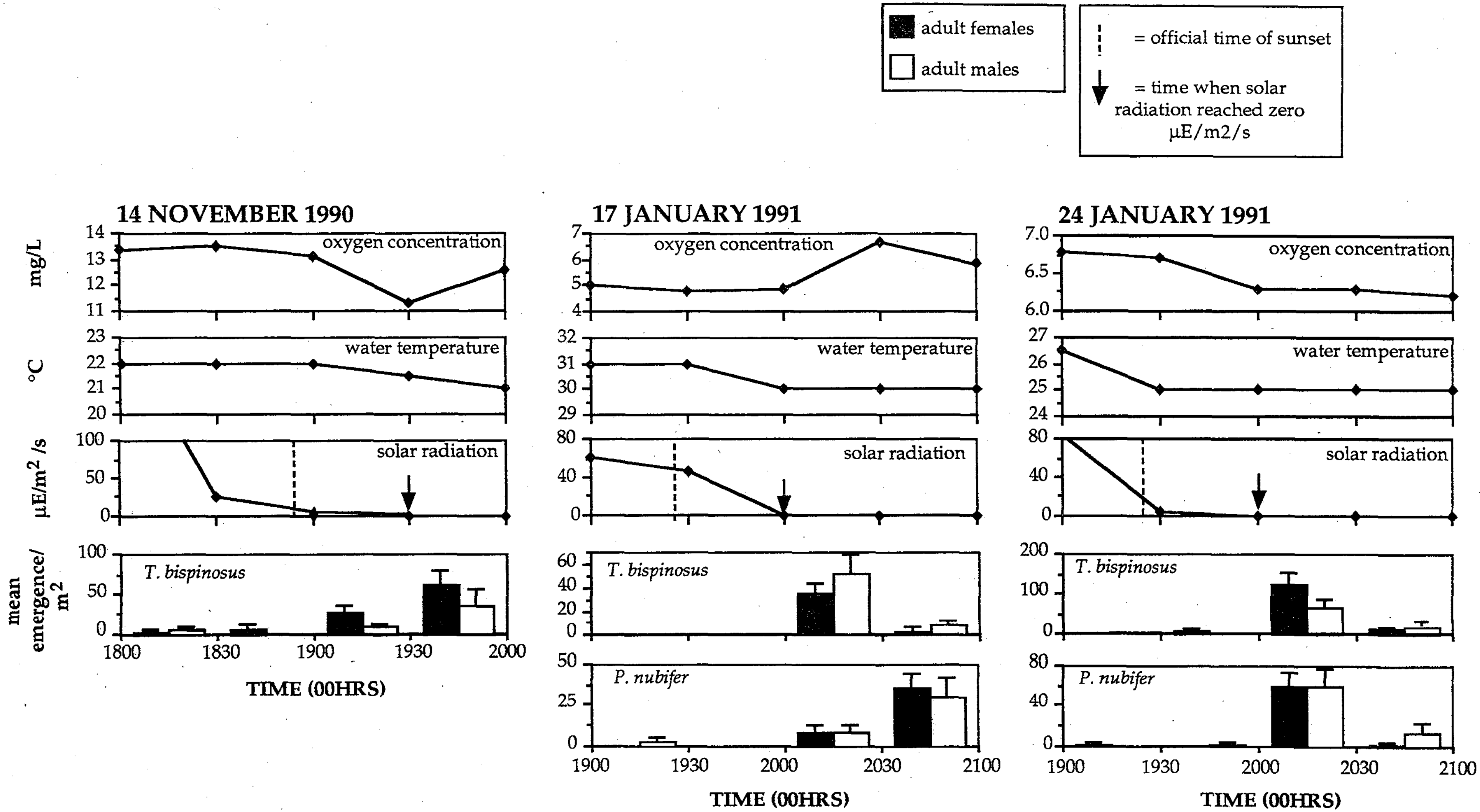

Figure 5.2. Changes in mean chironomid emergence (± standard error), dissolved oxygen concentration, water temperature and solar radiation, at dusk, on three occasions at North Lake.

Solar radiation appears to be the main factor influencing the time of emergence of*T. bispinosus* (Fig. 5.2). The time that solar radiation reached zero $\mu E/m^2/s$ was later during January than during November (reflecting seasonal differences in the time of sunset) and the time of maximum emergence of *T. bispinosus* was correspondingly later. Although other factors such as water temperature and dissolved oxygen concentration also varied during a 24 hour period (Fig. 5.1 and 5.2), the time of emergence of *T. bispinosus* was not consistently related to these other parameters. These variables may influence the response of chironomids to the main environmental cue but this is difficult to discern from field monitoring (Wrubleski and Ross 1989). Laboratory experiments (Kureck 1979 and Danks and Oliver 1972) have revealed that factors such as temperature and the animal's own endogenous rhythms can either trigger emergence or modify a chironomid's response to the main external factor.

Polypedilum nubifer only emerged in substantial numbers on two of the occasions in which traps were sampled intensively at dusk. On one of these occasions (24 January) the maximum period of *P. nubifer* emergence occurred in the first half hour period (2000 to 2030 hours) after solar radiation reached zero $\mu E/m^2/s$ at 2000 hours (Fig. 5.2). On 17 January *P. nubifer* reached peak emergence in the second half hour period (2030 to 2100) after solar radiation reached zero $\mu E/m^2/s$ (2000 hours) (Fig. 5.2). On this latter occasion the water temperature was 5°C higher than on the first occasion (Fig. 5.2). It is possible that *P. nubifer* is responding to light, but this response is being modified by temperature. This has been observed for other species, for example Kureck (1979) found that the diel emergence of *Chironomus thummii* is synchronised to light intensity but is modified by water temperature. The maximum period of emergence was the same for both males and females.

Other species of chironomid can cause nuisance problems in Perth (such as *Chironomus occidentalis* at Lake Monger and *Tanytarsus barbitarsus* at Rottnest Island). These may have different diel patterns of emergence to *P. nubifer* and *T. bispinosus.*

Monitoring the effectiveness of Sumilarv treatments requires an assessment of changes in the level of adult chironomid emergence. Thus, the use of emergence traps by local government authorities may become more widespread if the insect growth regulator Sumilarv is used as a replacement to Abate. These results indicate that for routine measurement of emergence of the main problem species, *P. nubifer,* the traps only need to be set from late afternoon to early morning to catch the majority of adults. The results also suggest that fogging operations against *P. nubifer* adults (using synthetic pyrethroids) may be more effective after sunset than before. However, such operations only control the adults and not the density of larvae as the females have usually already completed egg laying.

B) EFFECT OF WIND ON THE MOVEMENT OF ADULTS

INTRODUCTION

In 1989/90 light trap data from North Lake was compared to wind speed and direction data for Perth collected by the Bureau of Meteorology. This analysis revealed no apparent correlation between wind data and the relative catches of traps placed on the western and northern sides of North Lake. However, it was hypothesised that Perth wind data may not accurately reflect local variations in wind conditions at North Lake and that local wind data may give better insights into the movement of adult chironomids.

The 1990/91 light trap data were compared to wind data collected at the Murdoch University meteorological station which is located a few hundred metres north of North Lake.

METHODS

Each light trap consisted of a mercury vapor lamp, powered by 240 volt mains electricity, suspended above a funnel which channelled dead or stunned adults into a collecting bottle. These were illustrated in the 1989 report (Davis *et al.* 1989). The traps were set on 53 nights between November 1990 and March 1991. Two traps were set on the northern side of the lake and two traps on the western side.

The wind data were recorded by a continuous roll data recorder at the Murdoch University meteorological station for 49 of the nights that light traps were set. From these data the wind speed and direction was calculated for the period between 1900 and 0100 hours.

RESULTS AND DISCUSSION

The average catches of the North and West traps (expressed as percentage of the total catch) are given in Figure 5.3. Each column presents the mean percentage catch for the days when wind blew from a particular wind direction. Relative trap catch is used rather than abundance because the latter may have been affected by the application of Sumilarv to North Lake on 16 November 1990.

Wind direction was from either the S or SSW on more than half of the dates on which light traps were set, whereas wind was from the WSW, W, WNW, NNW and N on only one night each. This created a bias in the results and the following discussion should be

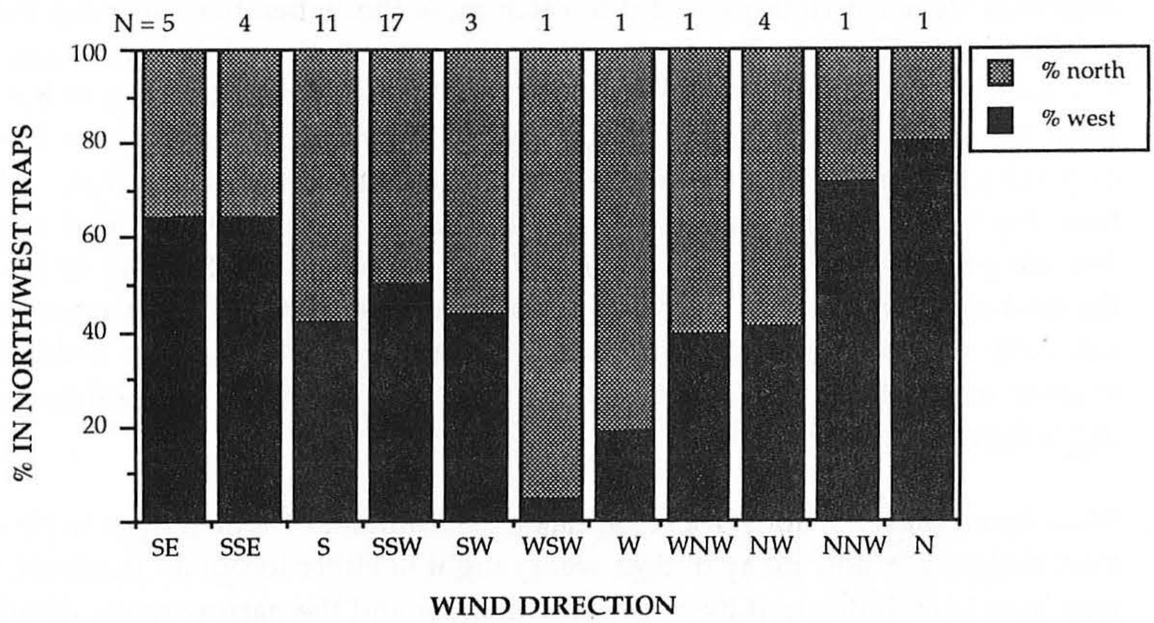

Figure 5.3. The mean percentage of the total light trap catch caught by the traps located to the north and west of North Lake, for each wind direction. The numbers above each column are the number of dates on which each wind direction was recorded.

viewed with caution. In addition light traps were only set on the northern and western sides of the lake where houses were present. This was because nuisance problems were only experienced on these sides of the lake and the initial objective of using the traps was to gauge the extent of the nuisance within urban areas. A more comprehensive study of the effects of wind speed and direction would require traps to be located on all four sides of the lake.

In general the northern traps tended to catch more flies when the wind was from a WSW or W direction (Fig. 5.3). This is most likely due to inhibition of the movement of swarms in a westerly direction. The western traps caught the larger proportion of the total catch when the wind was from the SE, SSE, NNW or N directions. Winds from the SE or SSE may have favoured the movement of swarms towards the western traps, while winds from the NNW and N may have inhibited the northerly movement of swarms, thus favouring the western traps by default. When the winds were S to SW or WNW to NW the total catch was more evenly divided between the northern and western traps. The southerly winds may have blown midge swarms from Bibra Lake towards both the western and northern traps. Wind did not blow from the NE or E direction on any of the nights that the traps were set.

Wind speed does not appear to be a major determinant of which light traps caught the most midges nor how many midges were caught in either location. However, this result may have been influenced by a restricted data set and the narrow range of wind speeds recorded (two to four metres per second on most dates).

In summary, wind appears to have some influence on the movement of adult chironomids, perhaps by inhibiting upwind movement and encouraging movement with the wind. However, even under moderate wind speed conditions the attractiveness of lights to midges may still be an overriding factor and movement across-wind appears possible. The effect of barriers of vegetation still requires investigation, and a study of the attraction of *P. nubifer* to various light sources and wavelengths (such as that undertaken by Kokkin and Williams (1989) on *Tanytarsus barbitarsus* in South Australia) would also be beneficial.

Chapter Six

Laboratory Determination of the Toxicity of Sumilarv® (pyriproxyfen) to Selected Aquatic Fauna

INTRODUCTION

Single species bioassays are recognised for their important role in the determination of relative toxicity and in their estimation of safe levels of toxicants. In the United States, guidelines for deriving numerical water quality criteria (Stephan *et al.* 1985) require bioassays on at least eight different families of freshwater animals before a safety criterion can be set for a chemical.

Sumilarv® has been tested on numerous aquatic organisms under laboratory conditions and the results of these studies are summarized in Table 6.1. Clearly, different organisms display varied sensitivity to this product. In addition, as with many chemicals, the age of the organism will affect its sensitivity.

Laboratory and field tests indicate that the rate at which Sumilarv® should be applied in Perth wetlands to control the nuisance chironomid, *Polypedilum nubifer* is 10kg product/ha (Davis *et al.* 1990). This is equivalent to 0.01ppm of the active ingredient, pyriproxyfen (assuming 50cm depth). Schaefer *et al.* (1988) investigated the effects of 0.01ppm pyriproxyfen on cladoceran and copepod populations in laboratory aquaria and did not detect any significant effects on the populations over a two week period. In addition, data provided by the Sumitomo corporation suggests that an application rate equivalent to 0.01ppm pyriproxyfen would be unlikely to harm carp, killifish, hornshell molluscs, freshwater shrimps, copepods, brine shrimps, dragonflies or daphnid cladocerans.

In order to determine the relative toxicity of pyriproxyfen (Sumilarv®) to species which comprise the local aquatic fauna, a series of laboratory bioassays were performed on indigenous fauna which coexist in chironomid habitats. Obviously it was impossible to test all wetland fauna and thus the organisms selected for these bioassays were generally the most abundant species representing a range of taxonomic groups. The only vertebrate tested was the goby, *Pseudogobius olorum*, an indigenous fish species which is predatory upon invertebrate fauna. All of the invertebrates tested were detritivores, a group which dominate the faunal composition in many of Perth's urban wetlands (Davis and Rolls 1987). Two insect types which undergo different patterns of development were tested; the

ORGANISM	CONCENTRATION (ppm)	EFFECTS	REFERENCE
RAINBOW TROUT - JUVENILE	0.85	LC50 96hr	
CARP - JUVENILE	0.45	LC50 96hr	
ALGAE	>1.00	EC50 96hr	
KILLIFISH - ADULT	2.7	LC50 96hr	
- EMBRYO	>10	EFFECTIVE LEVEL GIVEN	
- YOLK SAC FRY	~ 0.3	EFFECTIVE LEVEL GIVEN	SUMITOMO
- POST LARVAE	1	EFFECTIVE LEVEL GIVEN	UNPUBLISHED
- JUVENILE	3	EFFECTIVE LEVEL GIVEN	DATA
MOLLUSC (HORNSHELL)	1	EFFECTIVE LEVEL GIVEN	
FRESHWATER SHRIMP	~ 0.2	EFFECTIVE LEVEL GIVEN	
COPEPODA	0.1 - 1.0	EFFECTIVE LEVEL GIVEN	
BRINE SHRIMP	>10	EFFECTIVE LEVEL GIVEN	
DRAGONFLY	>0.1	EFFECTIVE LEVEL GIVEN	
CARP	4.16	LC50 96hr	SUMILARV BROCHURE
DAPHNIA	10	LC50 3hr	
CLADOCERA	0.01	NO SIGNIFICANT EFFECTS AFTER 14 DAYS	SCHAEFER et al. 1988
COPEPODA	0.01	NO SIGNIFICANT EFFECTS AFTER 14 DAYS	

Table 6.1 Summary of laboratory tests to assess the toxicity of Sumilarv® to nontarget fauna

corixid, *Micronecta robusta*, (Fig. 3.1 E) which is monomorphic and the mayfly, *Cloeon fluviatile* which is polymorphic. Both the toxic effects of pyriproxyfen and its influence on the emergence of *C.fluviatile* were investigated. Two crustaceans were also tested; the ostracod, *Candonocypris novaezelandiae*, and the amphipod, *Austrochiltonia subtenuis* (Fig. 3.1 F).

METHODS

Dilution water

Water was collected from Horsepaddock Swamp, a small wetland situated approximately 3km south of the Murdoch University campus and 100m south of North Lake (Fig 2.1). This was used as the dilution water for all laboratory tests involving fish and amphipods. When that source became unavailable, the water used in all other tests was obtained from Southern Swamp, a wetland located 100m east of North Lake. Both wetlands receive groundwater from the same flowline (Bayley *et al.* 1989). The pH of water collected from Horsepaddock Swamp and Southern Swamp was 8.29 and 8.33; conductivity was 1040 and 1320μS, respectively. All water was filtered through a sieve (53μm mesh) and autoclaved at 109°C for 45 minutes. The water was stored at room temperature in 20L black plastic drums until required.

Test material and preparation

The granular formulation of Sumilarv® was not used in these toxicity assays because accurate dilution of the product to very low concentrations was not possible. Instead, technical grade (96.6% pure) pyriproxyfen, the active ingredient of Sumilarv® 0.5G, was used. The pesticide was dissolved in methanol and required test concentrations were made up in dilution water. Background levels of methanol in all test solutions was less than 0.4% v/v.

Animal Collection and Acclimation

Invertebrates were collected by sweep sampling from various habitats which had not been previously exposed to Sumilarv®. In the laboratory, they were sorted according to size and placed into aquaria inside a growth cabinet where they were acclimated to the test conditions (20°C; 12hr day) and the dilution water over a 48hr period. The invertebrates were fed detrital material during the acclimation period.

The gobies, *Pseudogobius olorum*, were obtained from North Lake. The young age class of fish selected (< 8months old) for testing would have ensured that the not have been exposed to Sumilarv® previously. The fish were collected using a seine net (3mm mesh size), transferred to aerated aquaria and acclimated to laboratory conditions (18 ± 5°C; 12hr day) and dilution water for 48hr. The fish were fed live brine shrimp during this period.

Test procedure

Exploratory tests were performed in order to ascertain the approximate range of pyriproxyfen concentrations that affected each type of organism. Briefly, six different concentrations ranging from 0.0001ppm to 100ppm were tested. Equivalent amounts of methanol were also diluted and tested (solvent-added controls) and one control containing dilution water only was also included. There were no replicates. These tests took place over a 48hr period and the results were used to determine the range of concentrations to be tested more thoroughly. The range selected for precision testing included the lowest concentration that affected all or most of the test organisms and the highest concentration that affected none or only a few.

The procedure for the more complex precision testing varied for the different types of organisms. In general, each assay involved testing at least five different pyriproxyfen concentrations. Equivalent amounts of methanol, diluted to make up at least five solvent-added controls, were tested and a control consisting of the dilution water only was also included. All test solutions were replicated three times. Tests took place over a 48hr period and were repeated three times for each animal type. No attempt was made to sex the animals but a consistent size range was used for each test. The animals were not fed during the assays. The proportion of animals affected by the pesticide solution were corrected for the toxic effects of the solvent solution and the natural response in the controls using Abbott's formula (Abbott 1925). Death of the invertebrates was often difficult to determine and was undesirable for the fish, in terms of animal ethics. Thus, behavioural change rather than death was used as the toxicity criterion in many of the bioassays. Toxicity values for each organism were calculated from dosage toxicity curves using Probit analyses (SPSS 1988) and are expressed as either lethal concentration (LC) or effective concentration (EC) as determined by the toxicity criterion used for each animal type.

Amphipods: Ten individuals of *A. subtenuis* were added to 250 mL glass beakers containing 200mL of the test solutions. Pyriproxyfen concentrations tested were 0.001, 0.005, 0.01, 0.05, 0.1, 0.5 and 1ppm. Immobilization was used as the toxicity criterion. This was defined as the lack of rapid swimming motion in response to being gently prodded with a glass rod. In total, 1350 amphipods were tested.

Fish: Three *P. olorum* were added to aerated aquaria (20x15x15cm) containing 2L of the test solutions. Pyriproxyfen concentrations tested were 1,2,4,8 and 10ppm. Behavioural change was used as the toxicity criterion and this was defined as the loss of buoyancy control and righting reflex. The experiment was monitored every six hours and any fish showing signs of behavioural changes were removed from the aquaria, euthanased by cervical cord separation and scored as affected. At the conclusion of the 48hr test period, all fish were euthanased, weighed and measured. Fish used in the three replicate trials were of length (20.3±0.4mm, 20.7±0.3mm, 21.3±0.2mm) and weight (81.9±4.3g, 66.8±2.8g, 68.7±0.1g) respectively. In total, 315 fish (includes the number required for the range finding test) were tested and all experiments were performed under the guidelines of the Prevention of Cruelty to Animals Act, 1920.

Ostracods: A total of 990 individuals of *C. novaezelandiae* were tested. Ten ostracods were added to plastic petri dishes (5cm diameter) containing 10mL of the test solution. Pyriproxyfen concentrations tested were 1, 2.5, 5, 7.5 and 10ppm. The criterion used to assess toxicity to these animals was the inability to move rapidly in a circular motion.

Corixids: Four juvenile *M. robusta* were added to 250mL glass beakers containg 200ml of the test solutions. To account for the positive bouyancy exhibited by these animals, the beakers were lined with wire mesh onto which the corixids could attach in order to maintain their position in the solution. Pyriproxyfen concentrations tested were 0.1, 2.5, 5, 7.5 and 10ppm. The total number of corixids tested was 396. The toxicity criterion for these animals was death.

Mayflies: Only two assays were carried out using the mayfly, *C. fluviatile* The number of late instar mayfly nymphs that were used in the tests depended upon their availabiliy. Three mayflies per beaker were used in the first assay, four were used in the second. Nymphs were placed into 250mL glass beakers containing 200mL of the test solution. Pieces of flywire mesh (8 x 5cm) were placed in the beakers to provide perches from which the mayfly could emerge. Curtain netting was placed over the top of each beaker and held in place by an elastic band. This prevented emergent mayflies from escaping. Because exploratory tests had indicated that these animals were affected by a wide range of pyriproxyfen levels, the following concentrations were tested: 0.0001, 0.0005, 0.001, 0.005, 0.01, 0.05, 0.1, 0.5 and 1ppm. The number of animals available was limited and therefore only one solvent-added control was included and this was at the rate equivalent to the concentration of solvent in the 1ppm pyriproxyfen solution. Death was used as the toxicity criterion. Whilst all adult mayflies which emerged successfully were counted as alive, any partially emerged mayflies which had failed detach from the skin of the subimago stage were counted as dead.

Emergence study

Three replicates of the 0.01ppm pyriproxyfen solution, the solvent-added controls and the true controls were made up according to the method used in the bioassays. Late instar mayfly nymphs (*C. fluviatile*) were added to each of the beakers. The beakers contained a

ORGANISM	EC50 (ppm)	LC50 (ppm)	95% C.L.	SLOPE
Fish- Pseudogobius olorum	3.92	-	3.18 - 4.78	3.06
Amphipod- Austrochiltonia subtenuis	0.12	-	0.046 - 0.24	2.26
Corixid- Micronecta robusta	-	1.25	0.33 - 2.54	1.42
Ostracods- Candonocypris novaezelandiae	6.21	-	4.29 - 10.54	2.43
Mayfly- Cloeon fluviatile	-	0.17	0.094 - 3.6	1.44

Table 6.2 Median effective and lethal concentrations (48hr) of pyriproxyfen to selected fauna.

small piece of flywire mesh and were covered with nylon netting to prevent the emergent mayflies from escaping. Emergence was monitored every twenty four hours over a forty eight hour period and all emergent mayflies were removed. Any dead nymphs or adults were scored and removed. The test was repeated three times and, whilst ten mayflies were added to each beaker in the first two trials, only six were added in the third trial. The results from each trial were combined and differences between the proportion of mayflies emerging in the three treatment types were analysed by oneway analysis of variance (ANOVA).

RESULTS

The median effective and lethal concentrations of pyriproxyfen for the organisms tested ranged from 0.12 to 6.21ppm (Table 6.2). The concentration of the pesticide required to detrimentally affect to 50% of the gobies and the ostracods or kill 50% of the corixids were 392, 621 and 125 times higher, respectively, than the proposed field rate of 0.01ppm. The mayfly *C. fluviatile* and the amphipod *A. subtenuis* were more sensitive. A concentration of pyriproxyfen at ten times the proposed field rate, would kill 50% of the mayflies and harm 50% of the amphipods, respectively. The probit results indicated that a concentration of 0.01ppm pyriproxyfen would harm 1% of the amphipod population and kill 4% of mayflies.

Pyriproxyfen (0.01ppm) did not appear to inhibit the emergence of the mayflies. The rate of emergence from the pesticide solution was not significantly different from either the solvent-added control or the dilution water-only control ($P>0.38$).

DISCUSSION

Overall the bioassay results suggest that a concentration of 0.01ppm pyriproxyfen is not acutely toxic to any of the organisms tested. The organisms exhibited either median effective or median lethal responses that were at least ten times higher than this rate. Although the mayfly, *C. fluviatile* appeared to be the most sensitive of the organisms tested, pyriproxyfen (0.01ppm) did not inhibit the emergence of this species.

The age or life stage of an organism can have a considerable influence on its sensitivity to a toxicant and generally, juveniles are thought to be more sensitive to toxicant stress than adult organisms (Chapman 1983). Since these bioassays have only investigated the toxic effects of pyriproxifen on a restricted size/age class for each organism, it is possible that the toxicity values generated may be either over or underestimations of the true toxic response for these organisms. Long-term assays would provide more accurate estimations of the toxic response. These tests expose organisms to a series of concentrations of a toxicant for a portion of, or the entire life cycle and survival time, the number of young produced and population growth rate are measured. Long-term tests are often used to determine the maximum allowable concentration of a toxicant in an aquatic system. This is the highest exposure concentration of a substance that will not cause harm the test organisms in terms of survival, growth or reproduction (Buikema *et al.* 1982).

The bioassay results provide a measure of the relative toxicity of pyriproxyfen to a range of organism types and an estimate of the relative 'safety' of a particular rate of application (0.01ppm). However, the tests were performed under relatively narrow test conditions and do not incorporate the biotic and abiotic interactions which occur in the natural environment. Therefore, it would not be appropriate to extrapolate these results to the lake environment. Validation of these results, under more complex environmental conditions, would considerably increase the confidence that can be placed on predictions made from these laboratory results. Primarily, validation would determine whether the organisms tested in the laboratory, would respond in the same way to pyriproxyfen in the environment. Secondly, validation would determine whether organisms, other than those tested here, are affected by pyriproxyfen. Finally, it would determine whether the pesticide effects higher levels of biological organization such as a communities or ecosystems.

Validation studies are usually performed in *in situ* field enclosures or in artificial ponds and test a range of toxicant concentrations (Pontasch and Cairns 1991). The field experiment performed at North Lake (see Chapter Seven) examined the effects of only a single application rate of Sumilarv® (10kg product/ha; equivalent to 0.01ppm pyriproxyfen) and is therefore limited in its capacity to validate these bioassay results. Nevertheless, the field study does support the results of these laboratory tests.

Chapter Seven

Effects of Sumilarv® 0.5G on Chironomids and Non-target Organisms in a Lake Environment

INTRODUCTION

The decline in the effectiveness of the organophosphate pesticide, Abate®, against chironomids in some Perth wetlands led to the investigation of the insect growth regulator Sumilarv® 0.5G (pyriproxifen; JHA - S31183) as an alternative control agent. Following laboratory trials to determine a suitable application rate, Sumilarv® was tested against chironomids under field conditions on two occasions during the 1989/90 research programme (Davis *et al.* 1990). The results of that study suggested that Sumilarv® applied at a rate of 10kg/ha (0.05 kg AI/ha) effectively inhibited the emergence of all chironomid species present by more than 80% for at least a three week period. Sumilarv® is considered to be an effective alternative to Abate® for the short-term control of chironomid nuisance problems (Davis *et al.* 1990). In view of this, it is essential that some attention be focussed on the environmental safety of Sumilarv®.

In the aquatic environment the potential hazard of a substance is largely determined by its environmental fate (Stern and Walker 1978). Sumilarv® has not been shown to persist in the soil for long periods (>14days), nor is it rapidly displaced from the upper soil strata by leaching and therefore the pesticide is unlikely to penetrate into groundwater (Schaefer *et al.* 1991). The persistence of Sumilarv® in the water column, following application at rates ranging from 0.005 to 0.11kg AI/ha, has been examined in a number of studies (Schaefer *et al.* 1988; Schaefer and Muira 1990; Mulligan and Schaefer 1990; Schaefer *et al.* 1991). None of these studies were able to detect Sumilarv® in the water column after 48hrs. However, despite the undetectable levels of the pesticide after such a short period, its biological activity has been found to persist for up to two months (Schaefer *et al.* 1988). Mulligan and Schaefer (1990) found that the period of residual activity of the pesticide was related to water quality. The active ingredient of Sumilarv® readily adsorbs onto organic debris and the adsorbed material is then available for ingestion by the biota. After binding with organic matter Sumilarv® decays at an exponential rate (Schaefer *et al.* 1991). In 'clean' water with no organic content, the persistence of Sumilarv® is indirectly proportional to temperature and the amount of sunlight it is exposed to (Schaefer *et al.* 1988).

Sumilarv® is thought to have a limited bioaccumulative ability. This has been demonstrated in both the laboratory (Schaefer et al. 1988) and the field (Schaefer and Muira 1990). Three days after field exposure to the pesticide at the rate of 0.05kg AI/ha, the Sumilarv® concentration in fish tissues had fallen to undetectable levels.

Sumilarv® is being tested overseas for its potential as a mosquito control agent and is considered to be highly effective at rates between 0.005 and 0.01 kg AI/ha (Mulla *et al.* 1986; Schaefer *et al.* 1988). Studies that have investigated the effects of the pesticide on non-target organisms have concluded that, at the rates required to control mosquitoes, Sumilarv® does not adversely affect other organisms (Schaefer *et al.* 1988; Schaefer and Muira 1990). Because the rate of Sumilarv® required to control chironomids is up to ten times higher than that required to control mosquitoes (Davis *et al.* 1990), it was considered essential that the effects of Sumilarv® on organisms which coexist with chironomids in Perth wetland ecosystems be determined.

After consultation with the EPA and the Health Department, it was agreed that up to three whole lake field trials of Sumilarv® would take place at either Lake Monger or North Lake, during the 1990/91 research program. In addition to monitoring the effect of Sumilarv® on chironomid populations, extensive monitoring would be undertaken to determine any effect that the pesticide might have on the non-target plankton and invertebrate populations.

Lake Monger was chosen as the preferred site for these trials because the composition of the chironomid community and the physico-chemical parameters of the lake were different from North Lake where limited field trials had already taken place. However, only one site was found to be suitable for the location of enclosures at Lake Monger. In the event that chironomid larval densities did not increase sufficiently in that area for experimental purposes, the field trials would be relocated at North Lake.

Throughout the summer of 1990/91 chironomid larval densities at Lake Monger remained low (< 500/m^2 - see Chapter One) and substantial nuisance problems were not experienced in the surrounding residential area (A. Van Leeuwen pers. comm.). Hence, no pesticide applications were required at this lake during the summer months.

At North Lake, chironomid larval abundance had exceeded the threshold level of 5000 larvae/m^2 by mid spring (third and fourth instar larvae only). Larval density had increased from less than 3000 larvae/m^2 recorded in late September to over 10000 larvae/m^2 recorded two weeks later in early October. The nuisance species, *Polypedilum nubifer*, accounted for 85% to the total density at that time. Based on these figures it was agreed that Sumilarv® would be applied to North Lake in conjunction with an enclosure experiment. The pesticide application took place on the 16th November 1990 and although a whole lake treatment had been intended, only the littoral region could be sprayed because sufficient pesticide had not arrived at that time.

METHODS

The enclosure experiment was conducted at North Lake between November 8th and December 6th 1990. Six $25m^2$ enclosures were constructed in the littoral region on the eastern side of the lake. These were identical to the enclosures used in a similar experiment with Sumilarv® at North Lake during the 1989/90 research (Davis *et al.* 1990). The enclosures were erected in water up to 40cm deep, one week prior to the application of Sumilarv® 0.5G.

Seven days after the construction of the enclosures, Sumilarv® was added by hand to three of the enclosures at a rate of 10kg product/ha (0.05kg AI/ha). The remaining three enclosures were to act as controls. Following this, all of the enclosures were covered with black plastic to avoid any further addition of pesticide when the lake was sprayed.

Sumilarv® was applied by helicopter at the rate of 10kg product/ha to the littoral region of the lake. In order to assess the evenness of the pesticide distribution, plastic buckets were used to collect the pesticide as it was applied. Two buckets were suspended above the water surface by a one metre length of PVC pipe (20mm in diameter) which was bolted to a metal star picket. This apparatus was considered preferable to the aluminium trays, which had been used previously to collect the pesticide, as the trays tended to be sunk by wind and wave action generated by the helicopter (Davis *et al.* 1990). Prior to the pesticide application the collecting apparatus were placed at twelve sites located at regular intervals, in the littoral region, around the lake. Immediately after the application, the buckets were taken back to the laboratory where the Sumilarv® granules collected in each pair of buckets were pooled and weighed so that the average application rate to the littoral region of the lake could be calculated.

The effects of Sumilarv® on both chironomids and non-target fauna were evaluated by monitoring emergence and faunal densities in both the treated enclosures and the control enclosures. Chironomid emergence and the emergence of other non-target fauna were monitored using emergence traps which were more robust, than those used previously (Davis *et al.* 1989). The traps were constructed from white plastic funnels (20cm high and 24cm internal diameter), with a jar lid fitted to the outside of the apex end so that a specimen bottle (600mL) could be attached. The wide end of the funnel was placed on the lake bed and held in place by metal pegs which were pushed through perimeter holes of the funnel and into the substrate. When the trap was submerged the upper half of the specimen jar contained air in which the emerging fauna were trapped.

Emergence was monitored on four occasions prior to the pesticide application, and then on seven occasions afterwards, over a two week period. On each occasion, three traps were set in each enclosure overnight and collected the following morning. The emergent chironomids and non-target fauna were preserved in ethanol (100%) and quantified at a later date. The traps were only set in one half of the enclosures and they were positioned randomly within that half. Traps were never set in the same position on consecutive nights.

The abundance of chironomid larvae and other non-target fauna in the enclosures and in the area immediately surrounding the enclosures (external) were monitored on two occasions prior to the application of the pesticide. After the application, faunal densities in the enclosures were monitored on four further occasions over a two week period. No further monitoring occurred in the area external to the enclosures because the actual rate of application to the littoral region (6.9 ± 1.5 kg/ha) was not comparable that of either the treated or control enclosures. Faunal abundance was monitored using a column sampler which was designed to sample biota in both the water column and in the sediments. The column sampler consisted of a PVC pipe (10cm in diameter; 1m in length) with a serrated metal edge attached. This was hammered into the substrate to a depth of approximately 10cm and a rubber bung was placed on top of the pipe. The pipe was then tilted to enable a bung to be placed at the submerged end, thus trapping the water and the sediments within. The contents of the pipe were then emptied into a 250µm mesh net to remove the water and the net contents were shaken into plastic bags and preserved in ethanol (100%). Column samples were taken from the opposite half of the enclosures from where the emergence traps were set. This area was subdivided into three regions of equal size, with increasing depth. Thus, the region closest the shoreline was less than 10cm deep, the adjacent region was between 10 and 25cm deep, and the region bounded by the back wall of the enclosure was between 25 and 40cm deep. On each sampling occasion, one sample was taken randomly from each region.

In the laboratory, the column samples were washed through a set of three sieves (2mm, 500µm and 250µm). The material retained on the 2mm sieve was placed into a white sorting tray and all invertebrates detected by eye were removed, and identified to family level and preserved in vials in 70% ethanol for further identification. Material retained in the 500 and 250µm sieve was subsampled (by volume) and examined under a binocular microscope where they were identified to family level and counted. A portion of each familial group was then removed and stored as described for identification. Identification was to species level, where possible, otherwise to the lowest taxonomic grouping achievable.

Environmental parameters such as temperature and dissolved oxygen were measured in the enclosed and external areas using an oxygen probe. A water sample was taken from each enclosure and at three locations in the external area on two occasions prior to the pesticide application and then on five occasions afterwards. The samples were analysed for pH and chlorophyll-*a* concentration was determined by M. Lund at Murdoch University.

Analyses

Differences in the emergence of both chironomids and non-target fauna between the treatments and over time were analysed by two-way nested analysis of variance (ANOVA). Where significant differences were detected, *a posteriori* Tukey's HSD tests were carried out to determine which means were significantly different at the 0.05 level of probability.

The density of larval chironomids and non-target fauna in the three column samples were averaged for each enclosure and for the external area and the mean abundances were used in the analyses. Faunal densities, chlorophyll -*a* concentration and the environmental parameters (pH, temperature and dissolved oxygen) in the different treatment types and external area were compared over time using repeated measures ANOVA.

All dependent variables were tested for heteroscedasticity by Cochran's C test and those variables showing significant ($P<0.05$) heterogeneity were log transformed [$\log_{10}(n+1)$]. All of the analyses were run using the SPSS-X statistical package (SPSS 1988) and differences were accepted as significant at $P<0.05$ except where Cochran's C test showed that the variance was still heterogeneous after transformation. In this event, results of the ANOVA were only considered to be significant at the 0.01 level of probability (Longeragan *et al.* 1989).

The non-target invertebrate composition of the treated and control enclosures were compared by calculating the Bray-Curtis index according to the following formula:

$$D = \mathrm{SUM}\ |x_{1j} - x_{2j}|\ /\ \mathrm{SUM}\ |x_{1j} + x_{2j}|$$

where: x_{1j} = abundance of species j in the control enclosures;
x_{2j} = abundance of species j in the treated enclosures.

RESULTS

Environmental Parameters

Water temperature in the enclosures fluctuated between 19 and 26°C during the course of the experiment with few observed differences between temperature measured at the water surface and substrate level (Fig. 7.1 a,b). There were no significant differences in temperature between the two enclosure types or the external area ($P>0.6$) during the entire experiment.

Although dissolved oxygen concentration was measured at approximately the same time of day in all three areas, the specific time at which the measurements were taken was not the same on each sampling occasion. Consequently, large fluctuations in dissolved oxygen concentration were recorded at both the water surface and substrate level in all three areas (Fig. 7.2 a,b). However, dissolved oxygen concentration at both the water surface and substrate level did not differ significantly ($P>0.3$) between the three areas during the experiment. There were no apparent differences in dissolved oxygen concentration between the water surface and the substrate level at any time (Fig. 7.2 a,b).

The pH values recorded during the experiment ranged from 6.7 to 8.5 with no significant differences in the pH recorded in the three areas prior to the application of the pesticide and up to one week afterwards. After two weeks the pH values in the enclosures began to deviate from the external area. The pH recorded in the external area twenty days after the application of the pesticide was significantly higher ($P<0.05$) than the pH of the water within the enclosures (Fig. 7.3).

Larval chironomid abundance

The population of larval chironomids in the enclosures was made up primarily of first and second instar larvae with the contribution of third and fourth instar larvae rarely exceeding 10% of the total. There were predominantly three species of chironomids present during the experiment, *Polypedilum nubifer*, *Tanytarsus bispinosus* and *Cryptochironomus griseidorsum* (Fig. 7.4). Densities of first and second instar *P. nubifer* in the external area were significantly ($P<0.05$) higher in than the treated enclosures prior to treatment. Pretreatment densities of third and fourth instar *P. nubifer* larvae were similar in the three areas. After treatment the densities of third and fourth instar larvae in the control enclosures tended to increase whilst an overall decline was observed in the treated enclosures (Fig. 7.5 a). However, no significant differences between the two treatment types were discernable when the data were analysed statistically. The density of first and second instar *P. nubifer* in treated and control enclosures were not significantly different ($P>0.95$). Both treatment types showed an overall decline during the experiment (Fig. 7.5 b) with densities prior to the pesticide application being significantly higher

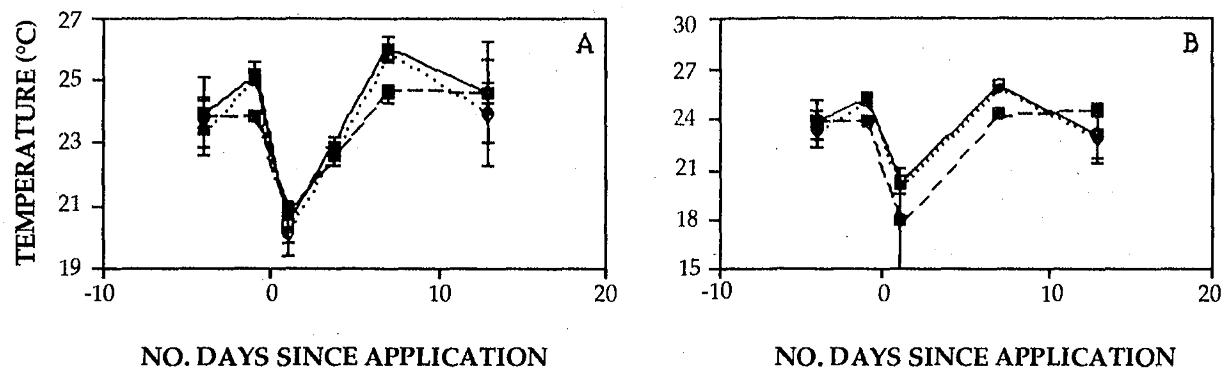

Figure 7.1 Changes in temperature of the water (mean ± s.e.) in the enclosures and external area at the water surface (A) and substrate level (B).

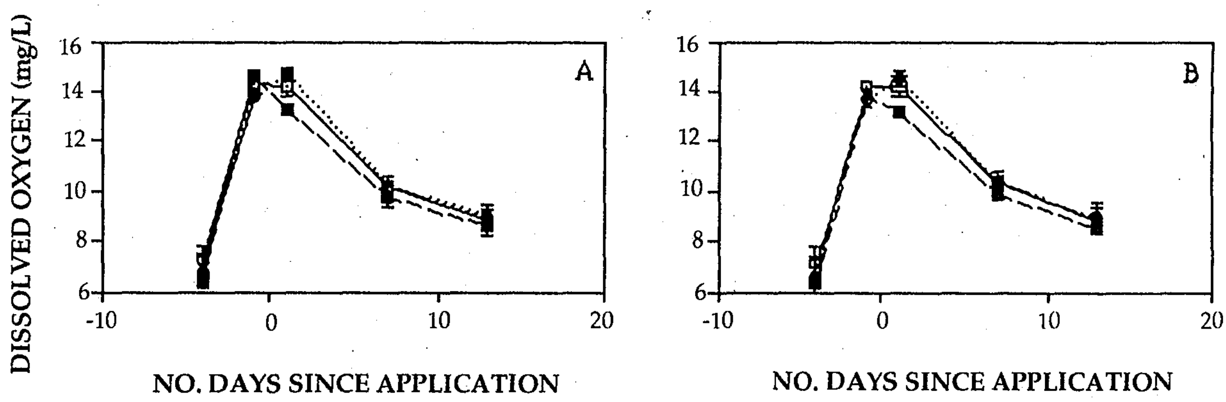

Figure 7.2 Changes in the concentration of dissolved oxygen (mean ± s.e.) of the water in the enclosures and external area at the surface (A) and substrate level (B).

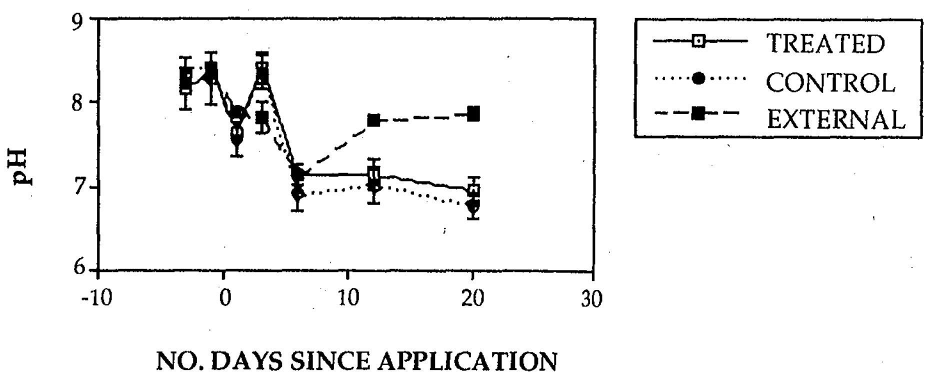

Figure 7.3 Changes in the pH of the water in the enclosures and the external area during the enclosure experiment.

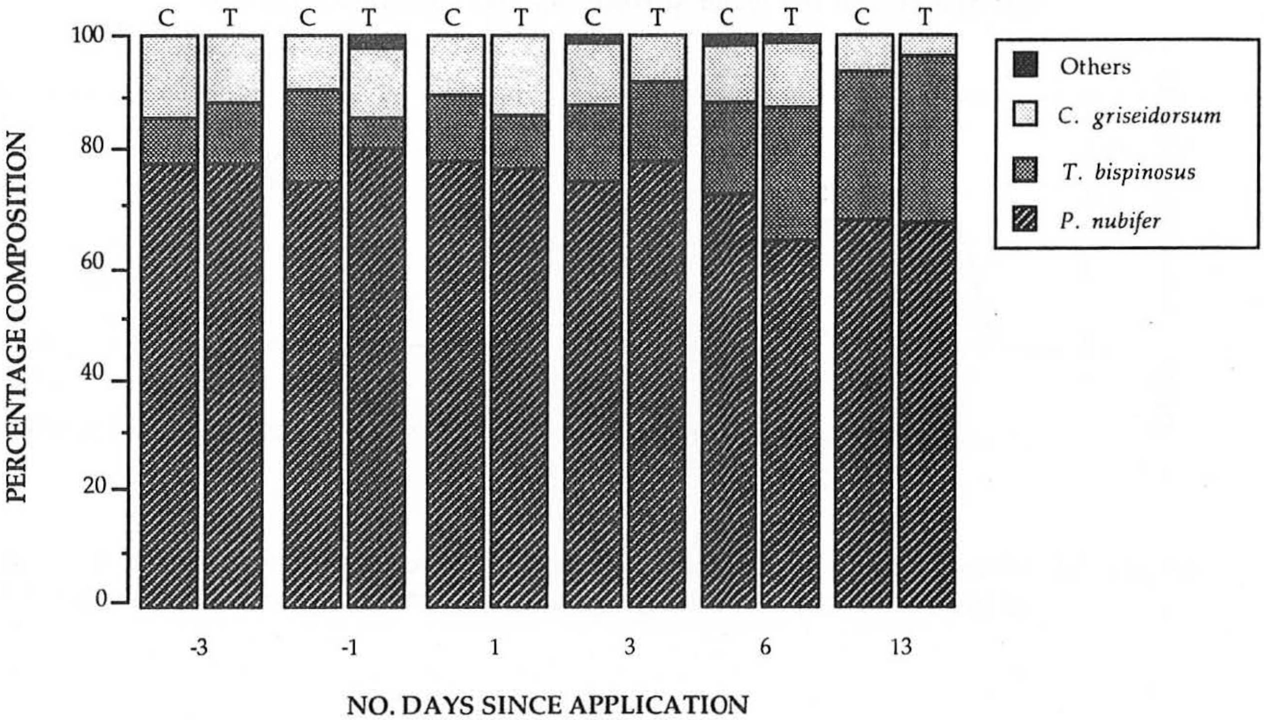

Figure 7.4 Changes in the chironomid species composition in the treated (T) and control (C) enclosures during the experiment.

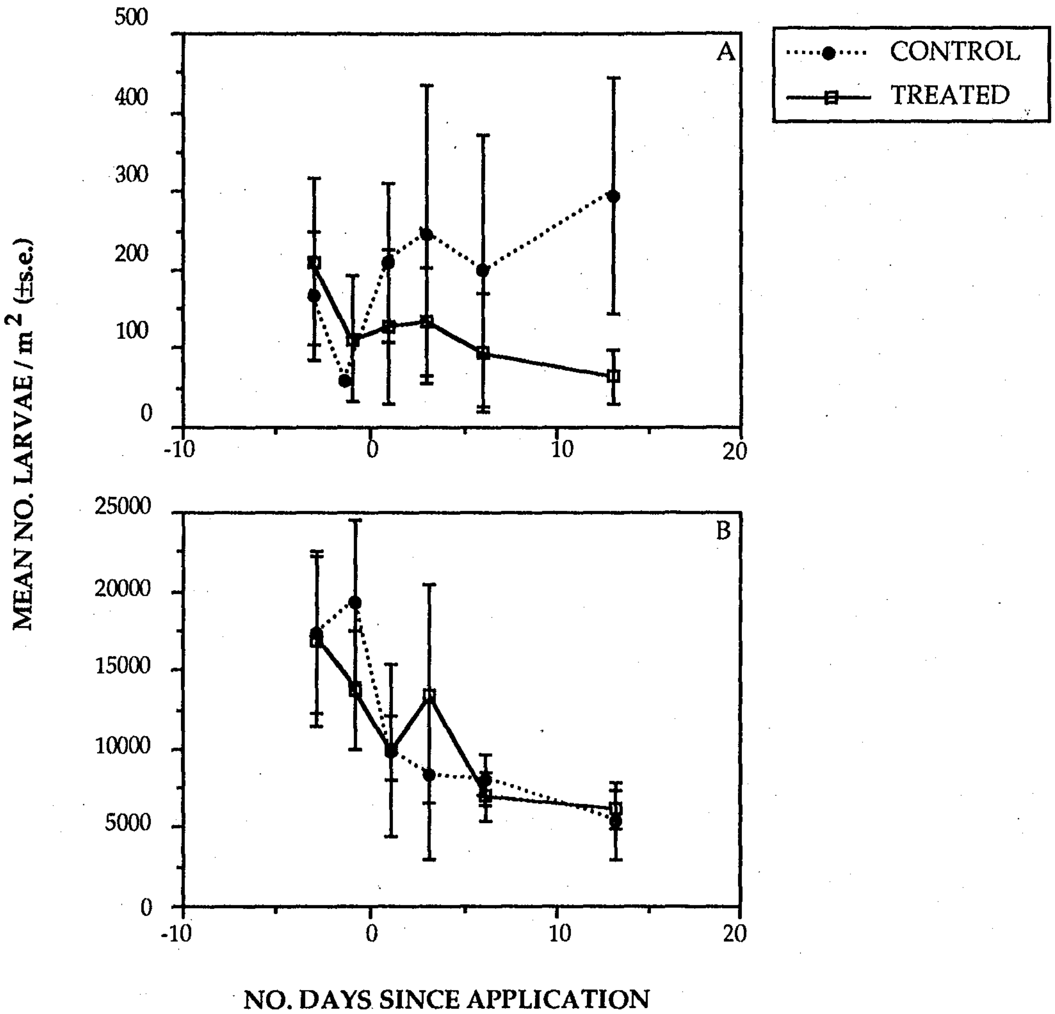

Figure 7.5 Densities of the chironomid *Polypedilum nubifer* in the treated and control enclosures: 3rd & 4th instar larvae (A); 1st & 2nd instar larvae (B).

($P<0.05$) than densities recorded afterwards.

Although the contribution of *T. bispinosus* to the total larval composition tended to increase with time in both the treated and control enclosures (Fig. 7.4), the actual densities of this species after pesticide application were not significantly different to those recorded prior to application ($P>0.3$) in either age group (Fig. 7.6 a,b). The density of *T. bispinosus* in the two treatment types did not differ ($P>0.62$).

The chironomid, *Cryptochironomus griseidorsum* did not contribute more than 15% to the total larval composition in either treatment type (Fig. 7.4). Two weeks after treatment, densities of the third and fourth instar larvae in the treated enclosures were lower than those recorded in the control enclosures (Fig. 7.7 a), but the difference was not significant. An overall decline in the density of first and second instar larvae in both enclosure types was observed (Fig. 7.7 b) with densities of larvae, prior to the pesticide application, being significantly higher than those recorded afterwards ($P<0.05$). No significant treatment effects were detected for either age group throughout the experiment ($P>0.59$).

Chironomid emergence

Despite *P. nubifer* being the dominant larval species, with larval densities as high as 25 000 larvae/m^2 being recorded, adults of this species did not emerge at rates exceeding 5 midges/m^2/night in either of the treatment types for the entire experiment. *Tanytarsus bispinosus* was the most abundant chironomid species collected in the emergence traps. Prior to the pesticide application the emergence of adult *T. bispinosus* was low (<100 adults/m^2/night) in both the treated and control enclosures. Following the pesticide application, emergence remained low in both enclosure types. After fourteen days the rate of emergence in the control enclosures began to increase whilst the rate of emergence in the treated enclosures remained low (Fig. 7.8). The pattern of change in the two enclosures might be considered to indicate that the emergence of *T. bispinosus* was suppressed in the treated enclosures by Sumilarv®. However, the results of the ANOVA suggests that there were no significant differences between the treatment types throughout the experiment (Table 7.1).

Non-target invertebrate abundance

Non-target invertebrate fauna from the following groups were collected from within the enclosures: nematodes, oligochaetes, amphipods, ostracods, copepods, ceratopogonids, corixids, cladocerans, hydracarinids, notonectids, hirudinids, stratiomyids, sciomyzids, tabanids, ephemeropterans and trichopterans. Only those that were consistently present in the enclosures were examined statistically. Densities of all abundant non-target fauna prior to the pesticide application were not significantly different between the two enclosures and the external area.

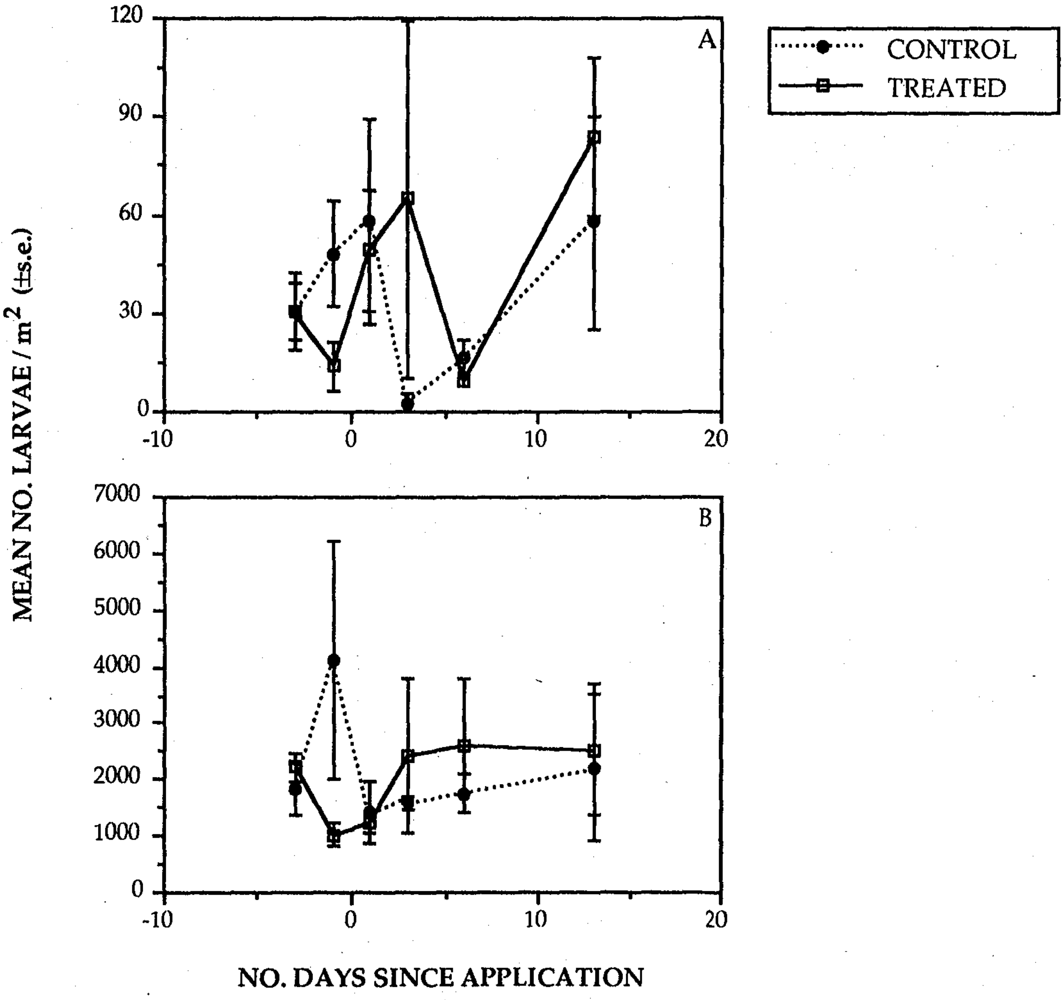

Figure 7.6 Densities of the chironomid *Tanytarsus bispinosus* in the treated and control enclosures: 3rd & 4th instar larvae (A); 1st & 2nd instar larvae (B).

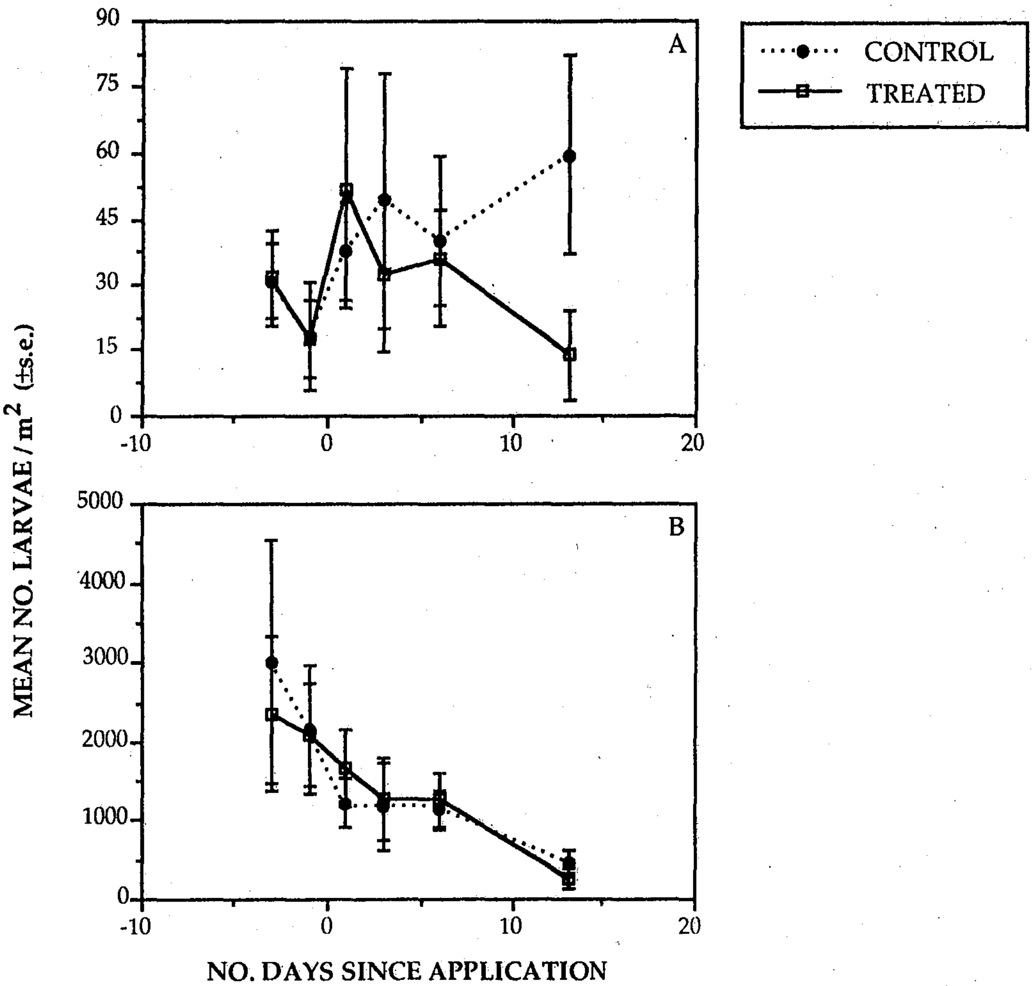

Figure 7.7 Densities of the chironomid *Cryptochironomus griseidorsum* in the treated and control enclosures: 3rd & 4th instar larvae (A); 1st & 2nd instar larvae (B).

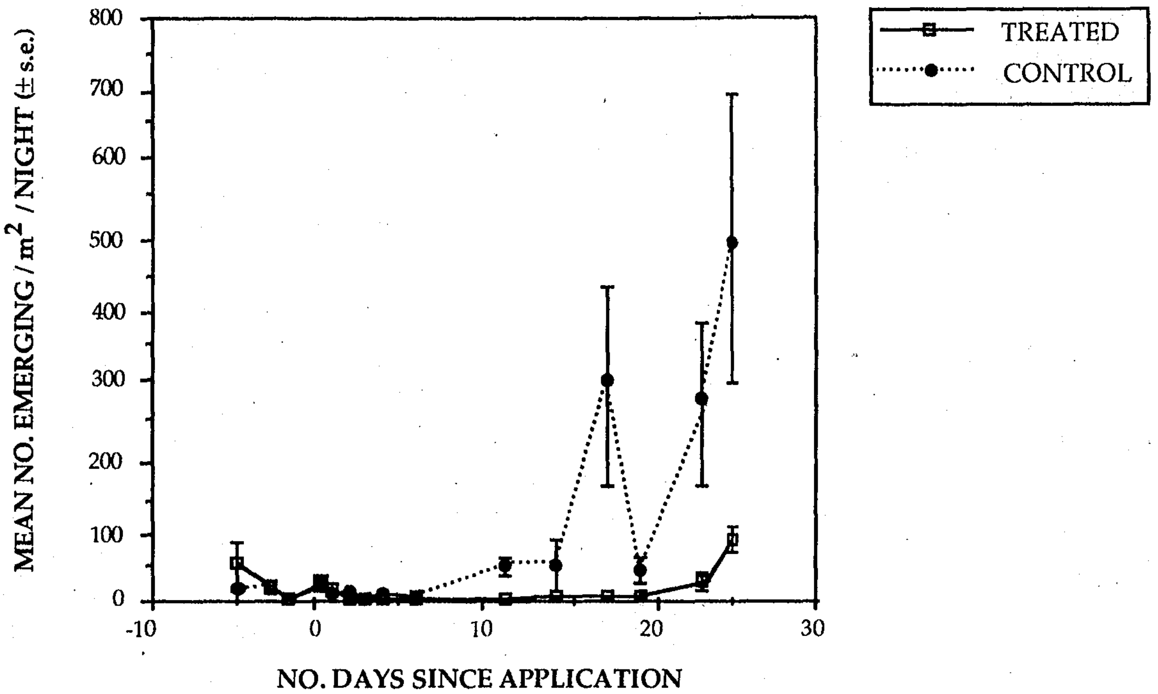

Figure 7.8 Changes in the rate of emergence of the chironomid *Tanytarsus bispinosus* emerging from the treated and control enclosures.

SOURCE OF VARIATION	d.f.	MS	F
TREATMENT (1)	1	0.002	0.46 n.s.
TIME (2)	14	1.800	0.63 n.s.
TREATMENT x. TIME (2)	14	0.002	0.09 n.s.
ERROR 1 ENCLOSURE within TREATMENT	4	0.004	0.20 n.s.
ERROR 2 TIME by ENCLOSURE within TREAT	52	0.020	1.00 n.s.
RESIDUAL	162	0.020	

Table 7.1 Results of nested analysis of variance on the rate of emergence of the chironomid *Tanytarsus bispinosus* during the enclosure experiment (d.f. = degrees of freedom; MS = mean squares; ns = not significant).

Six different types of zooplankton were collected in the column samples but only the calanoid copepod, *Calomecia tasmanica tasmanica,* and the cyclopoid copepod, *Mesocyclops sp.* were present throughout the experiment. The densities of both of these species appeared to decline sharply in the treated and the control enclosures after the treatment date (Fig. 7.9 a,b). However, pre and post-treatment densities were not significantly different ($P>0.01$). Densities recorded in the two treatment types did not differ during the experiment ($P>0.2$).

The composition of the non-target benthic fauna of the enclosures was dominated by nematodes, oligochaetes and the mayfly, *Tasmanocoenis tillyardi* (Fig. 7.10). Other invertebrates which featured consistently throughout the experiment included the caddisfly, *Ecnomus pansus,* the amphipod, *Austrochiltonia subtenuis,* the ostracod, *Cypericercus salinus* and the ceratopogonid, *Nilobezzia sp.*. There were very few differences in the composition of the invertebrate fauna between the treated and control enclosures throughout the experiment (Fig. 7.10).

Densities of nematodes recorded during the experiment were rarely lower than 15 000 animals/m^2 (Fig. 7.11 a). In the control enclosures the abundance of nematodes tended to fluctuate, with densities declining after the date of application. By contrast, densities recorded in the treated enclosures tended to increase after Sumilarv® was added. These opposing trends immediately after the Sumilarv® application, may account for the significant interactive effect ($P<0.05$) between treatment and time when pre and post–treatment data were compared. Overall, there were no significant differences in nematode densities between the treatments ($P>0.7$) and the densities within each treatment type did not change significantly over time ($P>0.6$).

Changes in the density of oligochaetes in both treatment types were not significantly different over time. Densities in both areas remained at approximately 5000 animals/m^2 until three days after the pesticide application when a significant ($P<0.01$) decline in oligochaete densities was recorded (Fig. 7.11 b). Following this, the abundance of these animals fluctuated until the final densities were marginally lower than pre–treatment levels.

Although densities of the ostracod C. *salinus* were an order of magnitude lower than the oligochaete densities, the pattern of change over time was remarkably similar (Fig. 7.11 c). No significant differences between treatment types were detected ($P=0.8$). In addition, densities recorded three days after treatment were significantly lower ($P<0.05$) than levels recorded previously. Final densities were not significantly different from pre–treatment levels.

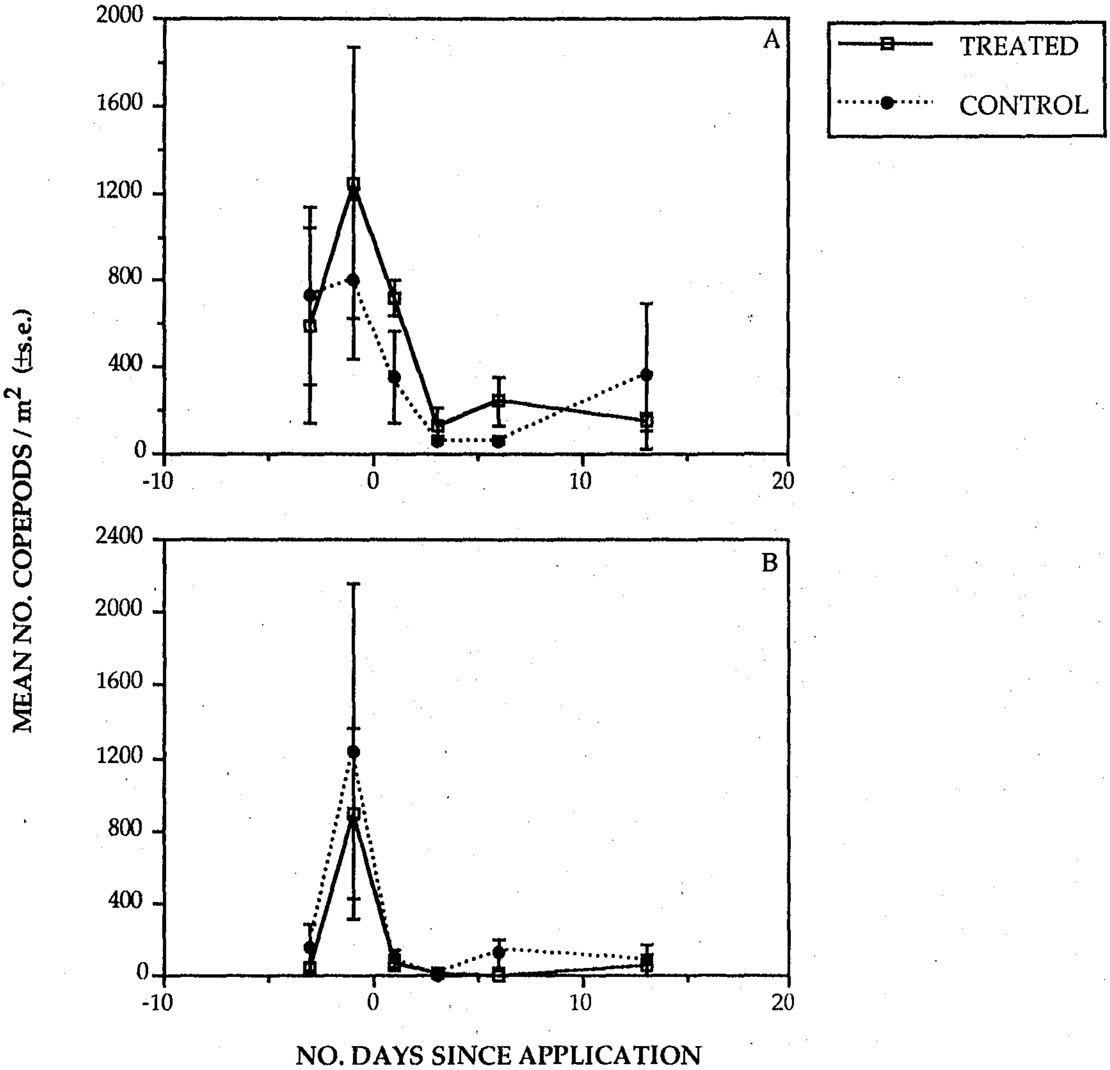

Figure 7.9 Densities of the copepods *Calomecia tasmanica tasmanica* (A) and *Mesocyclops* sp. (B) in the treated and control enclosures.

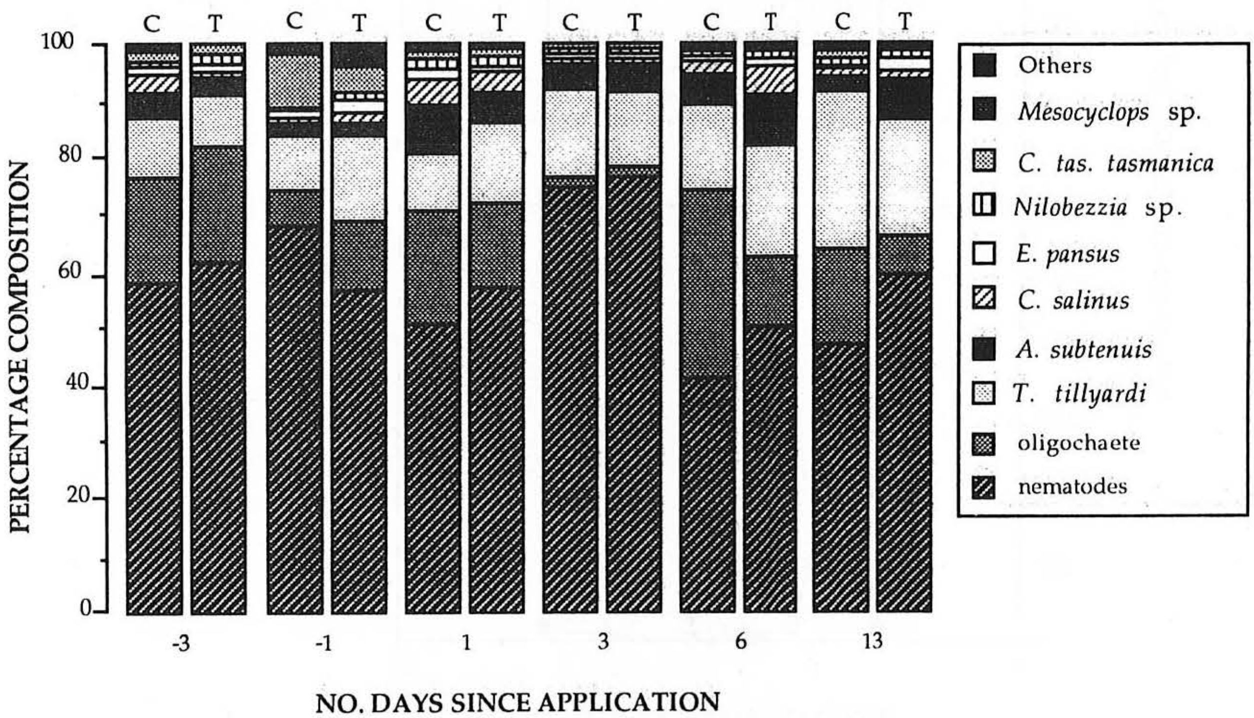

Figure 7.10 Change in nontarget species composition in the treated (T) and control (C) enclosures during the experiment.

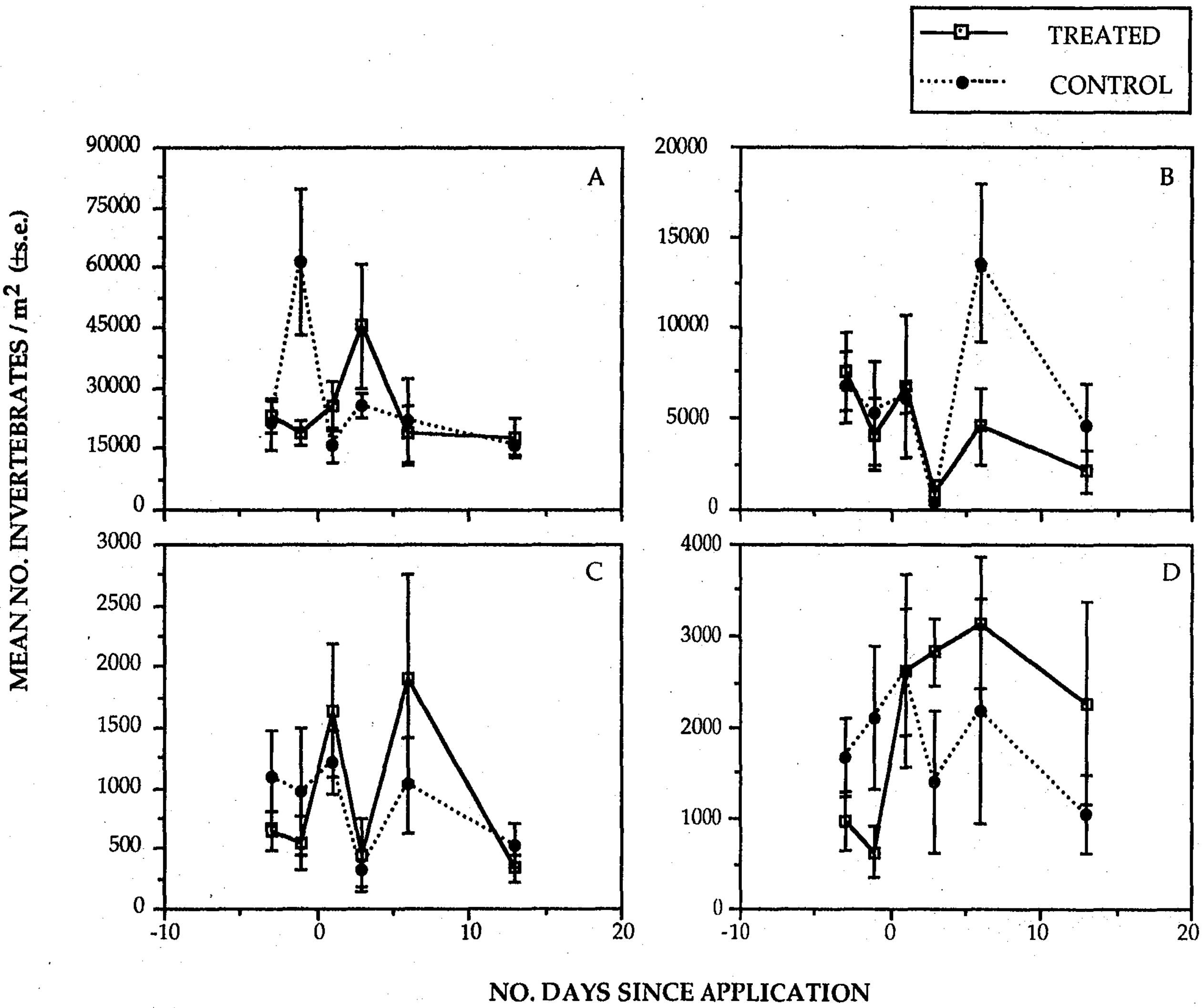

Figure 7.11 Densities of nontarget fauna in the treated and control enclosures: (A) nematodes; (B) oligochaetes; (C) the ostracod, *Cypericercus salinus*; (D) the amphipod, *Austrochiltonia subtenuis*.

No significant changes in density of the amphipod, *Austrochiltonia subtenuis* were recorded during the experiment (P=0.28). The density of this species in the two treatment types remained similar throughout (Fig. 7.11 d). Similarly, the abundance of the mayfly, *T. tillyardi*, the caddisfly *E. pansus* and the ceratopogonid *Nilobezzia sp.* (Fig. 7.12 a,b,c) did not differ significantly between the two treatment types (P>0.2) and densities did not change substantially throughout the experiment.

The Bray-Curtis index was used to compare the degree of similarity between the non-target invertebrate communities of the treated and control enclosures during the experiment (Fig. 7.13). In the strict sense, this index is actually an index of dissimilarity. The index ranges from 0 (identical samples) to 1.0 (no species in common). The degree of similarity between the control and treated enclosures was high throughout this experiment. The two areas were most similar on the day after application. Following this, the two areas diverged slightly but the index was never recorded over 0.55.

<u>Non-target emergence</u>

Numerous insect types emerged and were collected in the emergence traps, but only the caddisfly, *E. pansus* and the mayfly, *T. tillyardi*, emerged consistently enough to warrant further consideration, although the actual numbers emerging was low for both species (Fig. 7.14 a,b). The results of two-way nested ANOVA performed on data obtained for both species are summarized in Table 7.2. No significant difference between treatments were detected for either species. Temporal differences in the emergence of *T. tillyardi* in the control enclosures were detected and further investigation of these differences by *a prioriori* tests (Fig. 7.14 b) proved only that emergence from these enclosures fluctuated considerably over time.

<u>Phytoplankton abundance</u>

Chlorophyll-*a* concentration is a measure of the abundance of phytoplankton in the water column. The highest level recorded during the experiment was 95µg/L, but on average the levels recorded were between 40 and 60µg/L (Fig. 7.15). Throughout the experiment, no significant differences were recorded in chlorophyll-*a* concentration between the two enclosure types (P=0.39). Prior to the treatment there was no difference in phytoplankton abundance between the enclosures and the external area.

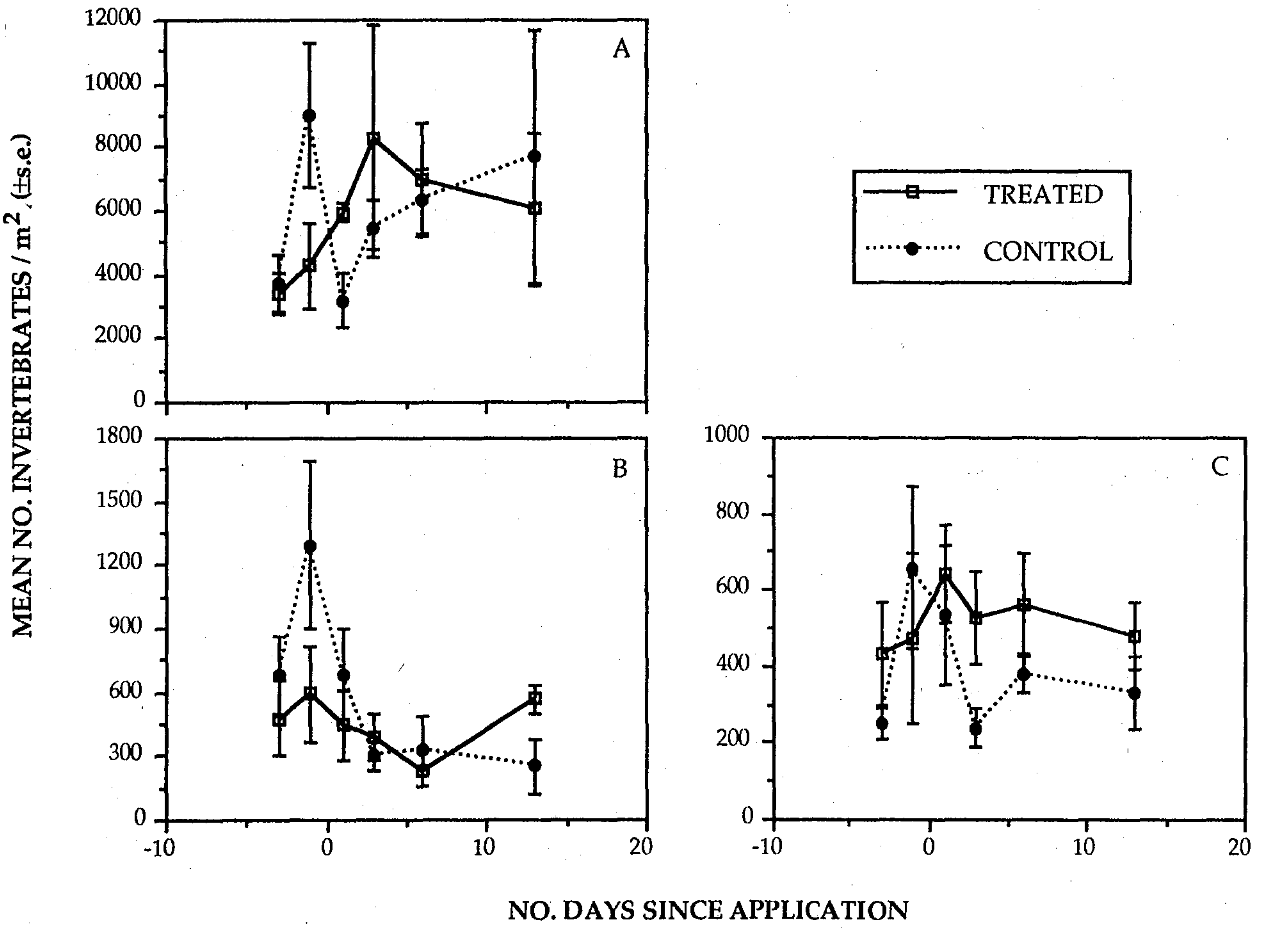

Figure 7.12 Densities of nontarget fauna in the treated and control enclosures: (A) the mayfly, *Tasmanocoenis tillyardi*; (B) the caddisfly, *Ecnomus pansus*; (C) the ceratopogonid, *Nilobezzia* sp..

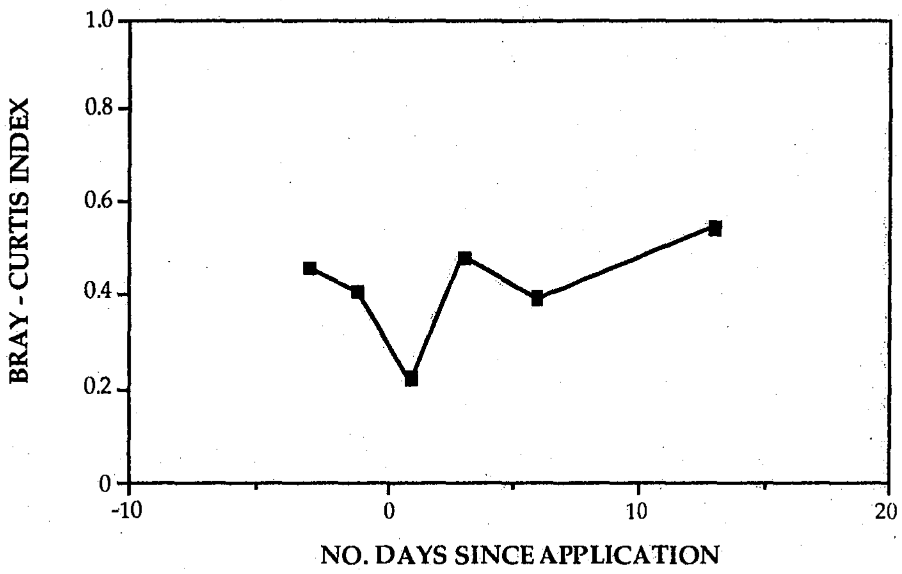

Figure 7.13 Changes in the Bray Curtis index during the enclosure experiment.

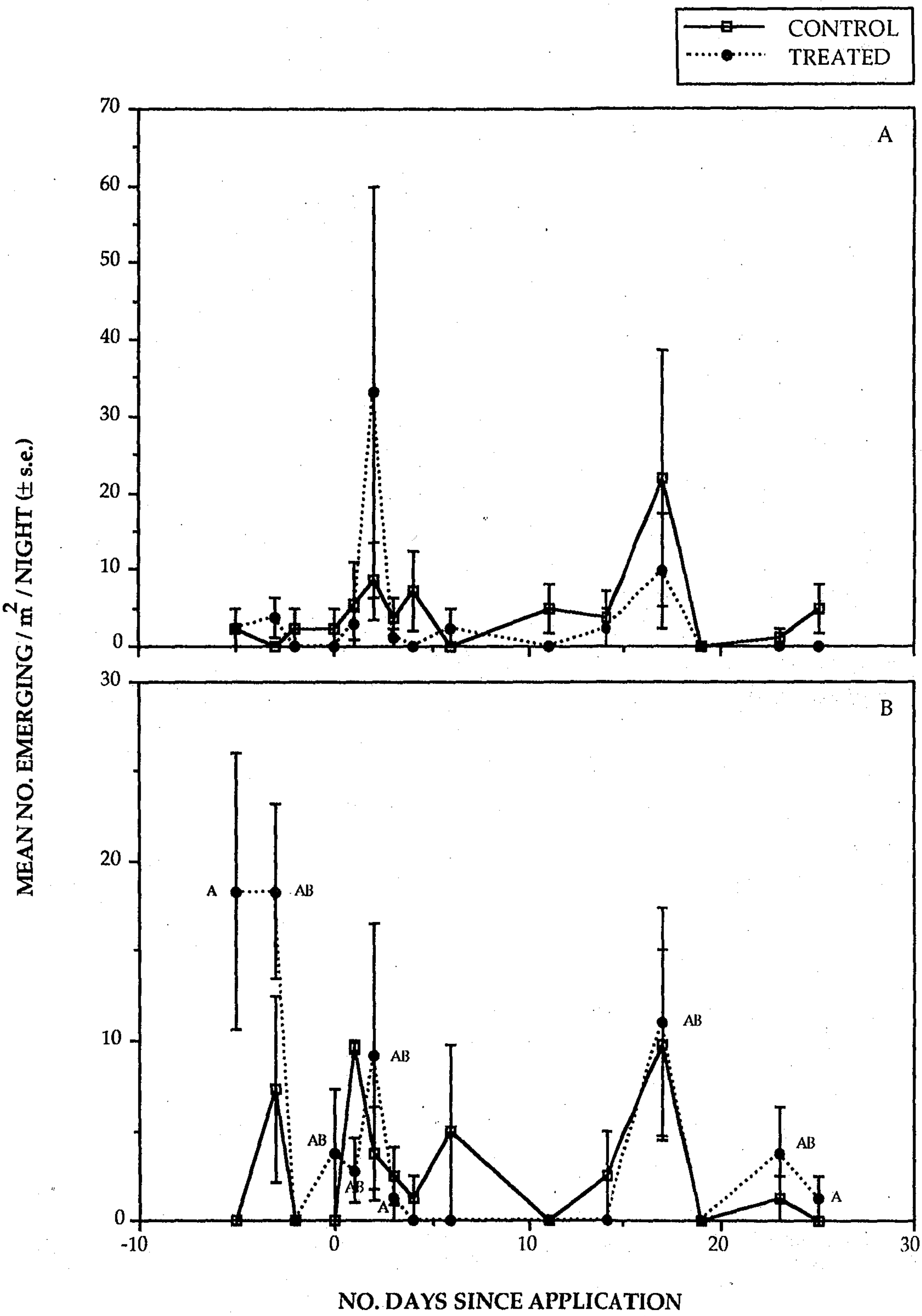

Figure 7.14 Changes in the rate of emergence of nontarget fauna from the treated and control enclosures: (A) the caddisfly, *Ecnomus pansus*; (B) the mayfly, *Tasmanocoenis tillyardi*. Means followed by the same letter are not significantly different (P>0.05).

SOURCE OF VARIATION	d. f.	M S	F
		Ecnomus pansus	
TREATMENT (1)	1	0.06	4.92 n.s.
TIME (2)	14	0.37	1.64 n.s.
TREATMENT x. TIME (2)	14	0.13	0.58 n.s.
ERROR 1 ENCLOSURE within TREATMENT	4	0.01	0.05 n.s.
ERROR 2 TIME by ENCLOSURE within TREAT	54	0.22	0.95 n.s.
RESIDUAL	174	0.23	
		Tasmanocoenis tillyardi	
TREATMENT (1)	1	0.43	1.35 n.s.
TIME (2)	14	0.67	2.36 *
TREATMENT x. TIME (2)	14	0.36	1.27 n.s.
ERROR 1 ENCLOSURE within TREATMENT	4	0.32	1.69 n.s.
ERROR 2 TIME by ENCLOSURE within TREAT	54	0.28	1.49 *
RESIDUAL	174	0.19	

Table 7.2 Results of nested analysis of variance on the rate of emergence of the mayfly, *Tasmanocoenis tillyardi* and the caddisfly, *Ecnomus pansus* during the enclosure experiment (d.f. = degrees of freedom; MS = mean squares; ns = not significant; * = significant at 0.05 level).

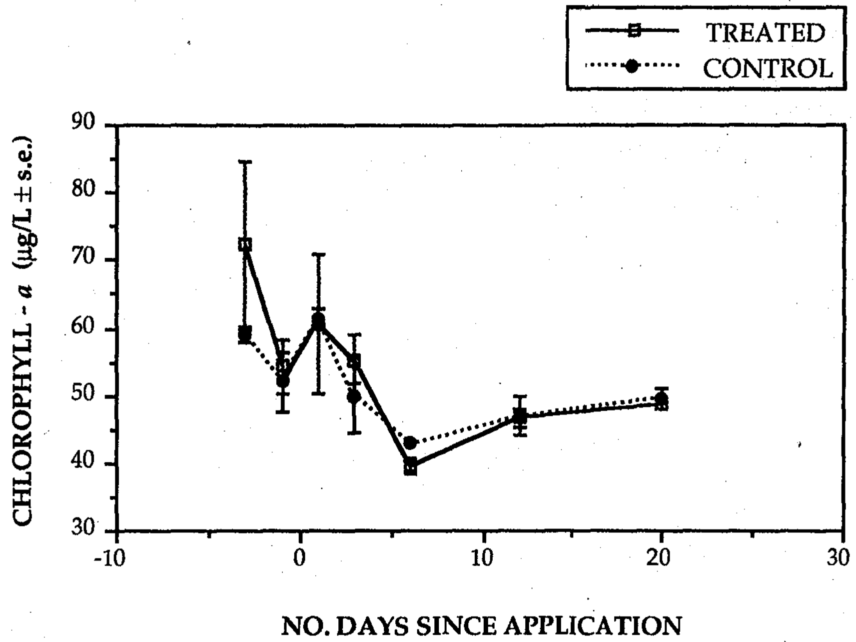

Figure 7.15 Changes in the concentration of chlorophyll-*a* in the treated and control enclosures.

DISCUSSION

The decision to treat North Lake with Sumilarv® in November was based on the densities of larval chironomids recorded on the October 22nd 1990. At that time, the density of chironomids in the lake was 5446 ± 889 larvae/m^2. It was considered important that densities be kept below a threshold level of 5000 larvae/m^2 to minimize nuisance swarms in the surrounding residential areas. The rate of emergence on the eastern side of the lake was monitored prior to the construction of the enclosures. On October 23rd, the rate of chironomids emerging from that area was 298 ± 71adults/m^2/night, of which *P. nubifer* constituted 85%. This rate of emergence and the combination of high larval densities and impending hot weather appeared likely to result in nuisance problems if the lake remained untreated. Chironomid emergence was measured again after the enclosures were in place and it was found that the emergence rate had not increased as expected, but had declined. In addition, *P. nubifer* had been replaced as the dominant emergent species by another chironomid, *T. bispinosus*. Prior to the pesticide application, the emergence of *T. bispinosus* was low and remained low for the two week period following application. The results of the statistical analysis suggest that Sumilarv® did not significantly inhibit the emergence of this species. This is in contrast with the results of field trials previously undertaken at North Lake (Davis *et al.* 1990). During those trials, Sumilarv® applied a similar rate effectively inhibited the emergence of all chironomid species present by more than 80% for at least a three week period. The chironomid species present were *P. nubifer*, *C. griseidorsum*, *Chironomus alternans* and *Kiefferulus intertinctus*. There are two possible explanations for the contrasting results. Firstly, Sumilarv® may have had some effect in suppressing the emergence of *T. bispinosus* but the patchy emergence of this species from the control enclosures generated large error terms which concealed these trends when the data were analysed Alternatively, *T. bispinosus* may be more tolerant to the pesticide than species tested previously.

Since Sumilarv® is an insect growth regulator and can inhibit the emergence of both mosquitoes (Schaefer *et al.* 1988) and chironomids (Davis *et al.* 1990), it also has the potential to inhibit the emergence of other insect species. In this experiment, there were no apparent differences in the emergence of the caddisfly *E. pansus* and the mayfly *T. tillyardi* between the two enclosure types. Although Sumilarv® did not appear to inhibit the emergence of these species, little confidence can be placed in this result since the number of emergent animals of both species was low and fluctuated so much that any discernible effect of Sumilarv® may have been masked.

Sumilarv® had no discernable, adverse effect upon the abundance of either early or late instar larvae for any of the species present. Similarly, previous field trials at North Lake did not identify any significant effect of the pesticide on chironomid larval abundance (Davis *et al.* 1990). Information obtained from the manufacturers of Sumilarv® suggested that the rate of application used in these experiments would have a detrimental effect on fourth instar chironomid larvae (Sumitomo, unpubl. data). An explanation for this anomaly may be that, by grouping of fourth instar larvae with earlier instar larvae, the effect upon the former is concealed. Alternatively, the species that Sumitomo tested may have been more sensitive to the product than any of the species which inhabit North Lake.

The reduction in the abundance of early instar larvae of *P. nubifer* and *C. griseidorsum*in both of the enclosures types may have been a result of the unusually low rate of emergence of these species during the experiment. By contrast, *T. bispinosus* was most abundant chironomid species emerging during the experiment and the density of early instar *T. bispinosus* larvae did not decline in the enclosures.

Sumilarv® did not reduce the abundance of phytoplankton as measured by chlorophyll–*a* concentration. This result is of particular importance since any detrimental effect upon phytoplankton production would affect the entire food chain.

The pesticide appeared to cause very little change to the populations of the non-target fauna. Zooplankton are considered to be amongst the more sensitive organisms to pollution, and yet, the two copepods, *C. tasmanica tasmanica* and *Mesocyclops sp.* did not appear to be sensitive to the Sumilarv® application. Similarly, the pesticide had no apparent effect on the abundance of the amphipod, *A. subtenuis* or the mayfly, *T. tillyardi*. The ostracod, *C. salinus* and the oligochaetes also demonstrated very little sensitivity to the Sumilarv® application.

The Bray-Curtis index was used to compare the two enclosure types and determine how similar they were, in terms of their species richness and abundance. No marked differences as a result of the Sumilarv® application were identified. At this point it may be pertinent to suggest that, within the bounds of this experiment, no detrimental effects upon the abundant non-target invertebrate species present in the treated enclosures resulted from the application of Sumilarv®.

To consider how this statement may be related to the biota within the lake, we need to examine how effectively the enclosures modelled the lake environment. Ideally, a field experiment should be performed at the same scale to which the results are to be extrapolated (Cooper and Barmuta, in press). Clearly this was logistically impossible under the circumstances of this experiment, and therefore *in situ* enclosures were utilized. Environmental parameters measured in the control enclosures and the lake were compared to determine any enclosure artefacts. Neither temperature, nor oxygen concentration deviated from that of the lake environment. The pH of the water in the lake was found to be higher (pH 8.0) than both the treated and control enclosures (pH 7.0) after the enclosures had been in place for almost three weeks. Whether this divergence is likely to influence the applicability of the experimental results is debatable. Schaefer *et al.* (1988) found that the stability of Sumilarv® in the water column increases with increased pH. Thus it could be argued that the more neutral conditions of the enclosures might have resulted in the decreased stability of the pesticide in the treated enclosures and thus reduced its biological affect. However, this is unlikely, given the small pH difference.

It is unfortunate that this field experiment was run in conjunction with the aerial application of Sumilarv® to the littoral region of the lake. Had the lake not been sprayed, it would have been possible to compare the biota from the lake and the control enclosures throughout the experiment to determine any 'cage effects' that the enclosures had upon the biological community. Instead, it was only possible to compare these areas in the period leading up to the pesticide application. Prior to application, no differences in biotic composition existed between the control enclosures and the littoral environment of the lake.

The enclosures appear to have satisfactorily mimicked the lake environment and thus the results of the experiment can be applied to the natural system. In doing so, the scale at which the results are expressed must be defined. Cooper and Barmuta (in press) suggest that an enclosure experiment of such small scale, both spatially and temporally, might be best applied to the study of macroinvertebrate populations, rather than community effects. With these limitations in mind, it could be suggested that Sumilarv® applied at the rate of 10 kg product/ha (0.05 kg AI/ha) would not adversely affect populations of the abundant non-target fauna at North Lake.

The invertebrate community at North Lake is unlikely to be representative of all Perth wetlands. Over the past century, much of the surrounding land has been cleared of vegetation, the lake has become nutrient enriched and its depth has increased due to agricultural and urban development. Insecticides (for chironomid control) and herbicides (for terrestrial weed control) have contaminated the water and exotic fish have been introduced. In addition, Sumilarv® had been applied to the lake on two occasions in 1989/90 and this could have resulted in reduced sensitivity of some species or the elimination of others. The combination of all these factors at North Lake could have encouraged a species composition more resilient to an application of the pesticide than would otherwise be the case. Sumilarv® may have a more detrimental effect on sensitive species and community composition at more pristine wetlands. However, it should be noted that truly pristine wetlands rarely support sufficient chironomid populations to cause nuisance problems and therefore these wetlands are less likely to require an application of Sumilarv®.

A recent U.S. study (Schaefer and Muira 1990) provided valuable information on the effects of Sumilarv® on organisms which were either not present, or not abundant in this study. The U.S. study investigated the effects of two different application rates of

Sumilarv® (0.05 and 0.11 kg AI/ha) on fauna inhabiting mosquito breeding areas. The abundant fauna included daphnid cladocerans, copepods, ostracods, oligochaetes, chironomids, damselflies, dragonflies, turbellarids, notonectids, gastropods and hydrozoa. The study found that at both rates, the pesticide application resulted in some suppression of daphnid cladoceran and ostracod populations. Neither rate of application affected chironomid larval populations, but predictably, the pesticide inhibited the development of chironomids into their adult form. In addition, there was some evidence to suggest that the pesticide caused the induction of morphogenetic abberations in emergent odonates (dragonflies and damselflies). These results are of particular relevance because, although a different formulation (emulsifiable concentrate) was used, the lower rate tested (0.05 kg AI/ha) was identical to the rate used in this study at North Lake and is the rate suggested for the effective control the chironomid nuisance species, *P. nubifer*, in Perth wetlands (Davis *et. al* 1990). Therefore, it is essential that the detrimental effects on aquatic fauna, which were detected by Schaefer and Muira (1990), be considered very carefully.

Schaefer and Muira (1990) considered the changes observed in cladoceran populations to be relatively minor. This is a somewhat subjective conclusion since the decline in the abundance of such important planktonic organisms could have serious implications for a wetland ecosystem. Cladocerans feed on algae and their presence in high numbers and subsequent grazing pressure has been shown to cause the rapid decline of phytoplankton abundance in lakes (Shapiro and Wright 1984; see also Chapter Three). At Lake Monger, a clear water phase is often associated with the presence of daphnid cladocerans in high numbers (M. Lund, pers. comm.). Chironomids feed upon algae and detrital material (Johannsson and Beaver 1983) and their presence in large numbers in wetlands is associated with high algal abundance. If cladocerans can contribute to the decline of algal biomass in wetlands, they could reduce the availability of the chironomid food source. Therefore their presence should be encouraged and the use of any pesticide that would reduce the presence of these organisms should be discouraged. Because cladocerans were not present in high numbers during this study at North Lake, the effect of Sumilarv® on these organisms could not be determined. In light of the results of the U.S. study (Schaefer and Muira 1990), the effects of Sumilarv® on cladoceran populations must be addressed.

Schaefer and Muira (1990) observed some abnormal and partially emerged dragonflies in the test ponds. Although this does not prove any adverse effects of the pesticide on the emergence of odonates, it is cause for concern. Dragonfly naiads are predators of chironomids. If Sumilarv® (at 0.05 kg AI/ha) does inhibit the normal emergence of these odonates, then it will, in turn reduce the naiad population and thereby diminish the potential for natural biological control of chironomid populations. Therefore further investigation of the effects of Sumilarv® on odonates in Western Australian wetlands is required.

At North Lake, no adverse effects of the pesticide on the population of the ostracod *C. salinus* was detected. The U.S. study found that the ostracod population was reduced by the Sumilarv® application. These contrasting results may be explained by the presence of different species of varying sensitivities between the two studies. However, this raises the issue of whether ostracod species which inhabit lakes other than North Lake might be harmed by a Sumilarv® application. Since ostracods are detritivores they compete with chironomids for their food source, and therefore, any detrimental effect on these organisms should be avoided.

The results of this experiment are promising and Sumilarv® appears to be a relatively safe compound for use in wetlands as part of an integrated approach to the control of nuisance chironomids. However, its widespread use cannot be recommended until further investigations have been undertaken to identify the effects of this compound on important aquatic organisms such as the cladocerans and odonates. Further investigations are already being undertaken by a member of the midge research team (K. Trayler) as part of an Honours project at Murdoch University. This is scheduled for completion in December 1991.

Chapter Eight

Conclusions and Implications for Management

Information obtained from the four years of research into Perth's midge problem will contribute to the better management of the problem in many ways.

Sumilarv®, an insect growth regulator with juvenoid activity has been identified as an alternative to the organophosphate pesticide, Abate®, for short term control of midges at wetlands where the latter no longer provides reliable control, for example, North Lake and Lake Monger. Laboratory trials indicate that an application rate of 10 kg product/ha (in water 50 cm deep) of Sumilarv® 0.5G should achieve good control in Perth wetlands. The suppliers of Sumilarv®, Wellcome Australia, have indicated that it will be available at a cost comparable to Abate® on a per hectare basis. Registration proceedings will start in 1992 and it is expected to be registered for use in Australia by 1994. Further trial use in wetlands may be permitted before then.

Results of field and laboratory tests of the effects of Sumilarv® on non-target aquatic fauna indicate that it appears to be a relatively safe compound for use at the proposed rate in the urban wetlands as part of an integrated approach to the control of nuisance chironomids. Widespread use however cannot be recommended until further investigations into the effects of this organism on important aquatic organisms such as cladocerans (water fleas) and odonates (dragonflies) are completed. These organisms did not occur in large enough numbers to be tested during the North Lake field trial but a recent U.S. study found that both groups were affected by Sumilarv®. Their results have serious implications because both cladocerans and odonates potentially exert either direct or indirect natural biological control on chironomid populations. Naiad (larval) odonates prey directly on larval chironomids while cladocerans (in particular, *Daphnia* sp.) act to reduce algal biomass in wetlands and so reduce an important chironomid food source. Further investigations are currently being undertaken by a member of the midge reasearch group (K. Trayler) as part of an honours project at Murdoch University which is scheduled for completion in December 1991.

Considerable care needs to be exercised with the large scale application of Sumilarv® to wetlands. The impact of an accidental spill on the aquatic ecosystem would be serious given that ten times the application rate could kill 50% of the mayflies and harm 50% of the amphipods present. We do not have any information on the effect that a spill of Abate® would have on a wetland where it had not been previously used. However, a similar or even greater effect from this broad spectrum pesticide could be expected.

Research by Mulla *et al.* (1979) in the United States indicated that Abate® can affect some non-target invertebrates at rates similar to those currently used for midge control. Great care must be exercised with the use of any pesticide in natural aquatic ecosystems. Patchy applications must be avoided because they will leave large areas untreated and sufficient midges present to recolonise treated areas. Use of a helicopter or boat appears preferable to the use of fixed wing aircraft for effective delivery of pesticide to target areas.

The results of this research program indicate that short term control of midge problems will be improved by monitoring larval densities, water depth and temperature so that larval thresholds and critical values can be used to predict appropriate times to treat. Pesticides (either Abate® or Sumilarv®) should not be applied unless the means to monitor the effectiveness of a treatment are also available. For Sumilarv® this means that rates of adult emergence must also be monitored in addition to larval densities. This can be simply achieved with the use of emergence traps. Monitoring programs similar to those already undertaken by the City of Stirling (Smyth 1991) and other municipal authorities must be established by all agencies responsible for the management of wetlands where midge problems occur. It is unlikely that midge problems will be successfully controlled if the status of midge populations at a wetland is not known. In addition, to reduce chironomid nuisance problems adjacent to a particular lake the populations of midges at nearby lakes also needs to be considered because of the dispersal ability of adult chironomids.

The onset and duration of midge problems at Forrestdale Lake appear to be determined primarily by water temperature and conductivity, both of which are related to lake depth. A rapid rise in the density of *P. nubifer* appears to occur four to six weeks after the median water temperature rises above 22 °C, unless the lake depth has fallen below 40 cms (in which case the lake will dry before *P. nubifer* reaches nuisance levels). If the lake depth is between 40 and 60 cms when the median temperature rises above 22 °C, a problem may still occur but it is likely to be shortlived. The longer the lake contains water during summer after the critical median weekly temperature has been exceeded, the longer problems with *P. nubifer* are likely to continue and to require intervention in the form of pesticide treatments. Abate® still appears to be an effective compound for midge control at Forrestdale Lake but proper timing of treatments is essential for good control.

Manipulation of lake depth leading to earlier summer drying in some years may be an effective means of controlling midge problems at Forrestdale Lake. While it may be argued that this approach would not be wholly compatible with the nature conservation role of the lake, nor is the use of pesticides. In addition, such manipulation may not be compatible with the control of Typha (bullrushes) at this lake. Similar methods are currently being investigated for the non chemical control of mosquitoes and ceratopogonids (biting midges).

The most constant predictor of larval midge density at North Lake was maximum temperature. Densities exceeding 8 000 larvae/m^2 generally occurred three to seven weeks after the maximum weekly temperature reached 20 ^{o}C. Although this information is of some practical value in indicating when midge problems may be about to develop it is too imprecise for decisions regarding the timing of pesticide treatments. Monitoring of larval densities, in conjunction with adult emergence, remains the best method for assessing whether or not a pesticide treatment is appropriate at North Lake.

Differences between lakes in terms of degree of enrichment, physico-chemical attributes and composition and abundance of midge populations means that no universal formula for the short term solution of midge problems can be achieved. Reliance on pesticides will not solve problems indefinitely as resistance will develop. All pesticides, including narrower spectrum compounds such as Sumilarv®, will affect wetland food chains in an adverse way. An integrated approach to midge control is required where several control techniques are applied simultaneously. For example the use of light traps in addition to limited pesticide treatments plus a reduction in nutrient inputs and the replanting of fringing vegetation are all techniques which can be used in combination to good effect at many wetlands. Manipulation of wetland water levels, as already discussed for Forrestdale Lake, may represent an alternative, non-chemical, short term solution to midge problems.

The problem of nutrient enrichment must be addressed as the key component of an ecologically sensitive midge control strategy for the wetlands of the Swan Coastal Plain. Research and monitoring undertaken during the final year of this study have provided evidence that midge problems are largely fuelled by algal blooms. Midge problems are a symptom of a disturbed system and an effect of poor water quality. The water quality of wetlands must be improved for a longer term solution to midge problems. Reduction of external inputs by diversion of inflowing drains, similar to the recent diversion of the veterinary farm drain at North Lake represents one possible approach. The conversion of septic tanks to mains sewerage needs to be considered at Forrestdale where septic leachate may be the main source of diffuse nutrient input to the lake. Other approaches that warrant serious consideration include the reduction of fertiliser use in wetland catchments and the establishment of buffer zones of fringing vegetation of sufficient width, height and structure to maintain wetland ecosystem function, filter nutrients from surface and subsurface runoff and to act as a mechanical barrier to midge swarms.

Appropriate urban planning is vital to prevent midge problems occurring at future residential developments sited near wetlands. In the absence of any specific research it is recommended that no development should be sited closer than 800 metres to a wetland and that any landuse activity proposed for within that distance should be wholly compatible with the objectives of wetland conservation. The removal of fringing vegetation must be avoided at all cost. Vegetation provides the basis of all wetland food chains and is integral to natural wetland structure and function. Its removal for urban or agricultural development is usually accompanied by the onset of excessive nutrient enrichment and subsequent midge problems. Problems with midges should be rare in non enriched wetland habitats where little fringing vegetation has been removed.

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Glossary

Analysis of variance: A statistical technique that allows the researcher to state with a specified level of confidence whether or not two or more data sets are significantly different from each other. For example, a confidence level of 95% ($P<0.05$) means that there is only a 5% chance of concluding that the data sets are different when in fact there are not. Basically, the decision is made by comparing the variation within the data sets to the variation between data sets.

Bioassays: tests which measure the response of living organisms to toxicants

Biomass: the total mass of living matter under consideration, e.g. algal biomass or invertebrate biomass.

Bioturbation: process by which benthic animals increase the transfer of nutrients from the sediment to the water column by physical means such as movement and burrowing

Blue-green algae: a class of unicellular, colonial or filamentous algae that contain unique pigments which often appear blue

Chlorophyll - *a*: a green pigment found in plants and involved in photosynthesis

Chlorophyta: green algae whose green pigments are not masked by other pigments

Clear water phase: phenomena whereby the water of a eutrophic lake will become clear, often as a result of zooplankton grazing on algae

Conductivity: measure of the ability of the water to conduct electricity which is dependent on the concentration of dissolved salts in solution

Correlation coefficient: a measure of the degree of relationship between variables

Cyanophyta: blue-green algae (see definition above)

Detritivores: an organism that feeds on detritus

Detritus: accumulations of dead and decaying animal and plant material

Diel emergence: patterns of emergence that occur in rhythms every 24 hours

Endogenous: metabolic or behavioural rhythms that originate within an organism and persist even though external conditions are kept constant

Eutrophic: the description given to lakes that are highly enriched with plant nutrients and which usually support very high levels of algal or macrophyte production

External loading: nutrient input to a water body from groundwater flow and surface runoff

Heteroscedasticity: the situation that exists when two sets of data do not have equal variance

Hypereutrophic: the description given to those water bodies that are extremely eutrophic (see eutrophic above)

Indigenous: native

Insect growth regulator: substance that interferes with growth and development of an insect. See Davis *et al.* (1990) for a description of how Sumilarv® works

Internal loading: transfer of nutrients from sediments to the overlying water

Ion: positively or negatively charged atoms or molecules that result when salts and other chemicals dissolve in water

Juvenoid: substance which can mimic the action of a natural juvenile hormone in insects

Larvae: juvenile stage of some insects that do not possess wings and must pupate before forming the adult stage

Littoral: zone of shallow water around the perimeter of a lake

Macrophyte: larger plants, as opposed to microscopic algae

Monomorphic: development of an insect with little or no change as it matures

Morphological aberrations: abnormalities in form

Naiad: nymph stage (see nymph below)

Nitrogen fixation: the biochemical conversion of elemental nitrogen (N_2) to ammonia (NH_3) performed by some blue-green algae and bacteria. An important process in lakes since most algae cannot use elemental nitrogen but can use ammonia

Nymph: juvenile stage without wings or with incomplete wings. Insects which have nymphs (dragonflies, mayflies) do not have to pupate to reach the adult form, unlike insects which have a larval stage (midges, mosquitoes) which must first form a pupa before becoming an adult

pH: a measure of the relative abundance of hydrogen ions in the water

Photosynthesis: the biochemical process by which plants convert carbon dioxide into organic carbon compounds using energy from the sun

Phytoplankton: small plants (mostly algae) suspended in the water column

Polymorphic: development of an insect whereby the individual undergoes changes in form as it matures

Soluble reactive phosphorus: organophosphate

Standard error: a measure of the amount of variation between samples

Tannins: complex compounds occurring in the bark and leaves of various trees and which can cause the brown pigmentation in water

Trophic status: A range of classifications used to describe the productivity of a wetland. These range from oligotrophic (very low productivity) through mesoeutrophic (moderate productivity) to eutrophic and hypereutrophic (high to very high productivity). The latter usually have high productivity due to nutrient enrichment

Zooplankton: small animals with weak locomotory power that live in the water column