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FERRIC DOSING OF RESERVOIRS: A REVIEW OF DOSING AND ITS IMPACT IN ANGLIAN REGION, NORTHERN AREA DURING 1994/5.

National Rivers Authority
Anglian Region

NORTHERN AREA BIOLOGY LABORATORY.

AUGUST 1995.

FERRIC DOSING OF RESERVOIRS: A REVIEW OF DOSING AND ITS IMPACT IN ANGLIAN REGION, NORTHERN AREA DURING 1994/5.

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EXECUTIVE SUMMARY

Monitoring of chemical (water and sediment), physical, phytoplankton, zooplankton and benthic invertebrates at Rutland Water, Covenham and Pitsford Reservoirs was maintained throughout 1994.

Results continue to show that the benthic invertebrates are affected at both an acute and chronic level. Communities from contaminated areas have reduced densities and diversities. There are also indications that chironomid larval development has been delayed in areas with high floc concentrations (as predicted by Radford (1994) and also found in laboratory tests).

Ferric sulphate dosing, although possibly successful at reducing P, has not controlled phytoplankton and cyanobacteria in particular.

At Covenham, where P control in 1994 was not as effective as in previous years and where P levels were still quite low, the phytoplankton have reverted to pre-dosing levels and succession patterns highlighting the instability of these reservoir systems.

It is emphasised, again, that P is just one (although a major one) of many controlling forces in lake ecology. As many of these factors/forces as possible must be investigated and addressed (eg. fish/zooplanton/plant interactions, mixing regimes, hydraulic regimes etc.) before lake management and eutrophication control can be effective.

Preliminary results from the analysis of heavy metals in the W grade ferric sulphate are presented.

The continued use of direct dosing of ferric sulphate for P control is questioned and recommendations for future eutrophication management are made. Suggestions for future work are outlined.

CONTENTS.

1. INTRODUCTION

Anglian Water Services have continued to directly ferric dose their water supply reservoirs with the aim of reducing water column phosphorus and to control cyanobacterial blooms and scums.

Northern Area Biology staff (NRA) have continued to monitor the effects and effectiveness of ferric dosing at Rutland Water, Covenham and Pitsford reservoirs.

Previous reports (Extence et al. 1991, 1992a, 1992b, Christmas et al 1994) have reported the adverse effect that direct dosing has had on both sediment chemistry and benthic faunal communities. This report summarises some of the findings from routine NRA monitoring during 1994 (and early 1995 for benthic invertebrates at Rutland Water).

A summary of non-routine monitoring and relevant projects is also included. Preliminary results from the analysis of trace metal contaminants in W grade ferric sulphate is also included.

2. METHODOLOGY.

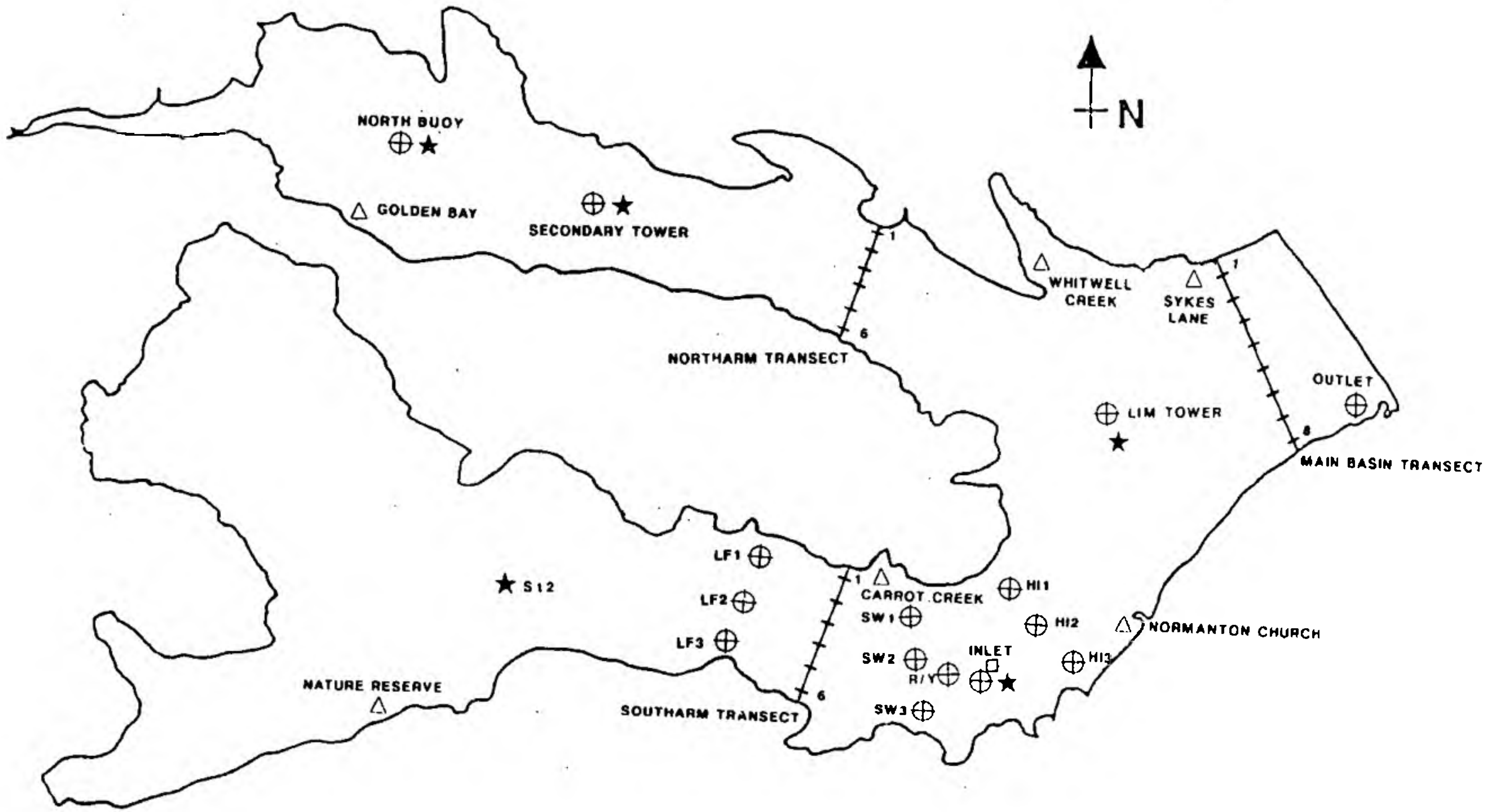
2.1 SAMPLING METHODS.

The sampling and sorting methods used in the benthic and littoral macroinvertebrate surveys and the water quality and phytoplankton surveys remained unchanged and have been documented in previous reports (Extence et al. 1992, Christmas et al 1994). The sampling sites were also unchanged and are shown in Figure 1. for Rutland Water and Figure 2 for Covenham. On the figures and throughout the report the following abbreviation have been used: IN for the Inlet, SW for the Slipway transect, LF for the Lodge Farm transect, HI for the Howell's Inlet transect, N1 for the North Buoy, ST for the Secondary Tower, S3 for the South Arm transect and N2 for the North Arm transect.

2.2 DATA ANALYSIS.

The techniques of cluster analysis and multi-dimensional scaling have been used to interpret patterns in community structure. Analyses were carried out using the computer package PRIMER. The data was 4th root transformed to increase the contribution of rarer species to the analysis, and Bray-Curtis sample similarity coefficients were calculated. Dendrograms were produced using group average linking. Ordination plots showing the relative similarity of samples in two dimensions were produced using multi-dimensional scaling.

FIG. 1. MAP OF RUTLAND WATER SHOWING ALL NRA ROUTINE SAMPLING SITES.

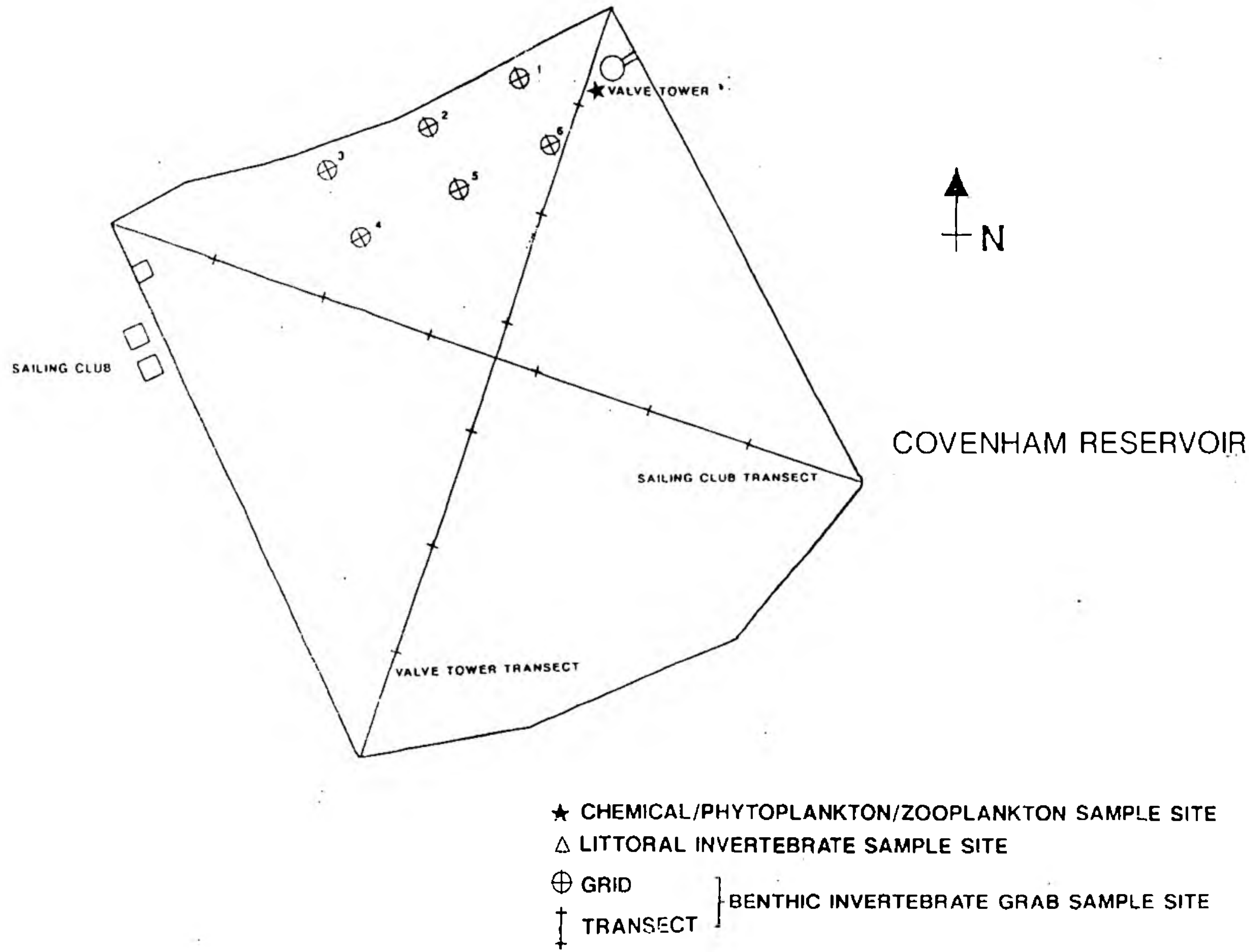


★ CHEMICAL/PHYTOPLANKTON/ZOOPLANKTON SAMPLE SITE

△ LITTORAL INVERTEBRATE SAMPLE SITE

⊕ GRID
+ TRANSECT
} BENTHIC INVERTEBRATE GRAB SAMPLE SITE

FIGURE 2.0 MAP OF COVENHAM RESERVOIR SHOWING ALL NRA ROUTINE SAMPLING SITES.



3. RESULTS AND DISCUSSION - BENTHIC AND LITTORAL MACRO-INVERTEBRATES AND SEDIMENT CHEMISTRY.

The results and discussion presented here are a summary of some of the benthic macroinvertebrate and sediment chemistry investigations carried out from January 1994 to June 1995 for the Rutland Grid, Rutland Littoral and Covenham Grid surveys, and includes some historical data for comparison.

3.1 RUTLAND WATER - GRID SURVEYS.

3.1.1 Sediment analysis.

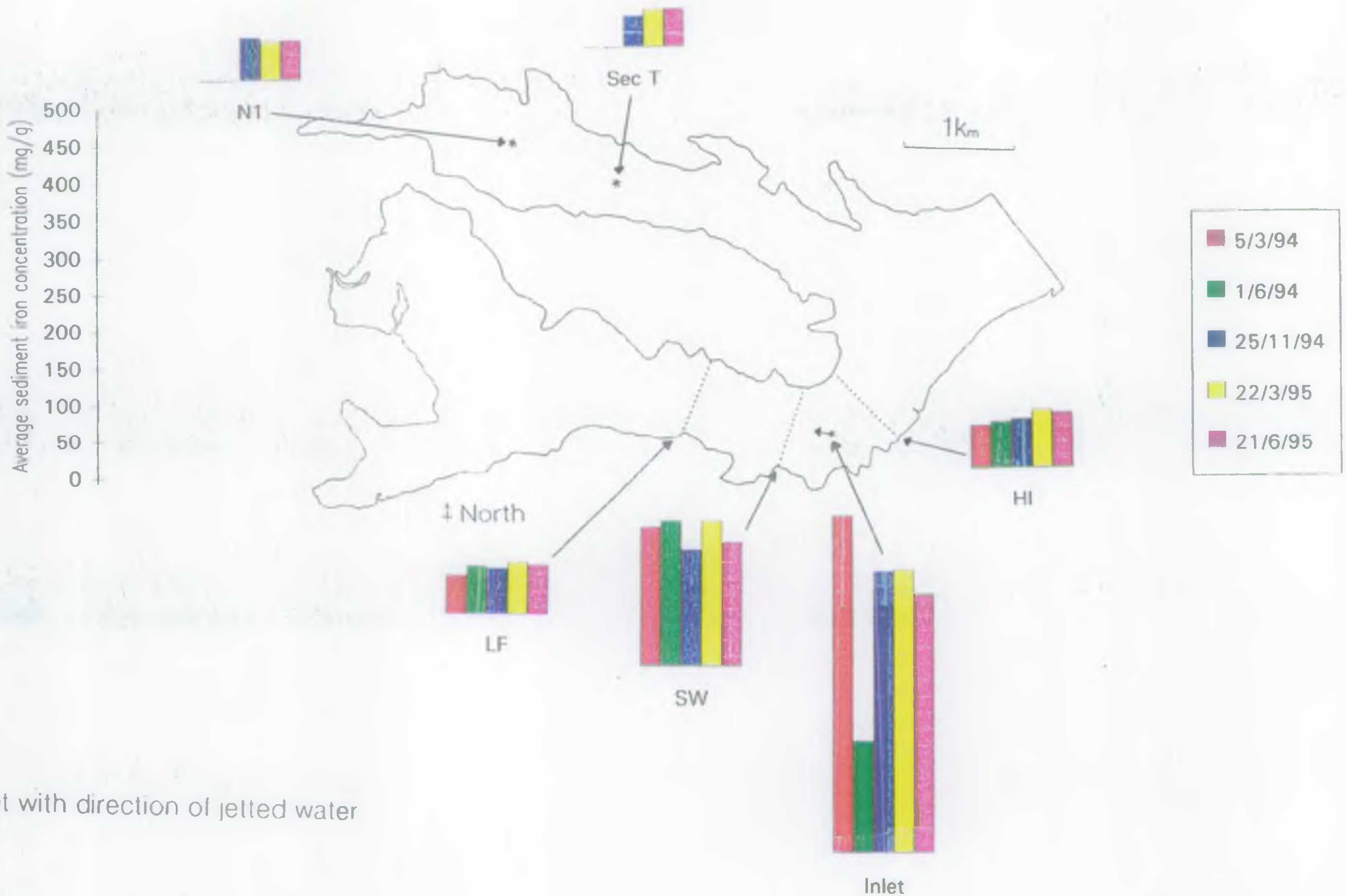
Average sediment iron concentrations in 1994/95 for the sites and transects on the grid survey are shown in Figure 3 (for a comparison with historical levels see Figure 1, Appendix A, which shows 1993 data). Monthly ferric sulphate inputs to the reservoir for the period 1994/95 are shown in Figure 4 and monthly river water inputs to the reservoir are shown in Figure 5 (data supplied by Anglian Water Services).

Sediment iron concentrations remained similar to previous years (Christmas et al. 1994). At the "control sites" (North Buoy and Secondary Tower) between 40 to 62.3 mg iron/g dry weight of sediment were present. Levels were slightly increased for the Howell's Inlet and Lodge Farm transects with iron concentrations of between 47.7 to 97.7 mg Fe/g and 43.5 to 98.1 mg Fe/g respectively. Ferric contamination remained high at the Slipway and Inlet sites with concentrations of between 120 to 314 mg Fe/g iron recorded for the Slipway sites and between 123 to 487 mg Fe/g for the Inlet sites.

The average sediment iron concentration recorded at the Inlet on 1/6/94 was unusually low compared to the other surveys. Prior to this period ferric dosing was carried out at very low levels, whilst the river water pumped into the reservoir would have been high in suspended solids throughout the winter. It is likely that the ferric floc was overlain with uncontaminated sediment, reducing the concentration of iron in the sediment. The same effect was not seen in June 1995 following a period of low/no dosing because pumping of river water into the reservoir was also suspended between January 1995 to May 1995.

A decline in the average sediment iron concentration at the Inlet seems to have occurred since January 1994. This may have been due to the reduced levels of ferric dosing since 1993 compared to previous years (3956 tonnes of ferric sulphate were dosed in 1990, 18902 tonnes were dosed in 1991, 16561 tonnes in 1992, 2511 tonnes in 1993, 2568 tonnes in 1994 and 1485 tonnes between January to April in 1995), and a subsequent dilution of iron in the sediment due to natural sedimentation and introduction of sediment with in-flowing river water. On the Howell's Inlet and Lodge Farm transects however, the average sediment iron concentration increased since January 1994, possibly indicating that the ferric floc is spreading.

Figure 3. Average sediment iron concentrations at sites on Rutland Water in 1994/95



↔ Inlet with direction of jetted water

FIGURE 4. RIVER INPUTS TO RUTLAND WATER AND WATER LEVEL, 1994.

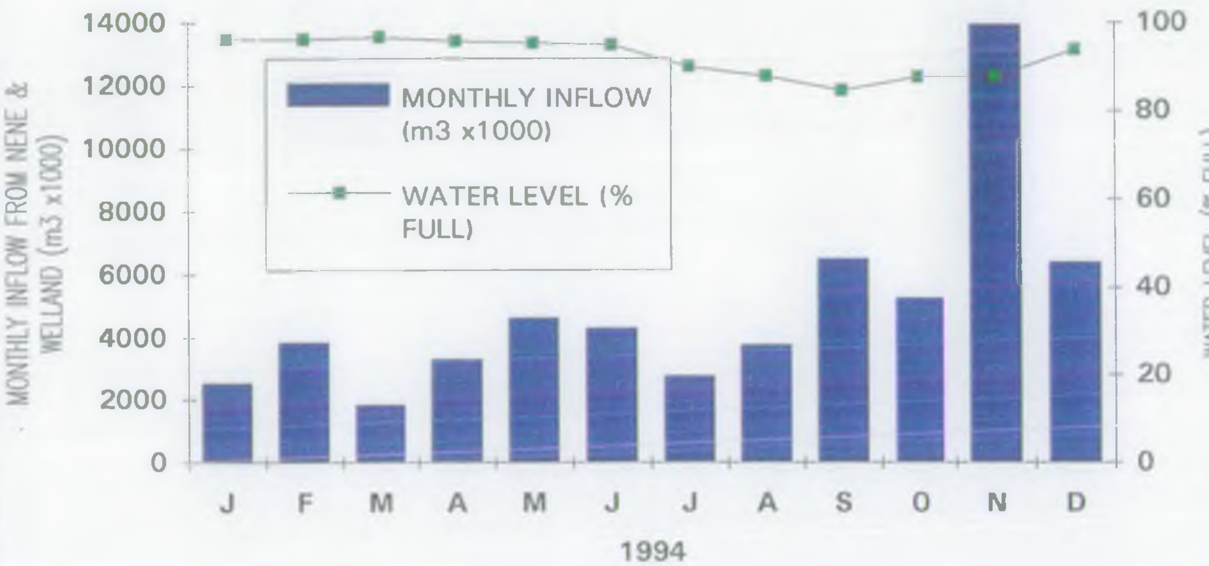
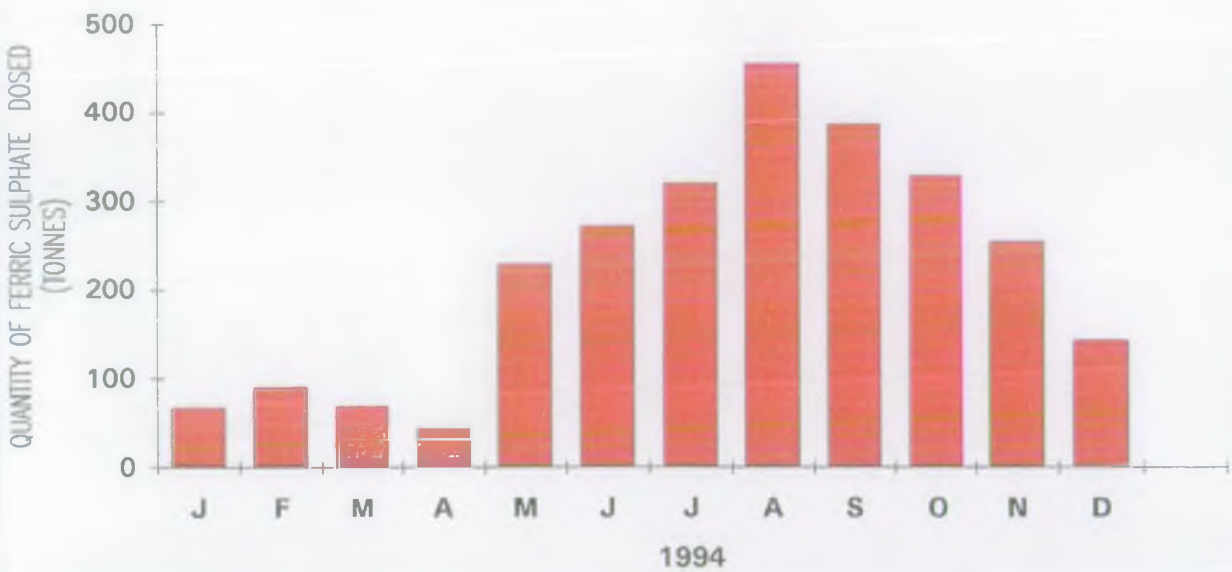


Figure 5. MONTHLY QUANTITY OF FERRIC SULPHATE DOSED DIRECTLY INTO RUTLAND WATER DURING 1994



3.1.2 Benthic macroinvertebrates.

The average number of taxa per m^2 recorded during the 1994/95 grid surveys are shown in Figure 6. Lowest average numbers of taxa (2 to 5 taxa/m^2) were recorded for the Inlet and the Slipway transect sites. At the North Buoy and the Secondary Tower an average of between 4 and 8 taxa/m^2 were recorded. Lodge Farm and Howell's Inlet transect sites had increased average diversities with between 5 to 10 taxa/m^2 recorded. The highest numbers were due to the presence of large clumps of *Dreissena polymorpha* in the samples. *Dreissena polymorpha* clumps tend to provide extra habitat for other species (see Section 3.1.5). A comparison with data from 1993 can be made by referring to Figure 2, Appendix A

An increase in the average numbers of taxa at all the sites occurred during 1994/95, and was a result of temporal population variations, as have been reported previously (Bullock *et al.*, 1982, Christmas *et al.*,1994).

On 1/6/94 the average number of taxa recorded at the Inlet was low, indicating that the fauna had not recovered despite the lower iron levels recorded at this site.

The average densities of macroinvertebrates (numbers of individuals per m^2) recorded for the grid surveys are shown in Figure 7. Lower average densities were recorded at the Inlet and on the Slipway transect (308 to 5918 individuals/m^2) compared to the North Buoy and the Secondary Tower (2618 to 10758 individuals/m^2) and the Lodge Farm and Howell's Inlet transects (2838 to 10758 individuals/m^2). (See Figure 3, appendix A for a comparison with 1993).

An increase in average macroinvertebrate densities was apparent throughout 1994/95 at the Inlet and on the Slipway transect, with numbers recorded in the most recent survey (June 1995) equivalent to those recorded at the "control" sites. No similar temporal pattern was evident at the North Buoy, the Secondary Tower or on the Lodge Farm transect, though at the Howell's Inlet transect there was a large increase in macroinvertebrate densities in June 1995. At the Inlet and on the Slipway transect the increases may indicate a possible recovery of the communities at these sites.

The elevated average number of individuals shown for the Howell's Inlet transect in June 1995 was due to the presence of very large *Dreissena polymorpha* clumps at two of the sites.

The average densities of chironomids (numbers per m^2) recorded for the grid surveys are shown in Figure 8. Lowest average densities of between 220 to 3725 chironomids/m^2 were recorded at the Inlet and on the Slipway transect. At the North Buoy and the Secondary Tower and on the Howell's Inlet and Lodge Farm transects densities were greater (between 1826 to 8492 chironomids/m^2), though temporal variation was high. The chironomid densities recorded at the Inlet and on the Slipway transect in June 1994 and June 1995 were higher when compared to the other surveys, which may indicate a possible recovery of the communities (see also Figure 4, Appendix A for comparison with 1993 data). Sediment iron

Figure 6. Average number of taxa at sites on Rutland Water in 1994/95

cf 93/4?

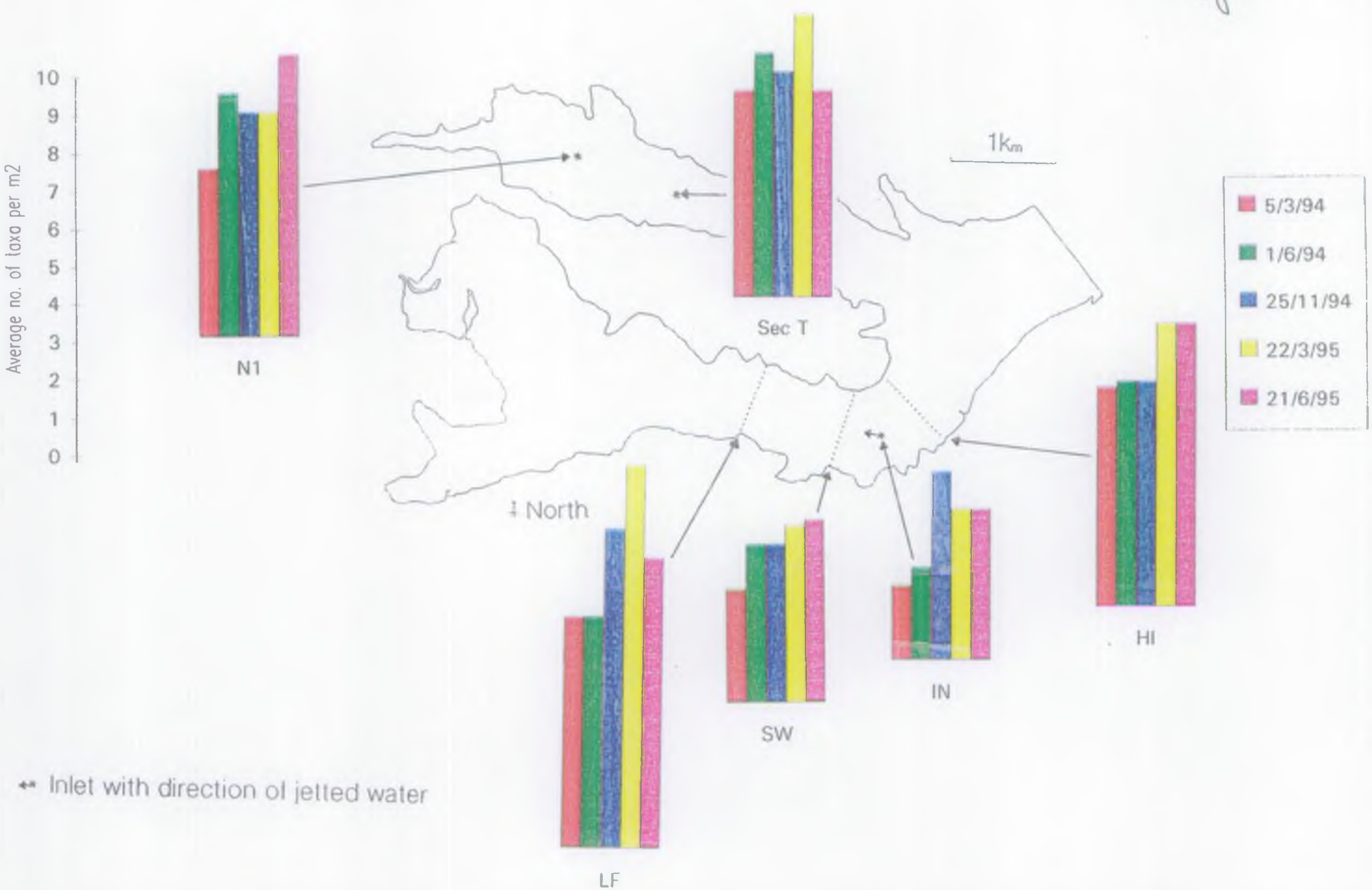


Figure 7. Average number of individuals at sites on Rutland Water in 1994/95

cf 93/4?

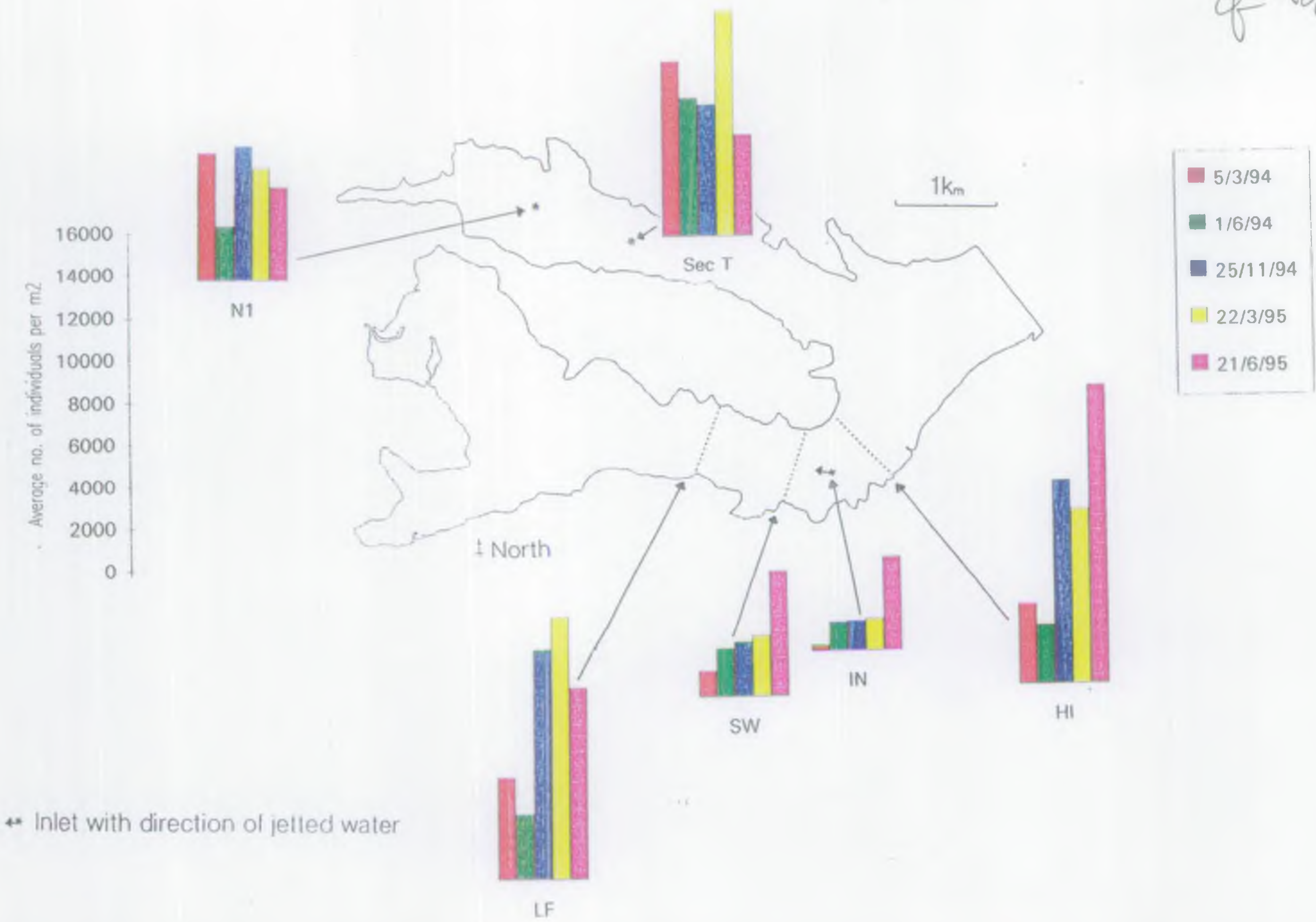
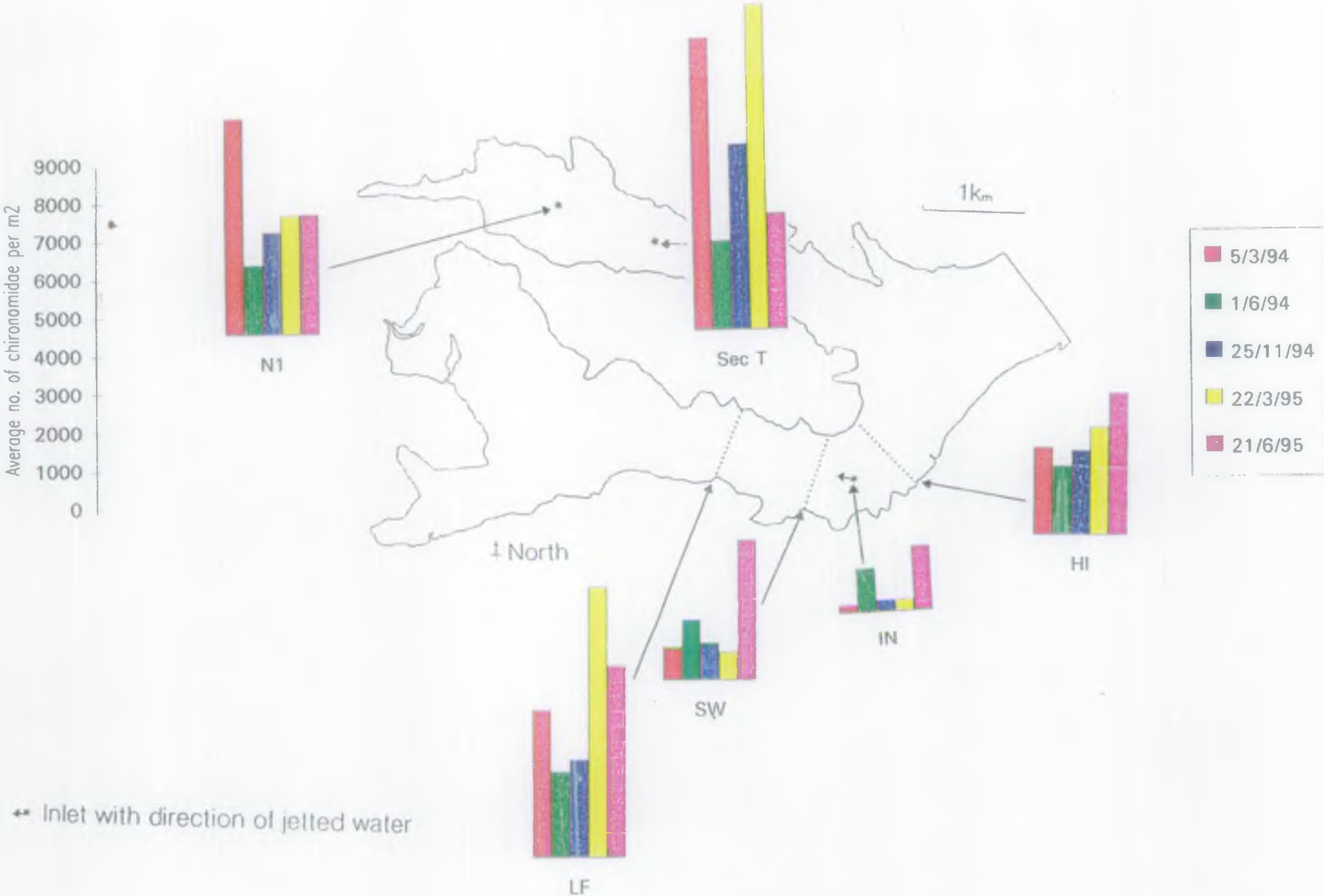


Figure 8. Average number of chironomidae at sites on Rutland Water in 1994/95



concentrations were reduced at these times at the Inlet, probably due the reduced dosing in the months prior to June in both years. However sediment iron concentrations were not reduced in June 1994 at the Slipway.

At the North Buoy, the Secondary Tower and on the Howell's Inlet and Lodge Farm transects annual variations in chironomid densities had peak numbers occurring in March and lowest numbers in June. A similar pattern was recorded by Bullock *et al.*, (1982) at Rutland Water between 1976 and 1981.

The reductions in larval numbers in June were probably due to the emergence of adult chironomids, whilst the peaks in March may have been artificial, probably being due to the higher proportions of larger, final instar larvae; these larvae are much easier to count (Bullock *et al.*, 1982).

The opposite pattern occurred at the contaminated sites (the Inlet and the Slipway transect), with peak numbers in June and lowest numbers in March. Apart from the obvious reduction in chironomid numbers at the contaminated sites (chronic effect), there may have been a sub-lethal effect of the ferric floc on the chironomid larval life cycle. Radford (1994) found that *Chironomus riparius* larvae exposed to ferric precipitates showed reduced growth and development resulting in delayed emergence. This could have been occurring at the affected sites in Rutland Water.

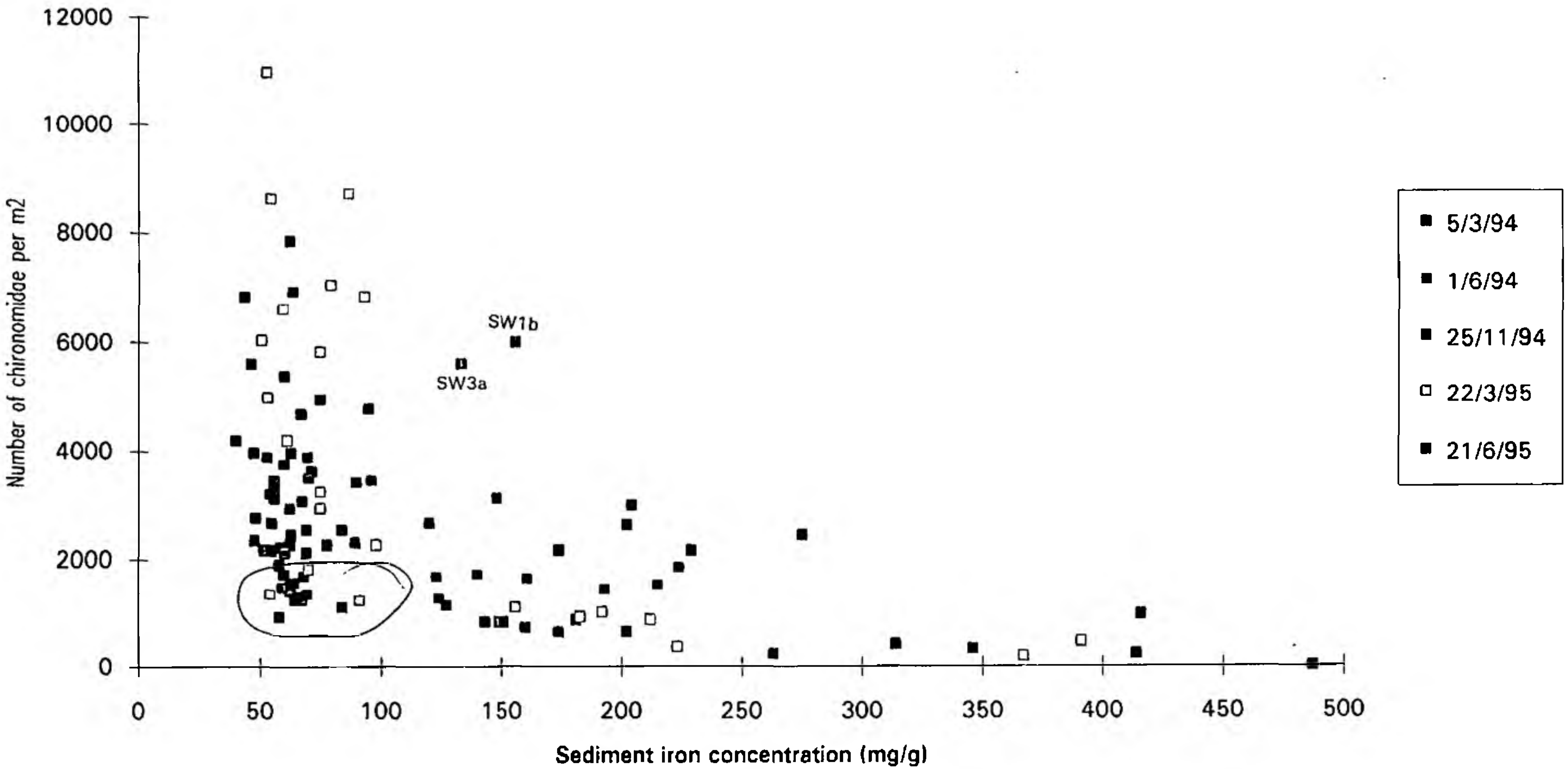
Alternative reasons for the increased densities of benthic invertebrates at certain sites may have been that populations were developing tolerance and adapting to the ferric contamination. Also, if the growth and development of the chironomid larvae at the affected sites was inhibited, larvae in March might have been smaller and large final instar larvae might still have been present in June due to delayed emergence. This could have affected the numbers recorded during these grid surveys, causing peak numbers in June which might be interpreted as a recovery of the communities.

3.1.3 Relationship between sediment iron concentration and chironomid densities.

Figure 9 shows the relationship between sediment iron concentration and chironomid densities. A marked decline in chironomid density was noted where sediment iron was greater than approximately 100 mg Fe/g (Brierley, 1992) and the pattern seen in Figure 9 agreed with that recorded in previous reports (e.g. Christmas *et al,* 1994).

There were two particular points which did not fit the pattern, these corresponded to sites SW3a and SW1b from the survey of 21/6/95 where chironomid numbers were moderately high despite sediment iron concentrations of approximately 150 mg Fe/g. It is possible that a recovery may have occurred at these sites, as mentioned above, however it is also possible that the increases at these sites for this survey were due to delays in chironomid development.

Figure 9. Relationship between number of chironomidae and sediment iron concentration at Rutland Water in 1994/95



of what? Nos &/or species? + habitat?

3.1.4 Faunal associations.

Cluster analysis and multi-dimensional scaling were used to aid interpretation of the patterns in the community structure for the grid surveys carried out on 5/3/94, 1/6/94 and 22/3/95.

3.1.4.1. March 1994.

The dendrogram from the cluster analysis for the March 1994 survey is shown in Figure 10. Three major groups were evident at the 59% similarity level. The main characteristics of these groups are identified in Table 1.

At the Inlet (replicate a) the macroinvertebrate fauna was very impoverished, only oligochaetes were present in low numbers (220 oligochaetes/m^2). The sediment iron concentration was highest at this site (487 mg Fe/g).

Group 3 also contained Slipway and Inlet sites which had impoverished fauna with 2 to 3 taxa/m^2 present and low invertebrate densities of 396 to 1188 individuals/m^2. The sediment iron concentrations at these sites were also high (between 143 to 414 mg Fe/g).

Group 1 contained sites where macroinvertebrate densities and abundances were high (9 taxa/m^2 and 3212 to 9328 individuals/m^2). *Dreissena polymorpha* was a dominant component of the fauna at both of these sites; the presence of this species has been shown to increase community diversity and abundance (see Section 3.1.5) The sediment iron concentrations at these sites were low (62.9 to 67.4 mg Fe/g).

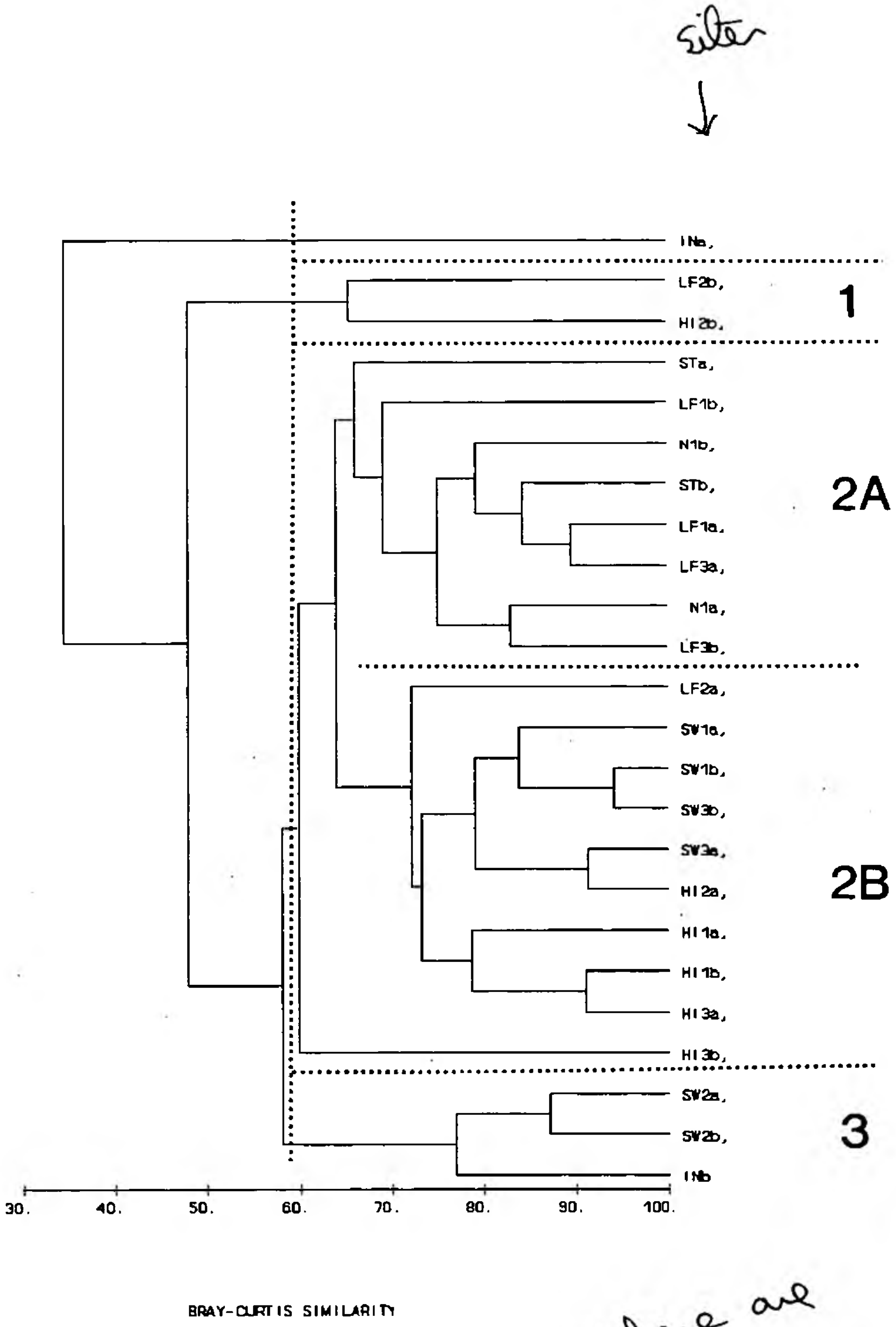
Group 2 contained sites with intermediate diversity and abundance (3 to 7 taxa/m^2 and 1056 to 9328 individuals/m^2). This cluster was divided into 3 sub-groups. Sub-group A contained North Buoy, Secondary Tower and Lodge Farm transect sites which generally had greater abundances of macroinvertebrates (3212 to 9328 individuals/m^2) than the Slipway, Howell's Inlet and Lodge Farm sites in Sub-group B (1056 to 4928 individuals per m^2). Sites in Sub-group A also had some different subdominant species (*Ocetis ochracea* and Ceratopogonidae) compared to Sub-group B (*Pisidium* sp.).

The average sediment iron concentration for sites in Sub-group A was 49.75 mg Fe/g and for sites in Sub-group B was 98.8 mg Fe/g, however there were sites present in Sub-group B (LF2a, HI1a, HI1b, HI2a and HI3a) which had sediment iron concentrations at background levels (40 to 60 mg Fe/g).

Site HI3b occurred in Group 2 separately from Sub-groups A and B, but differed only from the subgroups in the sub-dominant species present. Again the sediment iron concentration was low (51.5 mg Fe/g).

The MDS plot is shown in Figure 11a and Figure 11b shows the same MDS plot, but with circles whose size is proportional to the sediment iron concentration at each site.

Figure 10:
Cluster analysis dendrogram of data from Rutland Grid survey carried out on 5/3/94



where are these sites

Table 1: Cluster analysis groupings for Rutland Grid Survey 5/3/94

GROUP	SITES	AVERAGE NUMBER OF TAXA	AVERAGE NUMBER OF INDIVIDUALS	TYPICAL TAXA (dominant in bold)	AV. SEDIMENT IRON CONC.
Unclassified	INa	1	220	**Oligochaeta**	487
Group 1	LF2b, HI2b	9	6930 (3212-9328)	**Chironomidae/*Dreissena*** , Oligochaeta, *Valvata piscinalis*, *Pisidium* sp., *Crangonyx pseudogracilis*, *Asellus aquaticus*	65.15 (62.9-67.4)
Group 2a	STa, STb, N1B, N1a, LF1a, LF1b, LF3a, LF3b	5 (4-7)	5951 (3212-9328)	**Chironomidae**, Oligochaeta, Nematoda, *Ocetis ochracea*, *Valvata piscinalis*, Ceratopogonidae	49.75 (43.5-59.9)
Group 2b	LF2a, SW1a, SW1b, SW3a, SW3b, HI1a, HI1b, HI2a, HI3a	4.5 (3-7)	2424 (1056-4928)	**Chironomidae**, Oligochaeta, Nematoda, *Valvata piscinalis*, *Pisidium* sp.	98.8 (54.8-215)
Group 2 unclassified	HI3b	5	3080	**Chironomidae**, Oligochaeta, *Helobdella stagnalis*, *Valvata piscinalis*, *Bythinia leachii*	51.5
Group 3	SW2a, SW2b, INb	2.3 (2-3)	689 (396-1188)	**Chironomidae**, Oligochaeta	290 (143-414)

Figure 11a:
Ordination plot (MDS) of data from Rutland Grid survey of 5/3/94 (stress = 0.13)

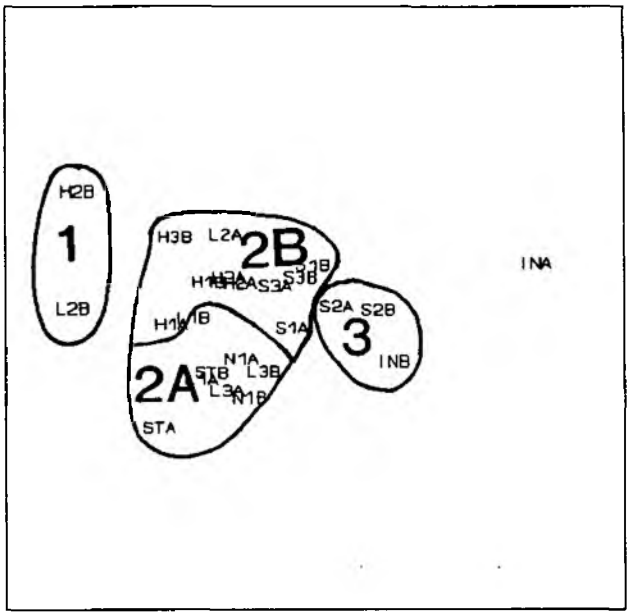
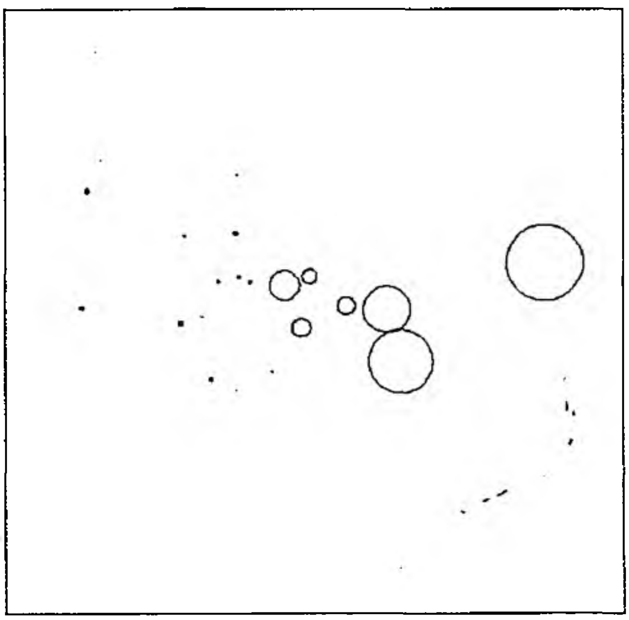


Figure 11b:
Ordination plot (MDS) of data from Rutland Grid survey of 5/3/94 with sediment iron data superimposed as circles (stress = 0.13)



what?

The ordination (MDS) plot generally agreed with the cluster analysis dendrogram and placed the sites in the cluster groups close to each other. From observation of Figure 11b it was apparent that there was a gradient related to sediment iron concentration.

Conclusions.

Sites where macroinvertebrate diversities and densities were low were found to have high sediment iron concentrations.

The sites with the highest macroinvertebrate diversities and densities were those where populations of *Dreissena polymorpha* were present.

Sites with background and intermediate sediment iron concentrations had similar community compositions (Group 2) suggesting that a combination of influences were important in shaping the communities at these sites, identification of these factors was difficult because of the limited environmental data available and the absence of obvious patterns from the data.

The separation of site HI3b from Sub-groups A and B was probably an anomaly due to the group averaging cluster method used rather than any great differences from the other sites in Group 2.

From both the cluster groups and the MDS plot it is apparent that there is a gradient relating community composition to sediment iron concentration. As sediment iron concentration increases macroinvertebrate diversities and densities decrease. However at the lower/intermediate sediment iron concentrations the relationship is not so strong, suggesting that other environmental factors combine to shape the community structure.

3.1.4.2. June 1994.

The dendrogram of the cluster analysis for this survey is shown in Figure 12. Four cluster groups were present at the 70% similarity level, the main characteristics of these groups are described in Table 2.

The clusters produced for the June 1994 data are not so well defined as in the March 94 survey, as shown by the higher level of similarity. The most distinct group (Group 1) contained sites where *Dreissena polymorpha* were present resulting in high macroinvertebrate diversities and densities. The average sediment iron concentration at these sites was low (65.5 mg Fe/g).

Group 2 contained North Buoy sites, some Lodge Farm transect sites and a Howell's Inlet site where macroinvertebrate diversity and abundance was moderately high (5 to 10 taxa/m^2 and 1848 to 4444 individuals/m^2).

Figure 12:
Cluster analysis dendrogram of data from Rutland Grid survey carried out on 1/6/94

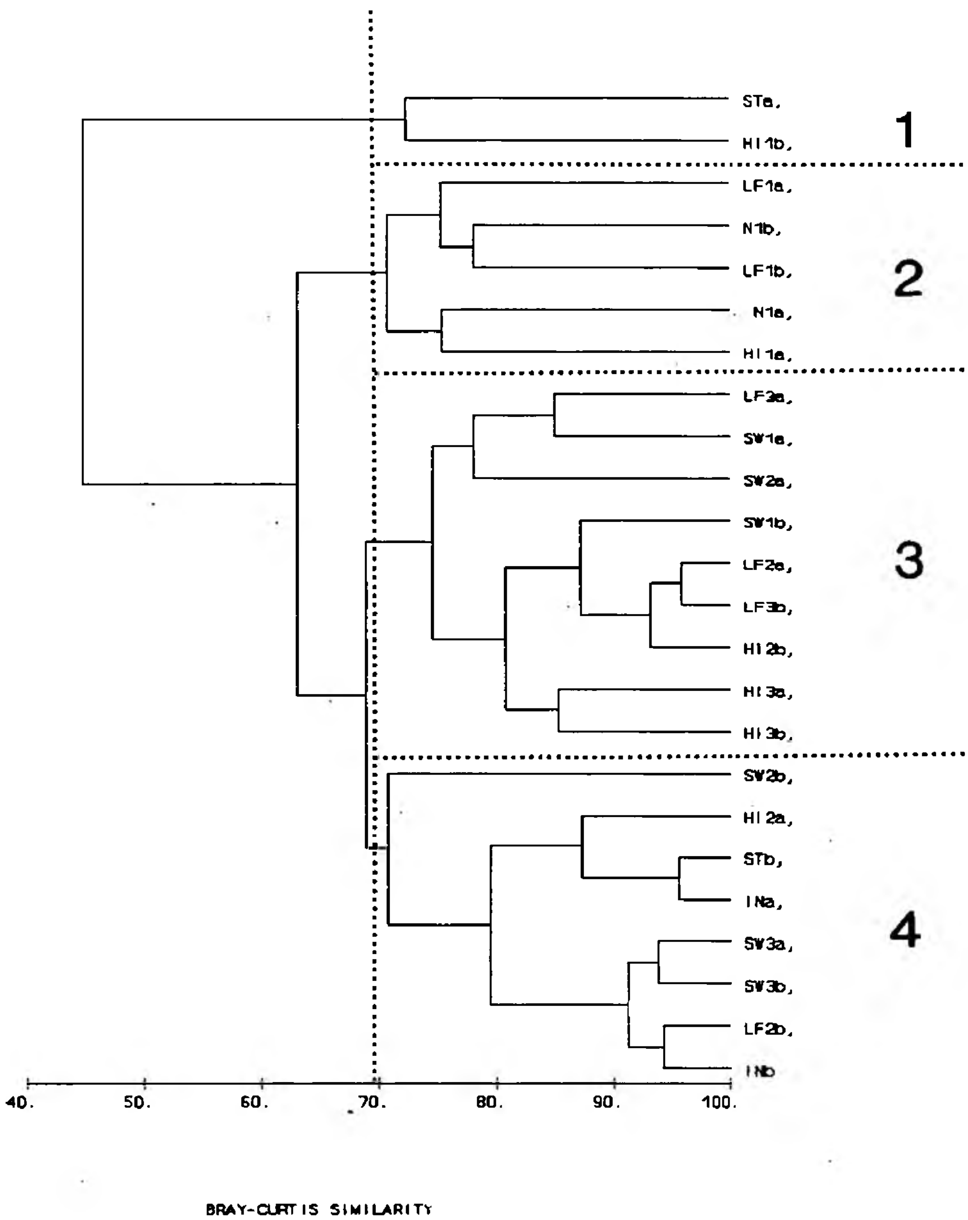


Table 2: Cluster analysis groupings for Rutland Grid Survey 1/6/94

GROUP	SITES	AVERAGE NUMBER OF TAXA	AVERAGE NUMBER OF INDIVIDUALS	TYPICAL TAXA (dominant in bold)	AV. SEDIMENT IRON CONC. (MG/G)
Group 1	STa, HI1b	10 (9-11)	7106	***Dreissena polymorpha***, Oligochaeta, Chironomidae, *Asellus aquaticus*, *Helobdella stagnalis*, *Crangonyx pseudogracilis*	**65.5**
Group 2	LF1a, LF1b, N1a, N1b, HI1a	6.8 (5-10)	3405 (1848-4444)	**Chironomidae**, Oligochaeta, Nematoda, *Valvata piscinalis*, Ceratopogonidae, nematomorpha, *Ocetis ochracea*, *Asellus aquaticus*, *Helobdella stagnalis*	**70.6** (56-96)
Group 3	LF3a,SW1a, SW1b, SW2a, LF2a, LF3b, HI2b, HI3a, HI3b	5.7 (4-7)	2581 (1084-3960)	**Chironomidae**, Oligochaeta, *Pisidium* sp., Nematoda, *Valvata piscinalis*, Ceratopogonidae, Nematomorpha, Hydracarina	**104** (56-224)
Group 4	STb, HI2a, SW2b, INa, SW3a, SW3b, LF2b, INb	2.8 (2-4)	1980 (924-2948)	**Chironomidae**, Oligochaeta, Nematoda, *Pisidium* sp.	**151** (54.5-263)

Group 3 contained some Lodge Farm, Howell's Inlet and Slipway transect sites where macroinvertebrate diversity and abundance was lower than at the sites in Cluster group 2 (4 to 7 taxa/m^2 and 1084 to 3960 individuals/m^2).

The main difference between Groups 2 and 3 was the sub-dominance of *Pisidium* sp. for sites in Group 3, and the presence of species such as *Ocetis ochracea* and *Asellus aquaticus* at sites in Group 2.

The Group 3 sites had higher average sediment iron concentrations (56 to 224 mg Fe/g) than the Group 2 sites (56 to 96 mg Fe/g), though several of the sites in both groups had only background concentrations of sediment iron.

Group 4 contained the sites which had the lowest macroinvertebrate diversities and abundances, namely some Secondary Tower, Howell's Inlet, Lodge Farm, Slipway and Inlet sites. These sites had the highest sediment iron concentrations (average 151 mg Fe/g), though STb and HI2a had only background concentrations of iron. Generally the fauna of the sites in the group was characteristic of sites contaminated by ferric floc, consisting only of chironomids and oligochaetes, with nematodes and *Pisidium* sp. present at some of the sites.

The MDS plot is shown in Figure 13a. Figure 13b shows the same MDS plot, but with superimposed circles whose size is proportional to the sediment iron concentration at each site. High similarities between groups resulted in the groups being positioned quite closely on the MDS ordination plot. There was no obvious gradient relating to sediment iron concentration apparent on this plot.

Conclusions.

Differences between the sites in this survey were low, though there was some indication that sites which had the lowest densities and diversities also had elevated sediment iron concentrations. However the gradient corresponding to sediment iron concentration as seen for the March 94 survey was not particularly evident.

The presence of Stb and HI2a in Group 4 was due to their low species diversities and abundances which cannot be adequately explained, but may have been due to natural variation.

3.1.4.3. March 1995.

The dendrogram of the cluster analysis for this survey is shown in Figure 14. Six groups were identified at the 65% similarity level, the main characteristics of these groups are described in Table 3.

Figure 13a:
Ordination plot (MDS) of data from Rutland Grid survey of 1/6/94 (stress = 0.14)

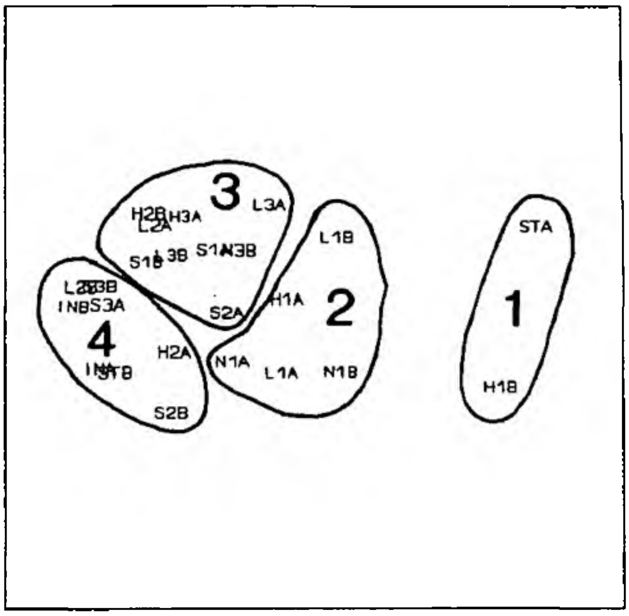


Figure 13b:
Ordination plot (MDS) of data from Rutland Grid survey of 1/6/94 with sediment iron data superimposed as circles (stress = 0.14)

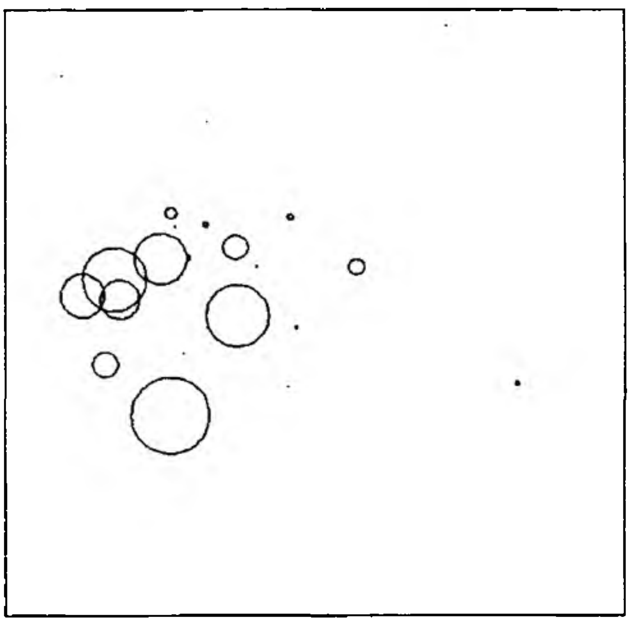


Figure 14:
Cluster analysis dendrogram of data from Rutland Grid survey carried out on 22/3/95

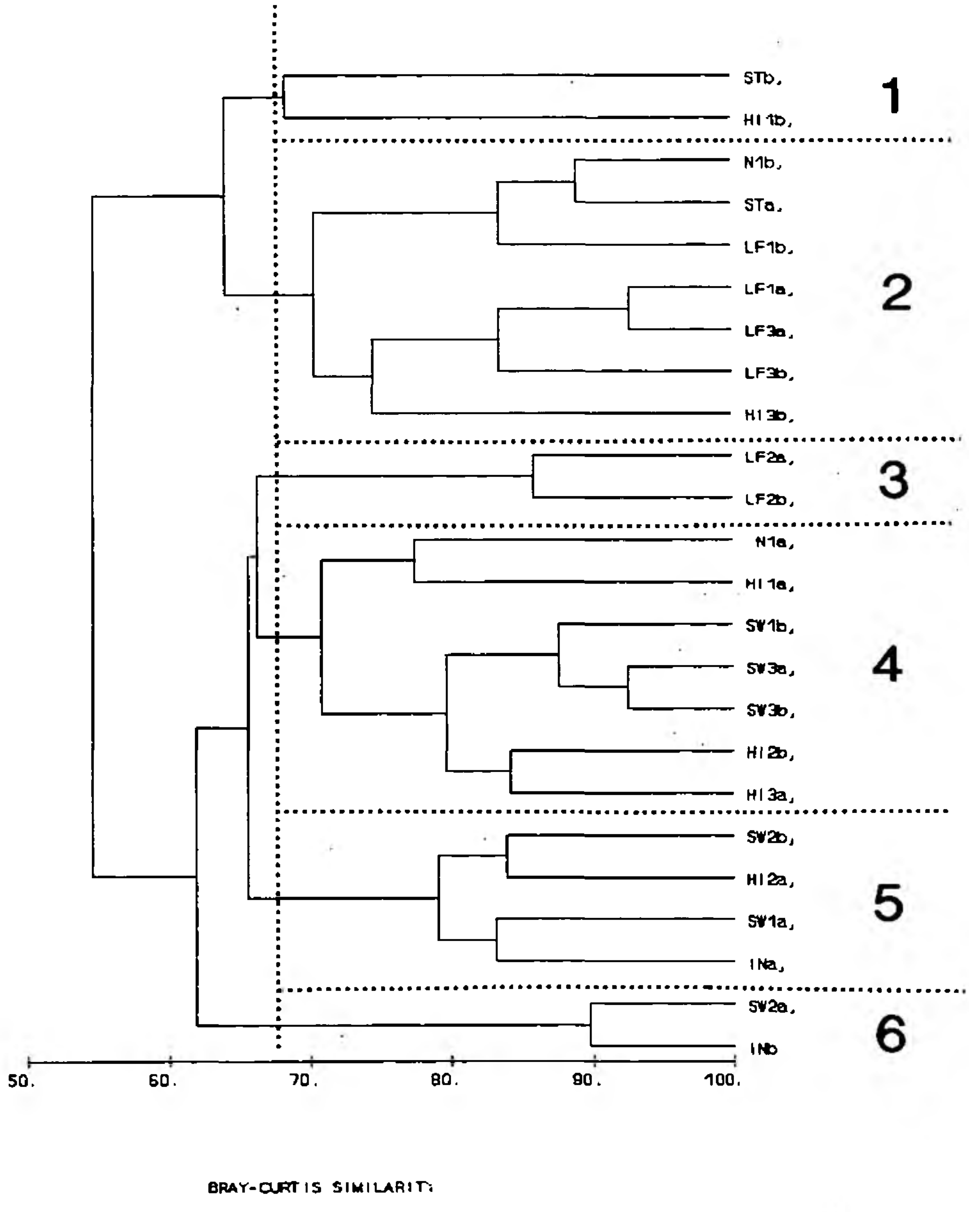


Table 3: Cluster analysis groupings for Rutland Grid Survey 22/3/95

GROUP	SITES	AVERAGE NUMBER OF TAXA	AVERAGE NUMBER OF INDIVIDUALS	TYPICAL TAXA (dominant in bold)	AV. SEDIMENT IRON CONC. (MG/G)
Group 1	STb, HI1b	7.5 (6-9)	10340 (7392-13288)	**Oligochaeta/chironomidae**, Nematoda, *Ocetis ochracea*, *Pisidium* sp.	**63.85** (52.9-74.8)
Group 2	N1b, STa, LF1a, LF1b, LF3a, LF3b, HI3b	10.2 (6-15)	12395 (8228-19448)	**Chironomidae/Oligochaeta**, *Dreissena*, Nematoda, Ceratopogonidae, *Caenis horaria*, *Potamopyrgus jenkinsii*	**62.8** (50.7-86.6)
Group 3	LF2a, LF2b	8 (7-9)	11442 (9992-12892)	**Chironomidae**, Oligochaeta, *Pisidium* sp.	**86.05** (79-93.1)
Group 4	N1a, HI1a, HI2b, HI3a, SW1b, SW3a, SW3b	5.7 (4-7)	4488 (2464-9812)	**Chironomidae/Oligochaeta**, *Pisidium* sp., Ceratopogonidae, *Valvata piscinalis*	**118** (54-192)
Group 5	SW1a, SW2b, HI2a, INa	4.5 (4-5)	2860 (1672-3916)	**Oligochaeta**, chironomidae, *Pisidium* sp.	**223** (91.2-367)
Group 6	SW2a, INb	3.5 (3-4)	1628 (1584-1672)	**Oligochaeta**, chironomidae, *Potamopyrgus jenkinsii*	**307** (223-391)

Groups 1 and 2 contained Secondary Tower, North Buoy, Lodge Farm and Howell's Inlet sites where macroinvertebrate diversity and abundance was high (6 to 15 taxa/m^2 and 7392 to 19448 individuals/m^2). The two groups separated due to differences in their sub-dominant species assemblages, in particular *Dreissena polymorpha* was present at the sites in Group 2.

All the sites in Groups 1 and 2 had low/background sediment iron concentrations (50.7 to 86.6 mg Fe/g). The average depth of the sites in these groups was 11.6 metres.

Group 3 contained two sites (LF2 replicates) both of which had diverse and abundant macroinvertebrate communities (7 to 9 taxa/m^2 and 9992 to 12892 individuals/m^2). These sites appear to have separated from groups 1 and 2 due to the sub-dominance of *Pisidium* sp., in other respects (numbers of taxa and individuals) these sites are more similar to those in Groups 1 and 2 than those in Groups 4, 5 and 6.

The average sediment iron concentration for the two sites in Group 3 was low (86.05 mg Fe/g). The average depth of the sites was 20 metres.

Group 4 contained North Buoy, Howell's Inlet and Slipway sites where macroinvertebrate diversities and abundances were intermediate (4 to 7 taxa/m^2 and 2464 to 9812 individuals/m^2). Sediment iron concentrations at these sites ranged from background levels (54 mg Fe/g) to concentrations of 194 mg Fe/g. The average depth of the sites in this group was 14 metres.

Group 5 contained Slipway, Inlet and Howell's Inlet transect sites where diversity and abundance of macroinvertebrates was low (4 to 5 taxa/m^2 and 1672 to 3916 individuals/m^2). Sediment iron concentrations at these sites were generally high (91.2 to 367 mg Fe/g). The average depth of the sites in this group was 19.25 metres

Group 6 contained a Slipway transect and an Inlet site where macroinvertebrate diversities and abundances were at the lowest levels recorded in this survey (3 to 4 taxa/m^2 and 1584 to 1672 individuals/m^2), these sites were the worst affected by ferric floc with sediment iron concentrations of 223 to 391 mg Fe/g. The average depth of these sites was 21.5 metres.

The MDS plot is shown in Figure 15a. Figure 15b shows the same MDS plot, but with superimposed circles whose size is proportional to the sediment iron concentration at each site.

The ordination plot (MDS) plot supported the cluster groupings of the sites, though there is an overlap between groups 1 and 3 suggesting that the sites are relatively more similar than the cluster analysis would suggest. A gradient related to sediment iron concentration is also apparent.

Figure 15a:
Ordination plot (MDS) of data from Rutland Grid survey of 22/3/95 (stress = 0.14)

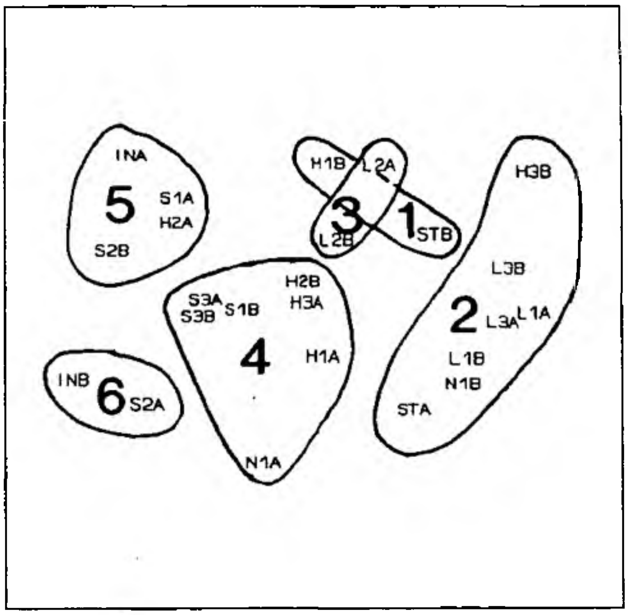
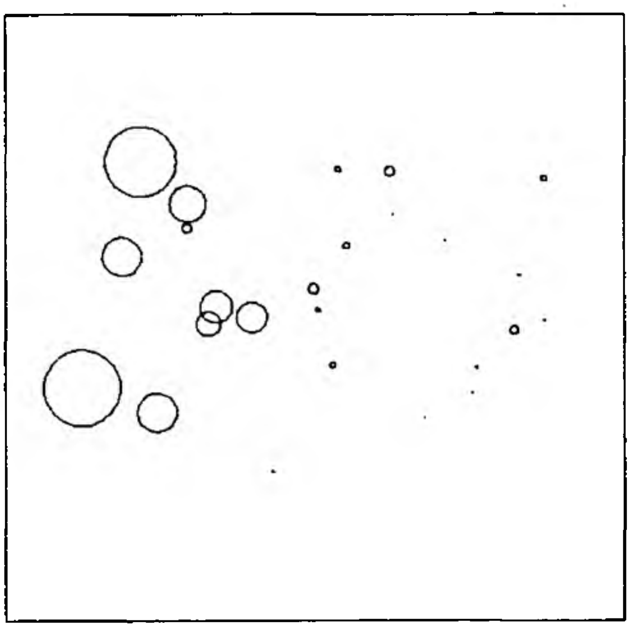


Figure 15b:
Ordination plot (MDS) of data from Rutland Grid survey of 22/3/95 with sediment iron data superimposed as circles (stress = 0.14)



Conclusions.

It was apparent that ferric contamination was an important factor influencing the community composition of sites in this survey. A gradient relating sediment iron concentration and macroinvertebrate diversity and density has emerged from both the MDS and cluster analyses, with densities and diversities declining as sediment iron concentrations increase. It was also interesting to note that the sites most affected by the ferric floc had oligochaetes as the dominant taxa whereas chironomids were dominant at the moderately affected/unaffected sites. This is a classic response of macroinvertebrate communities under toxic stress.

At a lower similarity level than that considered when defining the cluster groups, it appears that the first major split, occurring at around 55% similarity, separates the shallower, unaffected sites (Groups 1 and 2) from sites affected by ferric floc and deeper sites (Groups 3, 4, 5 and 6). Within this group of deep/floc affected sites, the next divisions separate sites with poor fauna, and these are generally characterised by high ferric contamination (Groups 5 and 6). Groups 3 and 4 generally contain deep sites and the moderately floc affected sites. Both ferric contamination and depth probably influenced the community compositions at the sites in this survey. However, for the ferric affected/deep sites the fact that the first division separated sites where ferric contamination was high indicates that ferric contamination may have been the more major influence where contamination occurred.

The fact that sites were well scattered on the ordination plot with relation to depth, also suggests that sediment iron concentration was the more important influence on community structure.

The statistical importance of some of the different environmental variables influencing the community composition of the sites will be investigated and reported elsewhere.

The overlap of Groups 1 and 3 on the ordination plot is probably due to the fact that the plot is two-dimensional, yet the MDS analysis considers more than two dimensions. Groups 1 and 3 are probably separated in the third-dimension in the MDS analysis, but this cannot be seen on the ordination plot. The separation in the cluster analysis is probably more enhanced because of the group averaging cluster method used.

3.1.5 Effects of the presence of *Dreissena polymorpha* clumps on community composition.

The effects of the presence of *Dreissena polymorpha* clumps on community composition are shown in Table 4.

Site replicates where *Dreissena* was present had increased diversities and abundances of macroinvertebrates compared to those where *Dreissena* did not occur.

In particular the numbers of *Asellus aquaticus* (detritivore) , *Crangonyx pseudogracilis* (detritivore), *Helobdella stagnalis* (predator), *Mystacides longicornis* (omnivore) and Oligochaeta (detritivore) were higher.

Table 4. The effect of the presence of clumps of *Dreissena polymorpha* on community composition.

	***DREISSENA* PRESENT**		***DREISSENA* ABSENT**	
SAMPLE SITE	**Replicate LF3a 25/11/94**	**Replicate HI3b 22/3/95**	**Replicate LF3a 25/11/94**	**Replicate HI3b 22/3/95**
Asellus aquaticus	1320	264	44	
Bythinia leachii	44			
Caenis horaria	176	44	132	
Certopogonidae		176	44	220
Chironomidae	2508	5808	3872	1804
Crangonyx pseudogracilis	3388	528	132	
Dreissena polymorpha	14124	2684		
Dugesia polychroa	176			
Helobdella stagnalis	220	88		
Hydracarina		44		
Mystacides longicornis	396	88		
Nematoda	44		44	
Ocetis ochracea	44	176	88	44
Oligochaeta	6732	8932	5500	1628
Pisidium sp.		176	44	88
Polycelis tenuis	396			
Potamopyrgus jenkinsii	44	88		
Valvata piscinalis		352	44	132
NUMBER OF TAXA	**14**	**14**	**10**	**6**
NUMBER OF INDIVIDUALS	**29612**	**19448**	**9944**	**3916**

Conclusions.

The differences were due to the greater habitat diversity provided by the mussel clumps which provide increased shelter and food. Sediment and detritus feeders were probably encouraged by the increased organic content of the substratum. The increased densities of such animals in turn would have lead to an increase in predator species.

3.1.6 Rutland Water grid surveys - overall conclusions

Ferric contamination remained very high at the Inlet and at the sites on the Slipway transect. There has been a decline in sediment iron concentrations during the 1994/5 period probably as a result of lower dosing rates during this period.

At the Howell's Inlet and Lodge Farm transects however there was a small increase in sediment iron concentrations during the 1994/95 period, indicating that the ferric floc may be spreading.

Low macroinvertebrate diversities and abundances were found at sites where sediment iron concentrations were enhanced. The mechanisms causing these reduced communities are still unclear. As concluded in previous reports (e.g. Christmas et al., 1994) it is not evident whether it is the toxic action of the ferric sulphate alone which is exerting a major effect on the invertebrates. It is now thought that a combination of factors associated with high sediment iron concentrations such as particle size effects, blanketing, contamination in the ferric and unique sediment chemistry may be acting alone or synergistically to produce the effects seen.

The macroinvertebrate communities at the sites which had high sediment iron concentrations were not only impoverished, but also possessed different species assemblages compared to the sites where sediment iron concentrations were at background levels, as reported in previous surveys.

It is important to recognise that ferric contamination was not the only factor acting to influence the structure of the communities at sites in the reservoir. In this report water depth was also identified as a major factor, but components of the ecosystem such as sediment chemistry and characteristics, water chemistry, and food availability would also combine to shape community composition.

Whilst contamination remains relatively high, there are indications that the communities had recovered at the contaminated sites, particularly for the survey carried out on 21/6/95. The recoveries may be due to the low levels of ferric dosing during 1994/95. At the Inlet sites a decline in sediment iron concentrations was apparent and it is thought that the ferric floc has been overlain with natural sediment providing an improved habitat for the macroinvertebrates. Unfortunately no information can be derived from measurement of the organic content of the sediments since a core of sediment 5 cm deep including old floc is tested; the old floc would 'dilute' a surface layer of organic matter.

Alternatively certain taxa in the communities may be developing tolerance and adapting to the ferric contamination.

Another possibility is that the apparent recoveries in numbers of individuals and chironomids were simply due to delayed temporal population fluctuations caused by sub-lethal effects of the ferric floc on the communities. The chironomids, which are a dominant component of the benthic communities particularly appear to have been affected in this way. The increases may not have actually been real, but were a result of the presence of larger, later instar larvae (see Section 3.1.2).

Conclusions about the causes of the possible recoveries are difficult to draw, but it is likely that a combination of the factors mentioned above have operated and the matter is further complicated by natural annual variations in communities.

3.2 RUTLAND WATER - BENTHOS SURVEYS.

Historical data from the Rutland Benthos surveys will be considered here with relation to sediment iron concentrations and ferric dosing regimes.

Data from the sites on the N2 and S3 transects only will be assessed in this summary report (see Figure 1 for the location of the transects).

3.2.1 Ferric sulphate dosing and sediment chemistry.

The monthly input of ferric sulphate to Rutland Water is plotted with the average sediment iron concentration of the S3 transect on Figure 16. Inputs of ferric sulphate to the reservoir have been reduced considerably in recent years (see Section 3.1.1), but the S3 transect sediment iron concentrations have generally remained at the levels attained during the intensive dosing of 1991/92 and have not declined despite the reduced dosing. The concentrations remain at levels greater than 96 mg Fe/g above which serious damage to chironomid communities has been identified (Brierley, 1992).

A comparison of the background sediment iron concentration at site 3 of the N2 transect with the average concentration recorded for the S3 transect is shown on Figure 17. The "background" iron level at N2 site 3 was around 60 mg Fe/g, whilst average concentrations for the S3 transect have fluctuated around 120 mg Fe/g, generally remaining greater than 100 mg Fe/g.

Comparison with Figure 16 shows that peaks in S3 iron concentrations normally corresponded to peaks in ferric sulphate dosing.

FIGURE 16. Monthly ferric sulphate input to Rutland Water and average S3 transect sediment iron concentrations 1990 to 1995

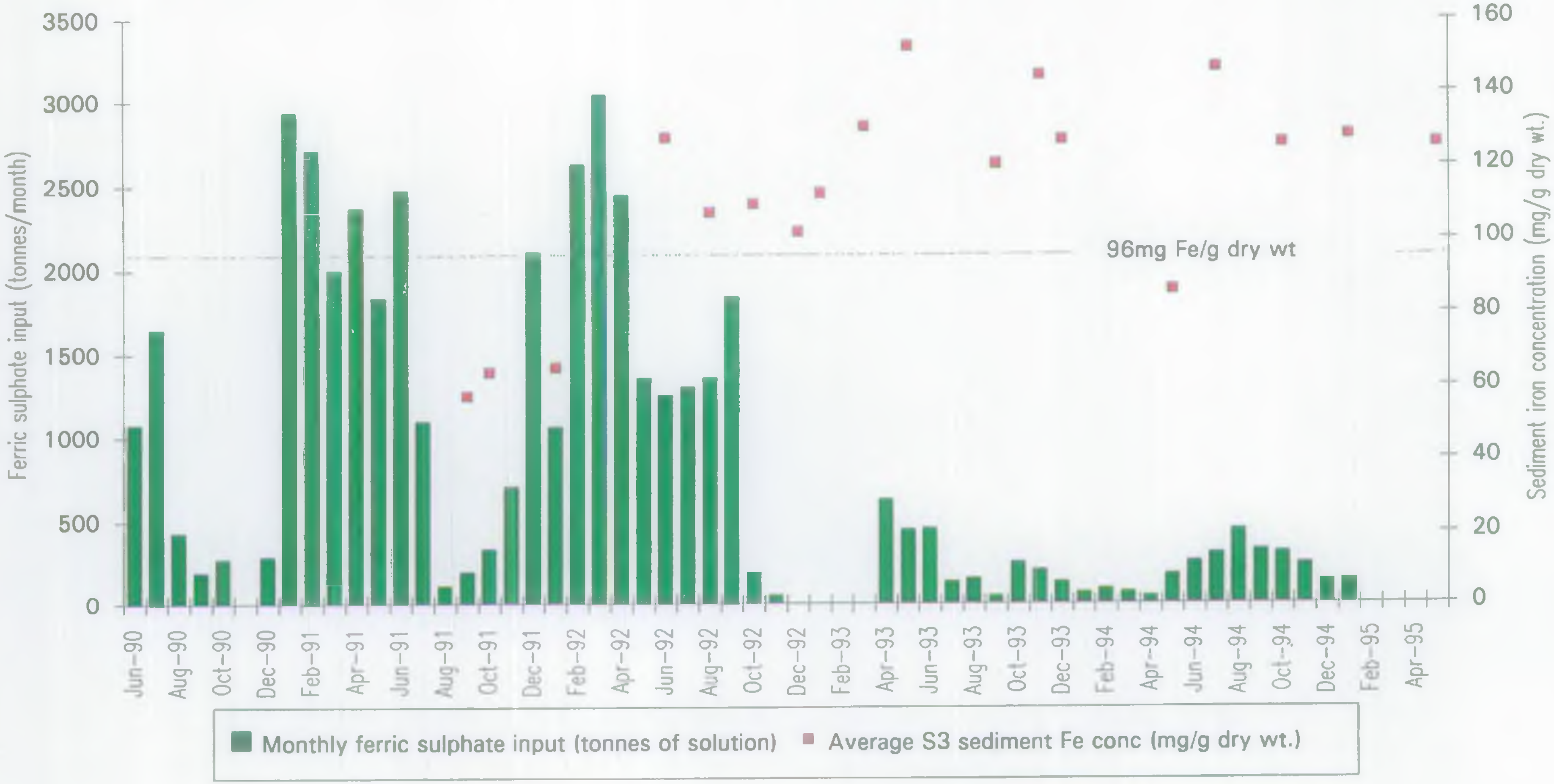
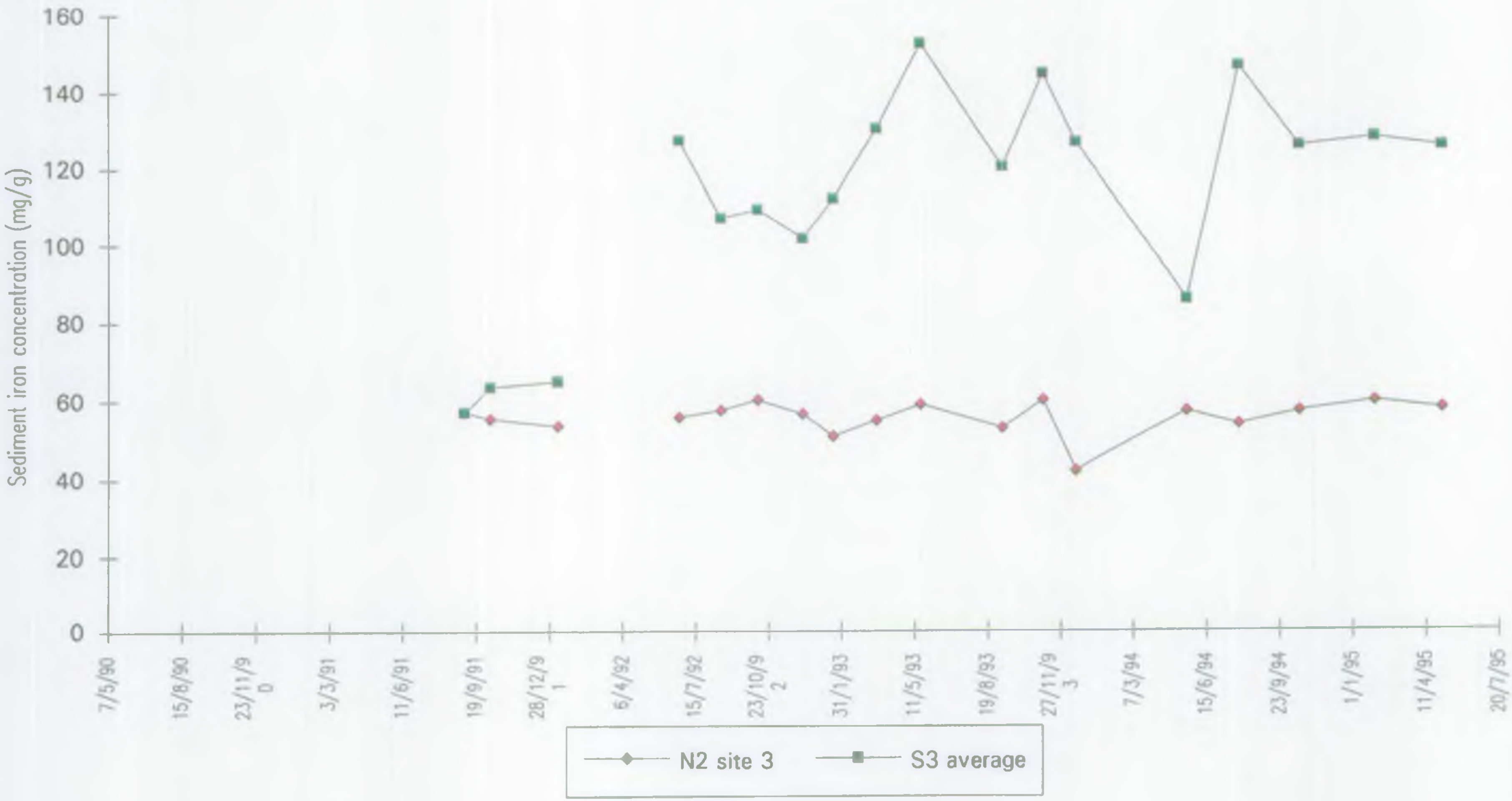


FIGURE 17. Sediment iron concentrations at N2 transect site 3 and sites on the S3 transect 1990 to 1995



3.2.2 Benthic invertebrates.

The average chironomid densities recorded on the N2 and S3 transects are shown in Figure 18. A decline in chironomid numbers occurred on both transects between June 1990 and June 1991, and this coincided with the start of ferric dosing (June 1990). The communities in the north arm may have been affected by the barge dosing during the period between 1/8/90 and 3/10/90, though this is thought to be unlikely.

Following the re-commencement of Inlet dosing in January 1991 chironomid densities on the S3 transect declined and numbers have remained below those recorded for the N2 transect. Peak dosing periods were generally followed by a decline in chironomid numbers recorded on the S3 transect e.g. from March to May 1991, November to December 1992 and September to October 1993 a decline in chironomid densities on the S3 transect occurred following periods of increased dosing.

The possible delayed development and emergence apparent at the Slipway and Inlet sites of the grid surveys were not apparent here, but the sediment iron concentrations on the S3 transect were lower than those at the Inlet and on the Slipway transect. If slight delays in development were occurring they might have been masked by natural variations, and by the declines seen following peak dosing periods.

The average densities of individuals recorded for each transect are shown in Figure 19. As for the chironomid densities a decline occurred between June 1990 and June 1991 on both transects. After January 1991 (the re-commencement of Inlet dosing) chironomid densities declined on the S3 transect, and generally remained at levels below those on the N2 transect.

The peak on 8/12/92 where numbers for the S3 transect were greater than those for the N2 transect was due to the presence of large *Dreissena polymorpha* clumps at an S3 transect site, and corresponds to a period where no dosing occurred.

Again peaks in dosing were followed by a decline in the numbers of individuals recorded, for example a decline in the densities of individuals recorded on the S3 transect occurred from March to May 1991, November to December 1992 and September to October 1993 following periods of increased dosing.

3.2.3 Conclusions.

The moderately high sediment iron concentrations recorded for the S3 transect have been maintained in 1994/95 above 96 mg Fe/g, and there has been no decline in concentrations despite reduced levels of ferric sulphate dosing, however, during periods following resumed or increased dosing, sediment iron concentrations were seen to increase.

FIGURE 18. Average densities of chironomids at N2 and S3 transect sites from 1990 to 1995

FIGURE 19. Average densities of individuals at N2 and S3 transect sites from 1990 to 1995

The benthic communities on the S3 transect had reduced numbers of chironomids and total individuals compared to "control" sites on the N2 transect where sediment iron concentrations were at background levels. There has been no indication of a recovery of the fauna, and following periods of resumed or increased dosing community diversities and densities were seen to decrease.

3.3 RUTLAND WATER - LITTORAL SURVEYS.

Two littoral surveys were carried out at Rutland Water during the period of 1994/95 on 24/8/94 and 8/3/95. A summary of the data obtained is given in Table 5.

Table 5. A summary of the data obtained during the Rutland Littoral surveys of 24/8/94 and 8/3/95.

	Average sediment iron concentration (mg Fe/g)		Average numbers of taxa/m²		Average numbers of individuals/m²	
	24/8/94	8/3/95	24/8/94	8/3/95	24/8/94	8/3/95
Whitwell Creek	273	70.8	11.3	12	189.0	267.7
Sykes Lane	29.2	43.1	10.3	4	176.0	27.0
Normanton Church B	53,5	57.2	5.0	10.3	57.3	173.3
Nature Reserve B	27.1	44.6	9.7	12.3	221.3	179.7
Golden Bay Pumping Station	26.0	33.4	12.3	13.0	84	366.7
Carrot Creek	58.0	53.6	10.0	11.7	117.3	137.3

3.3.1 Sediment Iron Concentrations.

Sediment iron concentrations ranged between 26.0 mg Fe/g and 273 mg Fe/g in 1994 and between 33.4 mg Fe/g and 70.8 mg Fe/g in 1995. The highest levels were found at the Whitwell Creek site in 1995 (273 mg Fe/g dry weight of sediment). The concentrations recorded from the littoral sites were mostly similar to or lower than those recorded for benthic 'control' sites (approx. 50-60 mg Fe/g). Sites in the south arm of the reservoir most likely to be at risk from ferric contamination did not show elevated sediment iron concentrations.

The high levels recorded in Whitwell Creek during 1994 (and in previous years) were due to the local geology, the site is in an area where natural ironstone is exposed at the surface of the substratum. The iron present is bound in the rock and therefore was not available to the fauna at the site.

3.3.2 Biological Data.

The average number of taxa recorded per sample ranged from 5 to 12.3 in 1994 and from 4 to 13 in 1995, the lowest numbers were found in the Normanton Church samples in 1994 and the Sykes Lane samples in 1995, due to difficulty in sampling the stony substratum.
There was little variation in the average numbers of taxa per sample for the other sites in both years.

The average numbers of individuals recorded per site ranged between 84 and 221.3 for 1994 and 27 and 366 for 1995, again the lowest numbers were recorded for the Normanton Church samples in 1994 and the Sykes Lane samples in 1995.

3.3.3 Conclusion.

The chemical and biological data indicated that the littoral sites on Rutland Water were not affected by ferric dosing.

Variations between sites were high for both chemical and biological parameters, but must be attributed to some other factor since none of the sites were contaminated by ferric sulphate floc.

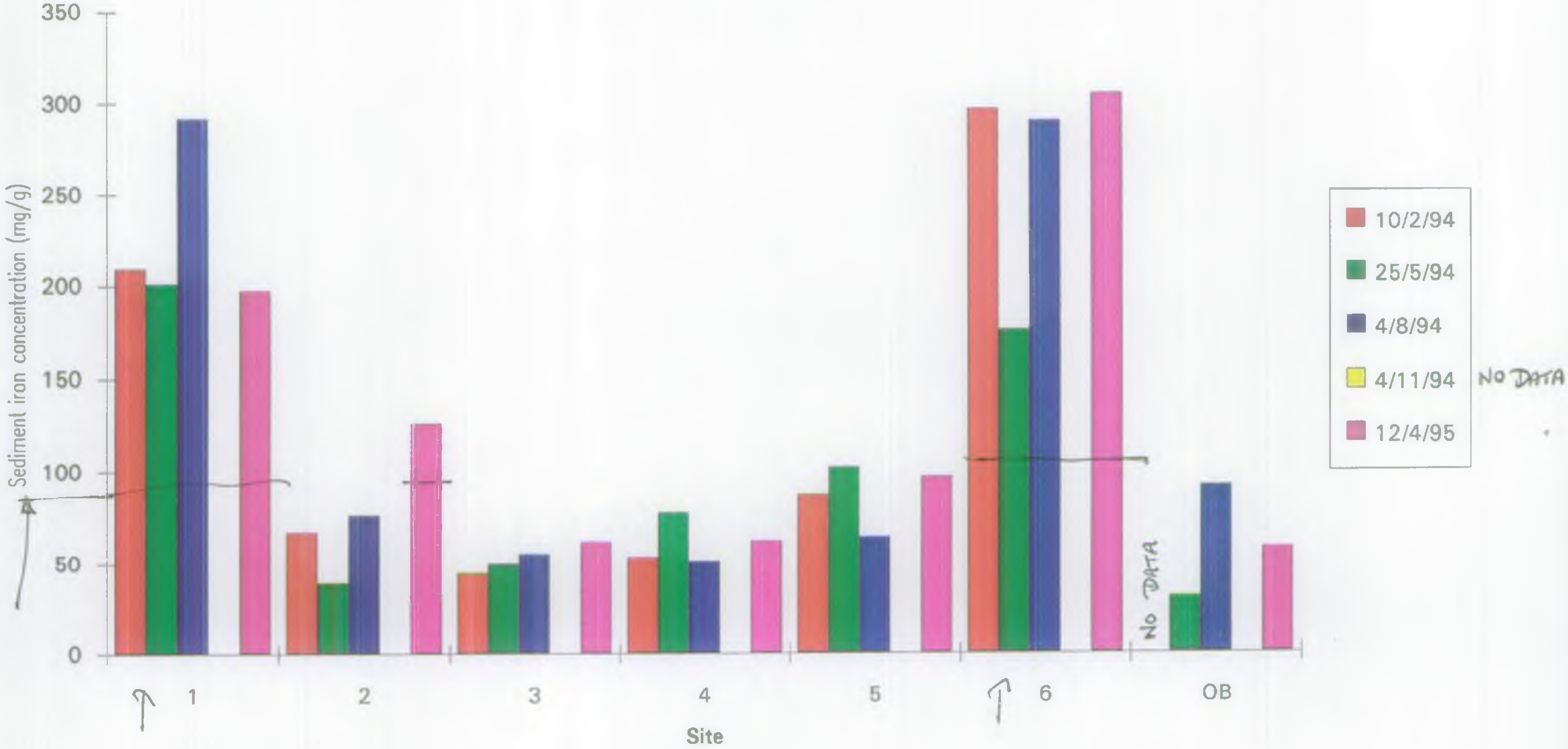
3.4 COVENHAM RESERVOIR - GRID SURVEYS.

3.4.1 Sediment Iron Results.

Sediment iron concentrations are shown in Figure 20. Highest levels were found at sites 1 and 6 which are closest to the Inlet. Sediment iron concentrations declined with distance from the Inlet.

Sites 1 and 6 had sediment iron concentrations between 150 and 350 mg Fe/g, this was well above the level of 96 mg Fe/g found to represent seriously reduced chironomid communities in Rutland Water. During certain surveys sediment iron concentrations approached or exceeded the level of 96 mg Fe/g at sites 2 (12/4/95), 5 (10/2/94, 25/5/94 and 12/4/95) and OB (4/8/94), though there was no indication of an increase in sediment iron with time except possibly at site 2.

FIGURE 20. Sediment iron concentration at sites on Covenham Reservoir during 1994/95



3.4.2 Benthic invertebrates.

Numbers of taxa recorded for the sites on Covenham Reservoir in 1994/95 are shown in Figure 21. There was little variation in the numbers of taxa recorded for the sites, and no indication that numbers were reduced, even at sites where sediment iron levels reached 200 mg Fe/g.

Numbers of individuals recorded for the sites on Covenham Reservoir in 1994/95 are shown in Figure 22 and were variable. For the surveys of 25/5/94 and 4/8/94 numbers increased with distance from the Inlet indicating that the higher sediment iron levels at sites 1 and 6 may have been affecting the biota, but the differences were slight, and the numbers recorded for these surveys in general were low compared to the other surveys. For the other surveys (10/2/94, 4/11/94 and 12/4/95) no clear relationship between macroinvertebrate diversity and density and ferric sulphate contamination could be deduced, although multivariate analyses and increased taxonomic information may elucidate this further.

3.4.3 Conclusions.

Although sediment iron concentrations were high at some of the sites in Covenham Reservoir and reached levels similar to those found to cause damage to benthic communities in Rutland Water, there is only slight evidence that there was an effect on numbers of individuals in Covenham Reservoir in certain years, and no other evidence of damage to communities.

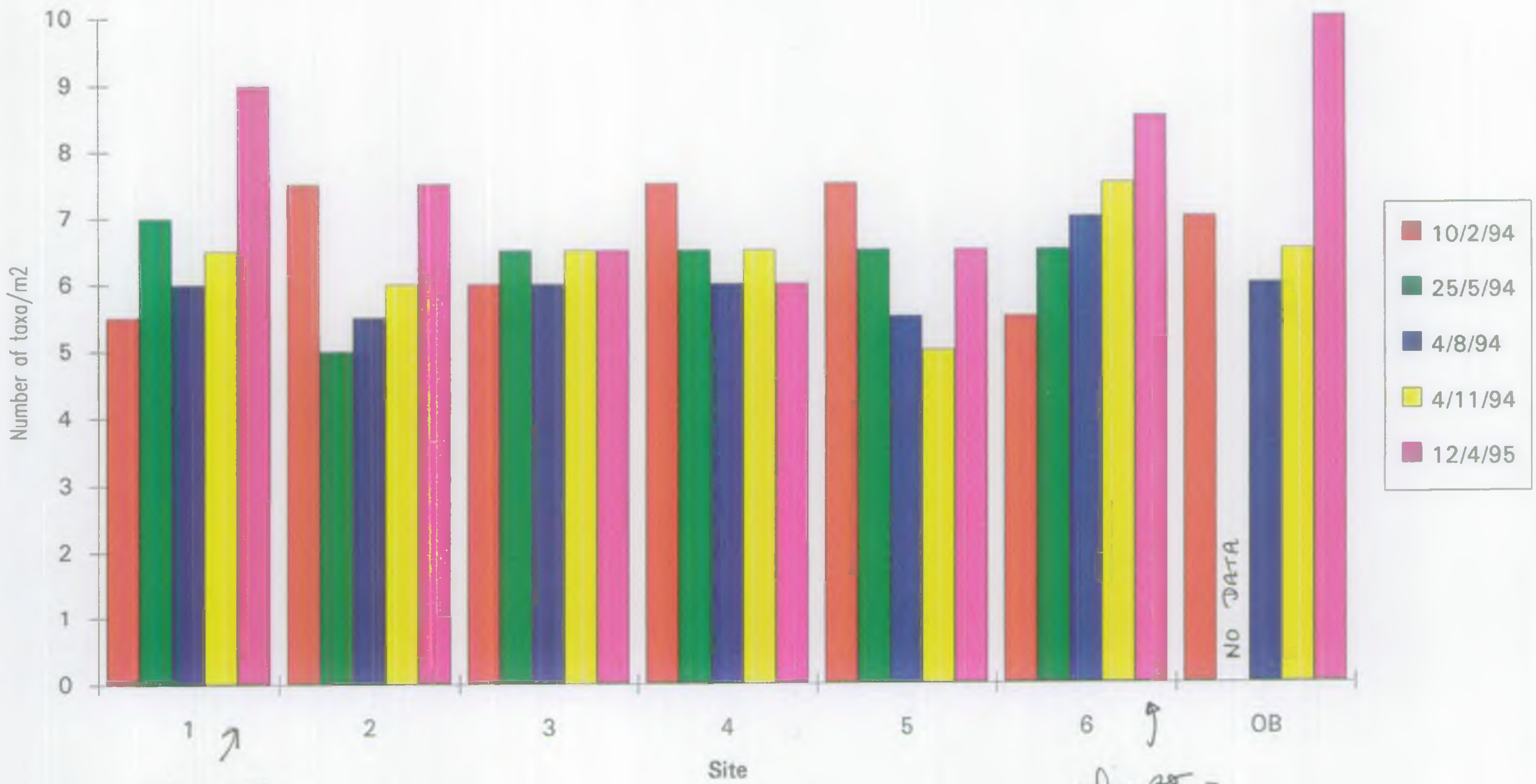
The reasons for this are not clear, it is possible that invertebrates are being introduced to the reservoir with the input canal water, though none of the species recorded stand out as being more typical of the canal than the reservoir except for *Gammarus pulex*.

Alternatively the floc in the sediment may have been overlain with natural sediment since ferric sulphate dosing has not been carried out during 1994 and 1995. The sediment cores taken for iron analysis would have combined the upper most layers with deeper floc contaminated sediment, and therefore the sediment iron concentration measurements would not have reflected a possible recovery in the upper-most layers of the sediment.

3.5 COVENHAM RESERVOIR - BENTHOS SURVEYS.

Data from the Covenham Reservoir benthos surveys carried out on 6/5/94 and 7/2/95 are still to be analysed fully and will be reported elsewhere.

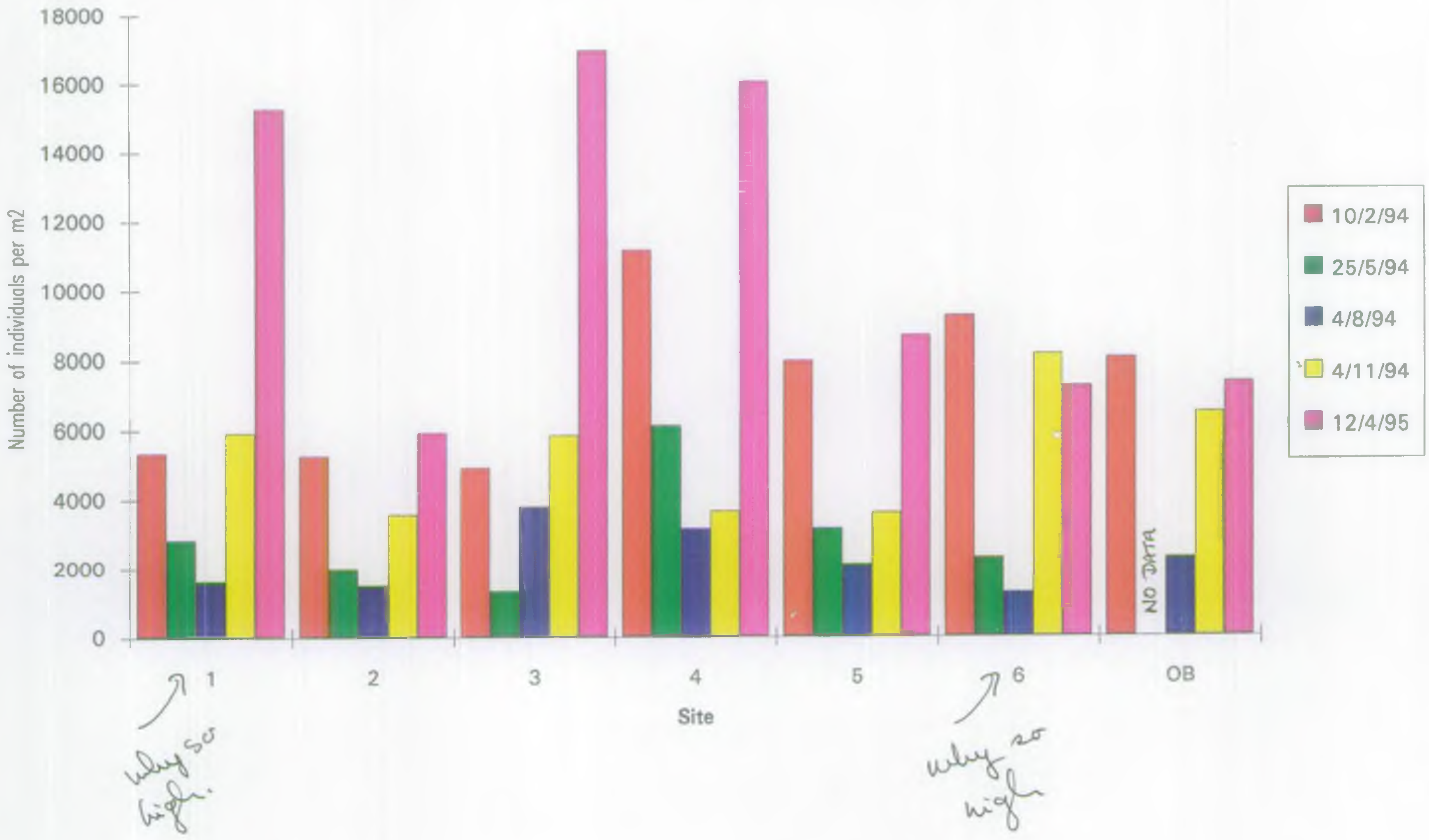
FIGURE 21. Numbers of taxa recorded at sites on Covenham Reservoir during 1994/95



why so high ?

why so ? high.

FIGURE 22. Numbers of individuals recorded at sites on Covenham Reservoir during 1994/95



3.6 PITSFORD RESERVOIR.

Data from the Pitsford Reservoir benthos/grid surveys carried out on 5/1/94, 12/4/94 and 5/7/94 are still to be analysed fully and will be reported elsewhere.

4. RESULTS AND DISCUSSION - PHYTOPLANKTON AND WATER CHEMISTRY.

This summary report includes data on phosphorus, phytoplankton and cyanobacterial populations from Rutland Water and Covenham reservoir for the year ending December 1994. A more detailed limnological report of the reservoirs in the Northern Area will be presented elsewhere.

4.1 RUTLAND WATER.

The results presented here are taken from historical data and the routine weekly sampling programme. The details of the sampling programme remain unchanged and have been reported in previous reports (Extence *et al.* 1992, Christmas *et al.* 1994). The sampling sites are shown in Figure 1.

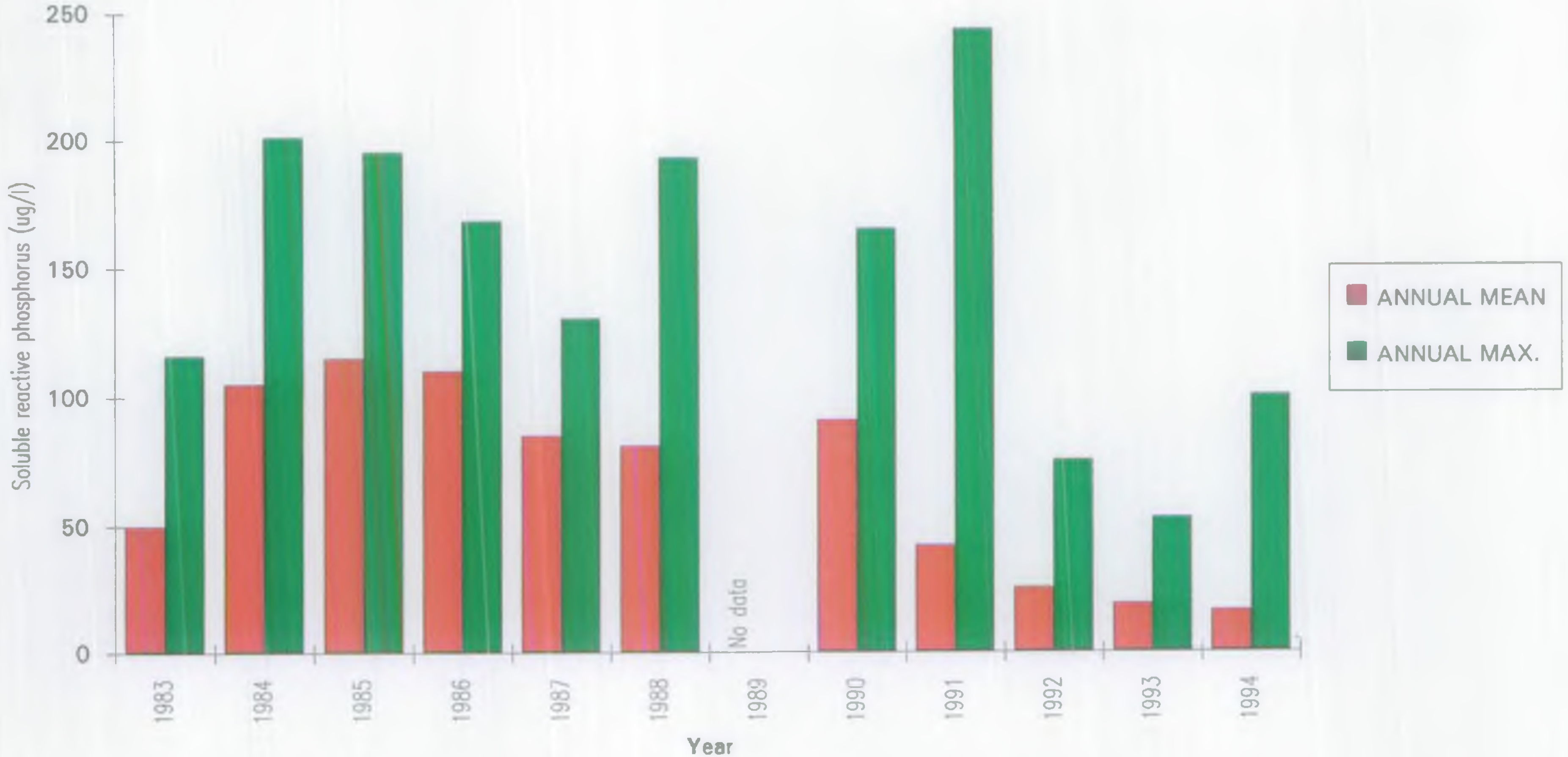
AWS have continued to directly dose Rutland Water with ferric sulphate during 1994. In total 2782 tonnes were added during the year when pumping from the rivers Welland and Nene was being undertaken (Figures 4 and 5).

The total quantity of ferric sulphate dosed into Rutland Water between May 1990 and December 1994 was 44714 tonnes.

4.1.1 Phosphorus.

The annual maxima and mean soluble reactive phosphorus (SRP) concentrations from 1983 to 1994 are shown in Figure 23. Since 1990, the annual mean SRP concentration has declined and this coincides with the period of direct ferric sulphate dosing. The rate of decrease in 1994 has reduced and the mean concentration in 1994 was 16 ug/l compared to 91 ug/l in 1990. The annual maxima have not followed a similar downward trend, as would be expected, and the maxima during 1994 was higher than in the previous two years.

FIGURE 23
Soluble reactive phosphorus annual mean and annual maxima at the Limnological Tower, Rutland Water, between 1983-1994



The fluctuations seen in SRP have resulted from the balance between uptake of P by phytoplankton and macrophytes and input pulses. The origins and magnitude of these inputs cannot always be determined but included river loadings, internal loading from the sediments, breakdown and recycling of macrophytes, planktonic and benthic organisms, inputs from birds and fish and the re-mineralisation after flooding of littoral zones.

The temporal changes in total phosphorus (TP) and soluble reactive phosphate (SRP) at the Limnological Tower (0-5 metres) from May to December 1994 are shown in Figure 24. Total phosphorus remained below 150 ug/l throughout this period with peaks during July, August and November. These peaks were similar in magnitude to those recorded from the Lim. Tower during 1992 and 1993.

Concentrations of SRP remained below 30ug/l throughout the period and fell below the limit of detection during July. This phenomenon occurs every year at Rutland Water and coincides with the main phytoplankton growth period.

The long term changes in total and soluble phosphorus have yet to be analysed in detail. Time series analyses are necessary to confirm whether the lower P levels are statistically significant. Further investigations will then be required to deduce whether any significant changes in concentrations are a result of inter year variability or due to changes in zooplankton and fish populations (top-down control) and/or changes in reservoir management such as ferric dosing, pumping regimes, mixing etc.

4.1.2. Phytoplankton biomass, succession and cyanobacteria.

The results presented here are from the Limnological Tower (0-5 metres) with some date from other routine sites.

The long term changes in the annual mean and maximum chlorophyll a concentrations between 1979 and 1994 are shown in Figure 25. No long term pattern was discernable in either mean or maximum concentrations.

The mean algal biomass during 1994 (4.7ug/l chl.a) was marginally lower than that recorded in 1992 (4.98ug/l) and these two averages are the lowest since monitoring began in 1979. Whether these annual average concentrations are statistically lower and outside than inter year variability has yet to be evaluated.

Annual maxima have varied considerably and the maximum of 26.6 ug/l for the Limnological Tower during 1994 was well within the inter year variations. A maximum of 220 ug/l was recorded during a bloom of cyanobacteria during August 1994 at the S12 sampling buoy and

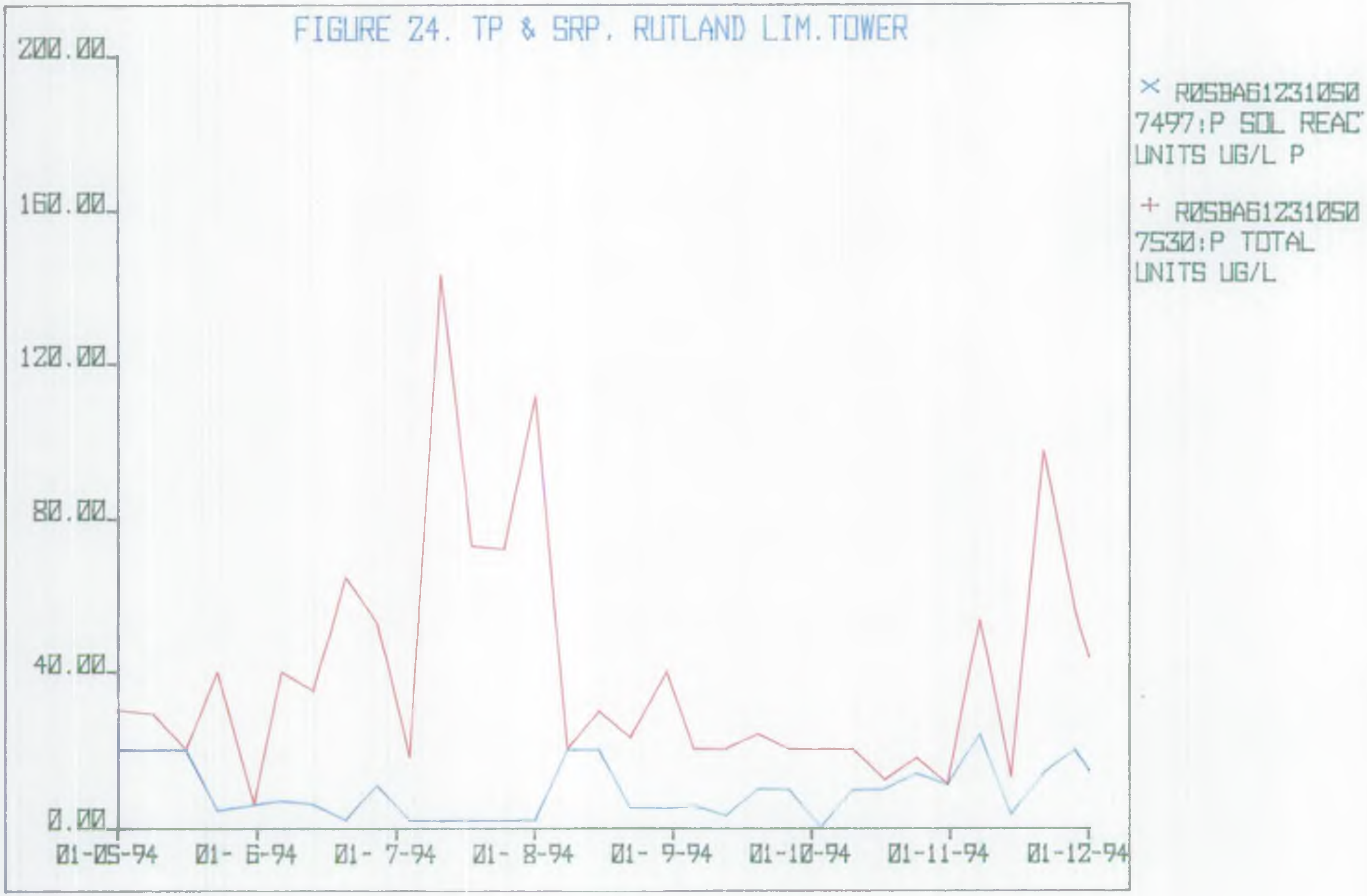


FIGURE 24. TP & SRP. RUTLAND LIM.TOWER
200.00
160.00
120.00
80.00
40.00
0.00
01-05-94
01- 6-94
01- 7-94
01- 8-94
01- 9-94
01-10-94
01-11-94
01-12-94
R05BA61231050
7497:P SOL REAC
UNITS UG/L P
R05BA61231050
7530:P TOTAL
UNITS UG/L

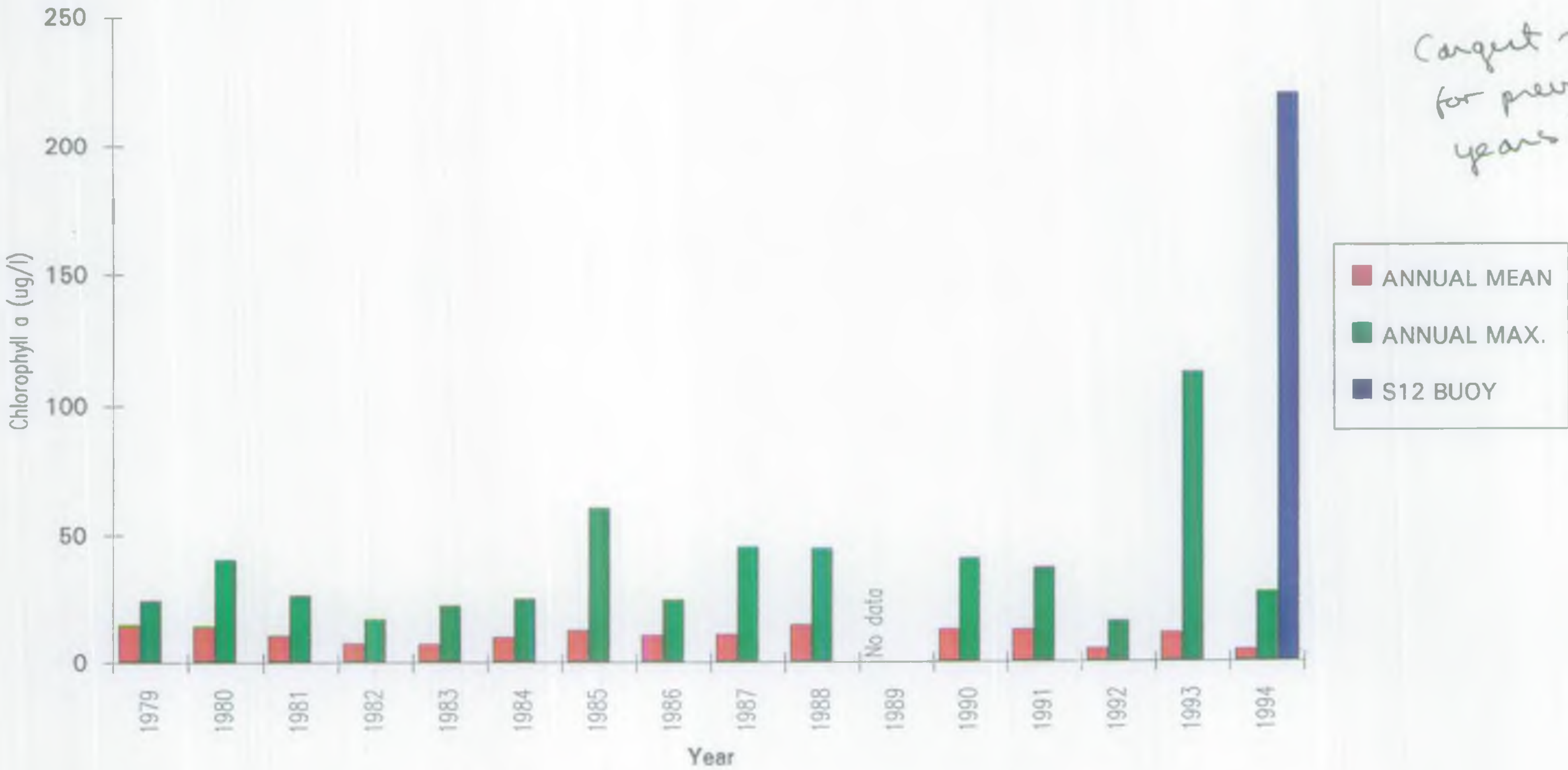
44

FIGURE 25.
Chlorophyll a annual mean and annual maxima at LT between 1979-1994 (included is 1994 maximum at S12)

95?

89 max?

Largest max for previous years?



has been included as this was the highest chlorophyll concentration noted during routine weekly sampling since sampling began in 1979.

The temporal changes in algal biomass at the Limnological Tower during 1994 are shown in Figure 26. The dominant taxa are noted. The spring peak was dominated by *Cryptomonas* spp. with small centric diatoms being sub-dominant. This is slight deviation from the normal spring peak as diatoms usually dominate, often with cryptophytes being sub-dominant. Whether or not this change in the successional pattern is important cannot yet be established as inter year variations in algal succession can be large so further data is required.

It is most probable that part of the reason for this change is due to alterations in the pumping regimes into the reservoir. Historically, the reservoir was "topped up" during the winter months when water in the Nene and Welland was abundant. In recent years, coinciding with period of direct ferric dosing, these patterns of pumping have been different and during the early months of 1994 smaller quantities of river water were pumped into the reservoir. This may have affected the spring peak in two ways, firstly the seeding of diatoms into the reservoir from the rivers would have been lower and secondly, nutrient loads from the rivers (especially silica) would have been lower.

In July colonial green algae and the cyanobacteria, *Aphanizomenon flos-aquae* and *Anabaena* spp. became dominant reaching a chlorophyll peak of over 10 ug/l. This community then declined and the largest peak of the year, reaching over 26ug/l at the Limnological Tower and over 220ug/l at the S12 buoy was recorded in late August. This major peak was dominated by *Aphanizomenon flos-aquae*, *Anabaena* spp. and *Microcystis aeruginosa* when densities reached over 300,000 cells/ml at the Limnological Tower. This bloom then declined as strong winds, rain and colder weather occurred in September. Figure 27 shows the temporal changes in densities of all cyanobacterial taxa from May 1990 to November 1994. There has not been a decline in the cyanobacterial densities since ferric dosing began in 1990.

4.2 COVENHAM RESERVOIR.

The results presented here are from historical data from 1982 to 1989 (samples were taken at the centre draw-off tap at the treatment works) and routine fortnightly samples taken at the Valve Tower (0-5metres) from 1990 to the end of December 1994.

Ferric sulphate dosing into the reservoir with inlet water from the Louth Canal was initiated in the summer of 1990. Tertiary treatment at Louth Sewage treatment works with ferric sulphate also commenced during 1990. Data on the quantities and timing of ferric dosing into Covenham Reservoir during 1994 were not available at the time of writing this report.

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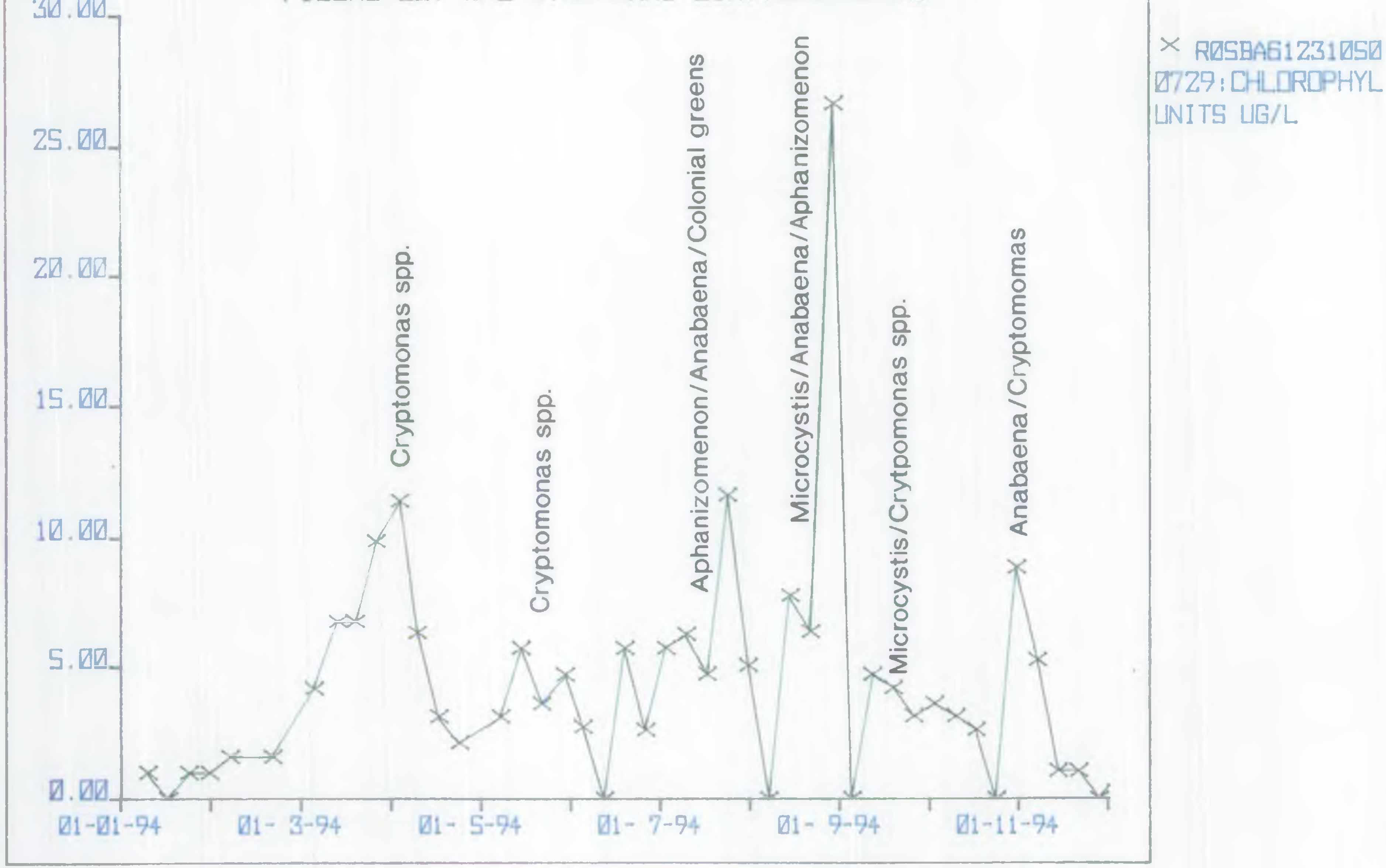


FIGURE 26. CHL a.RUTLAND LIM.TOWER(0-5m)

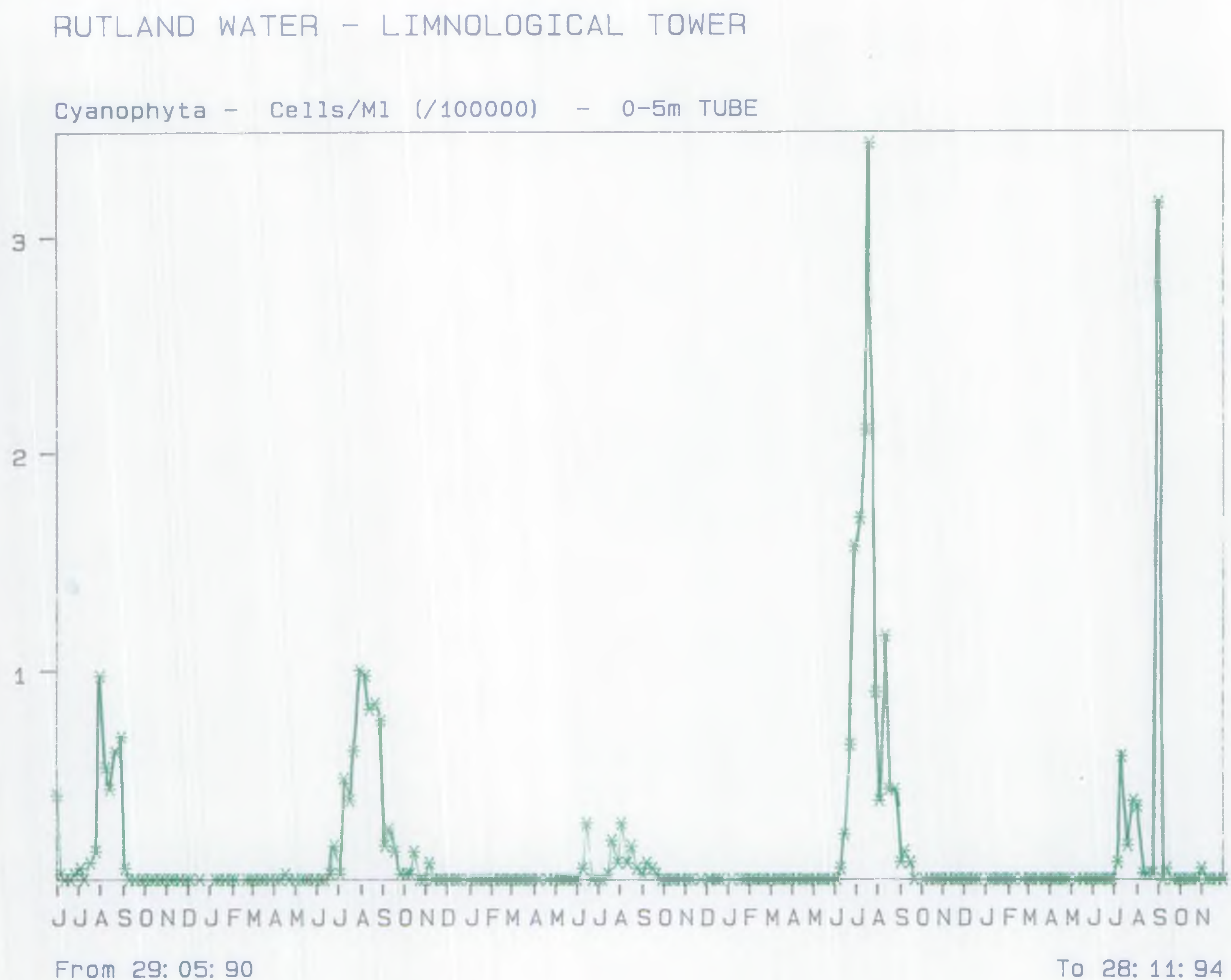


FIGURE 27. TEMPORAL CHANGES IN CYANOBACTERIAL DENSITIES, RUTLAND WATER LIM. TOWER (0-5m), 1990-1994.

4.2.1 Phosphorus.

The annual mean and maximum ortho-phoshate concentrations from 1982 to 1994 are shown in Figure 28. Both mean and maxima have declined greatly since dosing began in 1990. Direct ferric dosing stopped in October 1994 and during the earlier part of the year it would appear that P control by dosing was not as efficient as in 1991, 1992 and 1993. Mean and maximum ortho-phosphate levels were slightly higher as a result.

Temporal changes in TP and SRP during 1994 are shown in Figure 29. Levels were generally low throughout the year but there was a very large peak of TP, over 1.8 mg/l, in late February and two smaller peaks in June (just under 400ug/l) and September (less than 200ug/l). SRP remained below 100ug/l throughout the year with a maximum concentration of 92ug/l recorded in early June (coinciding with the TP peak). Unlike 1992, when SRP remained below 10ug/l (AWS's "target" level) for most of the year, SRP values fluctuated with peaks in February, June and July. This pattern of fluctuations, although at lower concentrations, were more typical of the changes in the reservoir before dosing started in 1990.

4.2.2 Phytoplankton biomass, succession and cyanobacteria.

The long term changes in annual maxima and mean chlorophyll a levels are shown in Figure 30. Data between 1982 and 1989 are from the Centre Draw Off (CDO) tap situated in the water treatment works whilst from May 1990 to December 1994 were from the Valve Tower (0-5metres).

Annual mean values during the period 1990 to 1993 were the lowest recorded throughout this sampling period. Annual maxima in 1990, 1992 and 1993 were also the lowest recorded. The period of low algal biomass during the early 1990's coincides with the lower TP and SRP levels. The successful reduction of phosphorus in the water column of the reservoir appeared also to have led to a slight reduction in algal biomass. This is not conclusive and requires further time series analysis and incorporation of other biological and limnological data.

For example, the zooplankton communities, which can have a very significant effect (either bottom-up and/or top-down) on the phytoplankton biomass and succession were very different during 1991, 1992 and 1993. The large *D. magna* became dominant in May 1991 and July 1993, the intermediately sized *D. pulex* becoming dominant throughout most of 1993 whilst the smaller *D. galeata* was dominant throughout 1992.

The annual mean and maximum algal biomass levels in 1994 were higher than the four previous years and the maximum value of 192ug/l was the highest recorded since 1982.

50

FIGURE 28. ANNUAL MEAN AND MAXIMUM ORTHO-PHOSPHATE LEVELS AT COVENHAM RESERVOIR, 1982 TO 1994.

DATA FOR 1982-1990 FROM CENTRE DRAW OFF SAMPLES AND FROM 1991-1994 FROM VALVE TOWER (0-5m).

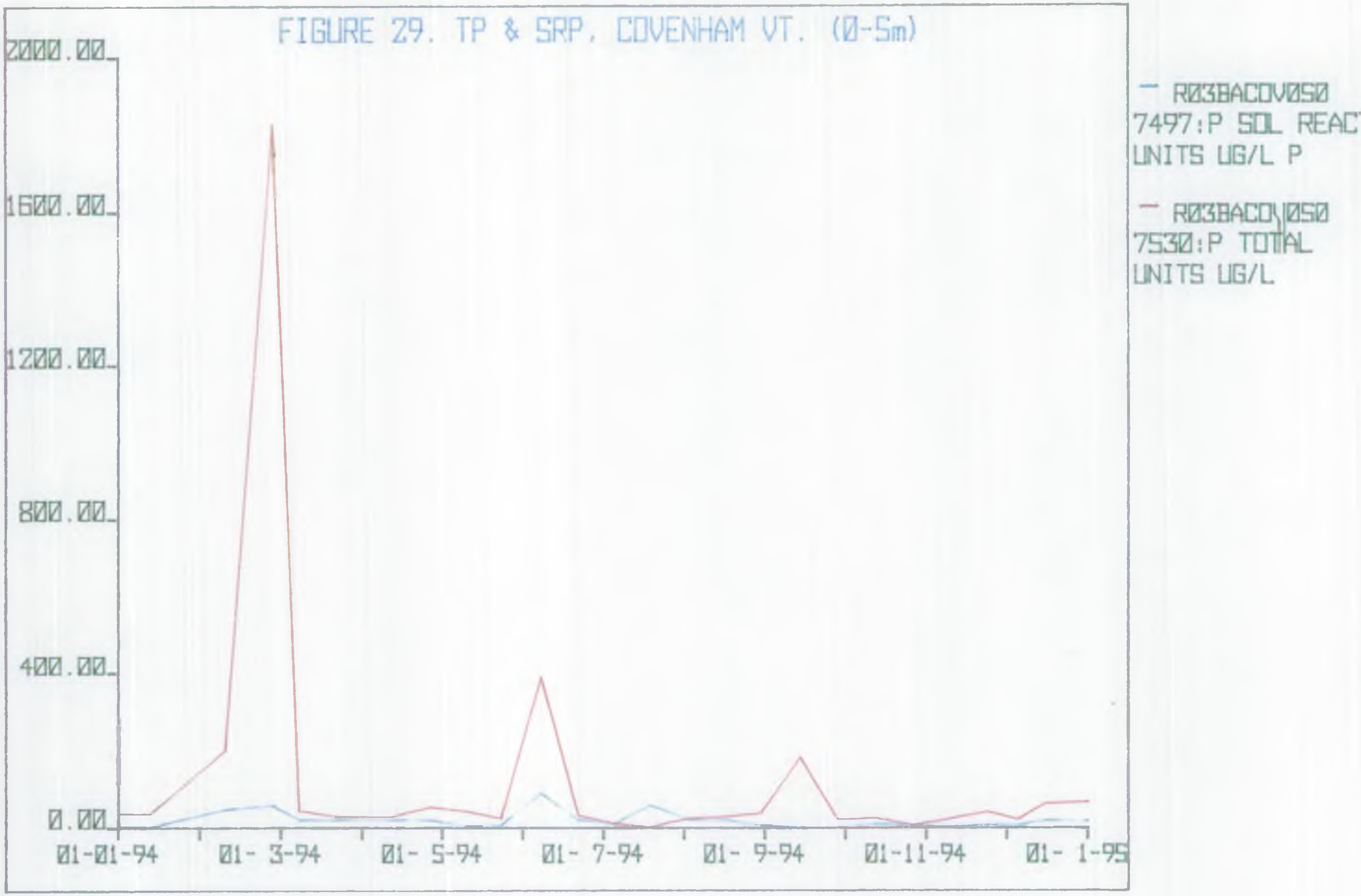
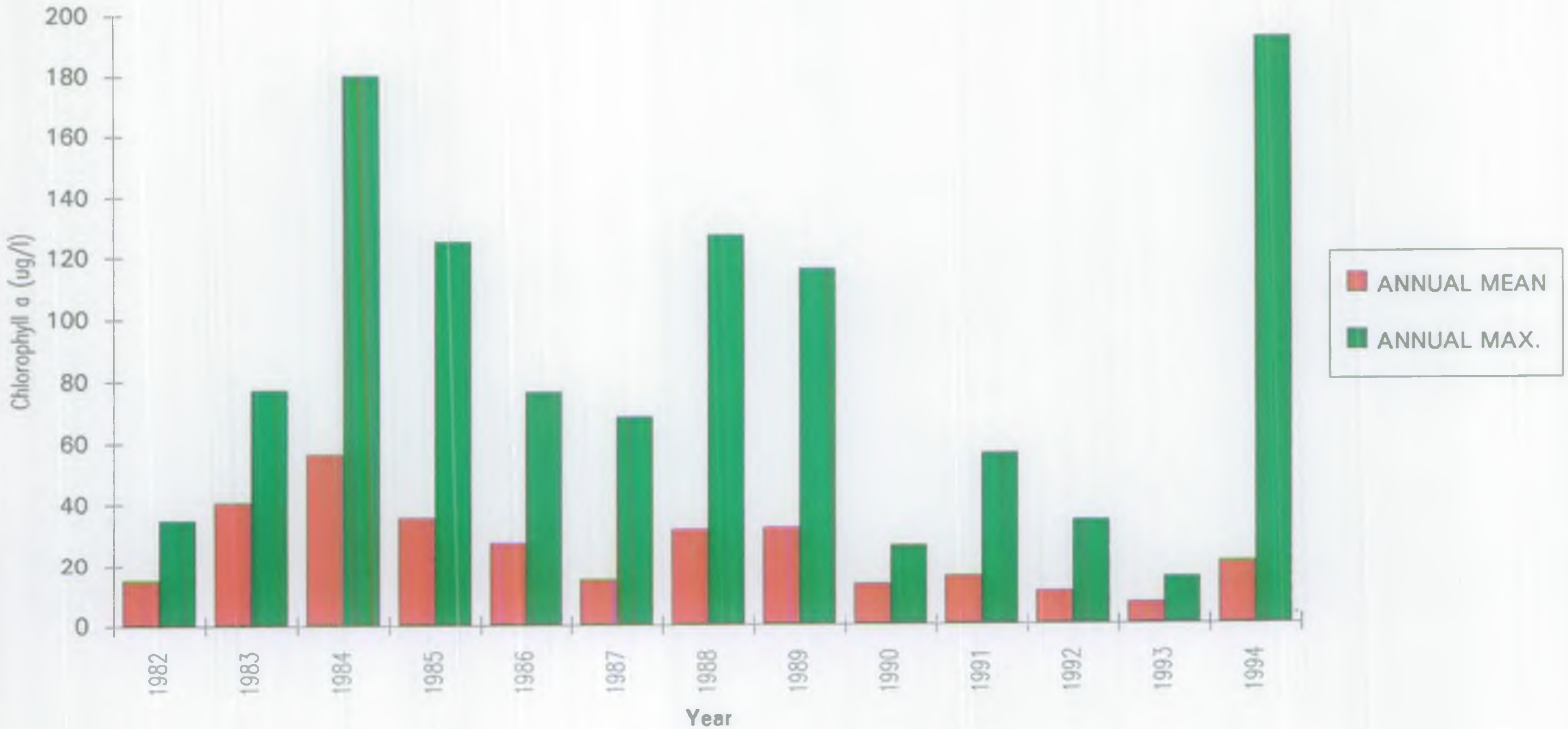


FIGURE 29. TP & SRP. COVENHAM VT. (0-5m)
2000.00
1600.00
1200.00
800.00
400.00
0.00
01-01-94
01- 3-94
01- 5-94
01- 7-94
01- 9-94
01-11-94
01- 1-95
R03BACOV050
7497:P SOL REAC
UNITS UG/L P
R03BACOV050
7530:P TOTAL
UNITS UG/L

FIGURE 30.
Chlorophyll a annual mean and annual maxima at Covenham reservoir between 1982-1994. Data between 1982-1990 taken from Centre Draw-off Tower, data from 1991 onwards taken from the Valve Tower.



The temporal changes in chlorophyll at the Valve Tower (0-5metres) during 1994 are shown in Figure 31. The spring diatom peak reached nearly 40ug/l in early March but rapidly declined as silica became limiting (NRA unpublished data). Concentrations remained below 10ug/l throughout the early summer but increased very rapidly in July reaching a peak of 192ug/l. This peak was dominated by the cyanobacteria, *Aphanizonenon flos-aquae* but as the biomass declined to just below 40ug/l and remained at that level throughout August and the early part of September, *Microcystis aeruginosa* became the dominant taxa. Both of these cyanobacteria remained the dominant taxa until the end of the year.

The temporal changes in cyanobacteria (all taxa) at Covenham Valve Tower (0-5metres) between 1990 and 1994 is shown in Figure 32. The peak biomass recorded during July and August 1994 was as a result of the very large bloom of *Aphanizonenon flos-aquae* which reached over 150,000 cells/ml.

The pattern and levels of biomass in 1994 were similar to those recorded in pre ferric dosing years.

5. OTHER WORK.

a) A Livingstone Deep Core was taken of the sediment at the Inlet of Rutland Water by workers from University College London. At the Inlet the depth of the ferric floc was found to be approximately 2 metres (4 years of floc plus sediment) which overlay approximately 15cm of natural sediment (15 years of "natural" sediments). Heavy metal analysis is yet to be carried out on the samples.

b) A BSc. student from the University of Leicester has carried out a project on the Zebra Mussels (*Dreissena polymorpha*) in Rutland Water. Spatial distribution and age/size characteristics of populations in the reservoir were assessed. Translocation experiments also were carried out. Caged mussels from uncontaminated areas of the reservoir were placed in contaminated and control areas of the reservoir and growth and mortality were evaluated.

c) The Institute of Freshwater Ecology has recently carried out a project concerned with the measurement of histological stress in certain macroinvertebrates, including Zebra Mussels (*Dreissena polymorpha*) from Rutland Water. Body burdens of heavy metals in the mussels were also studied.

d) Analysis of W grade ferric sulphate for trace metals started and preliminary results (including E & A West's typical analysis) are summarised in Table 6. The trace contaminants in W grade ferric sulphate were found to be extremely variable. Ferric sulphate quality control data from AWS were not available at the time of writing this report.

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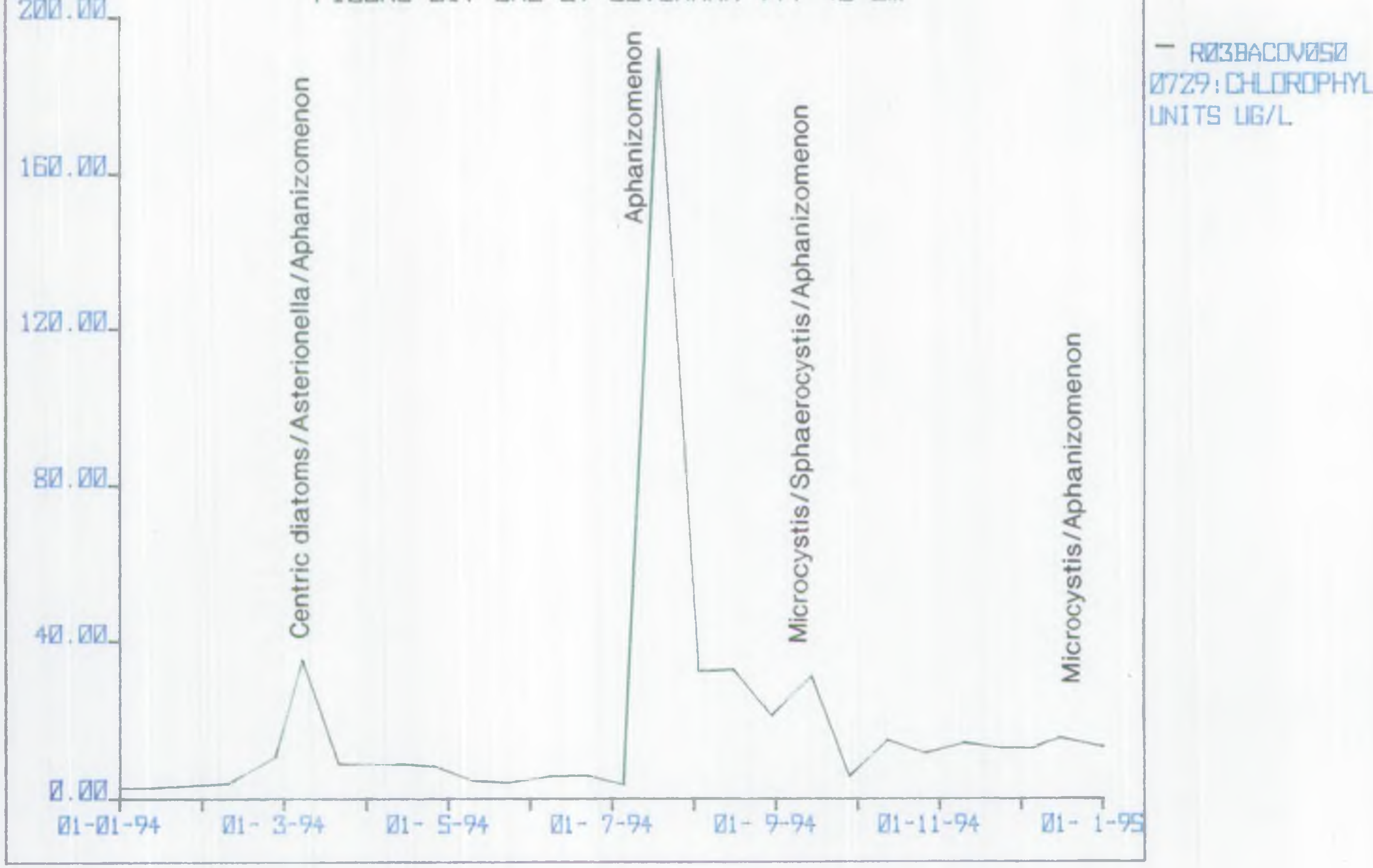


FIGURE 31. CHL a. COVENHAM VT. (0-5m

FIGURE 32. TEMPORAL CHANGES IN CYANOBACTERIAL DENSITIES, COVENHAM VALVE-TOWER (0-5m), 1990-1994

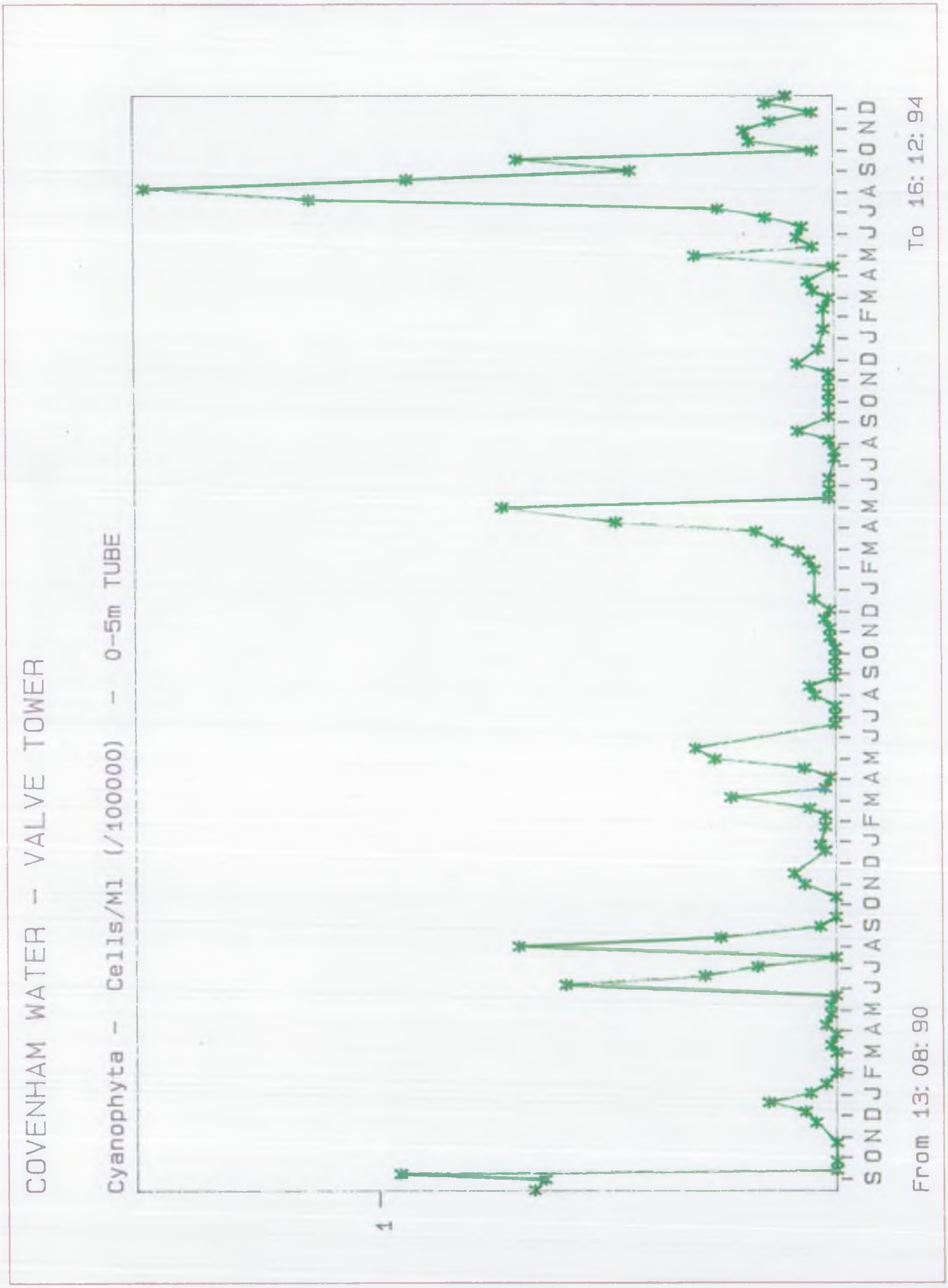


Table 6. Mean, maximum and minimum concentrations of trace metals found in W grade ferric sulphate.

METAL	MEAN CONCENTRATION (mg/l)	MINIMUM CONCENTRATION (mg/l)	MAXIMUM CONCENTRATION (mg/l)
Nickel	17.08	2.34	28.7
Chromium	5.62	4.53	11.7
Copper	3.45	0.76	3.95
Lead	1.2	0.23	7.55
Zinc	47.0	4.94	120.8
Cadmium	0.62	0.0044	3.02
Cobalt	21.9	12.6	27.18
Arsenic	0.108	0.34	<1.51
Mercury	0.009	0.0048	0.031
Iron	177000	164000	188000
Manganese	899.5	799	1057
Titanium	1308.5	832	1830

Between June 1990 and August 1995, 45983 tonnes of ferric sulphate have been dosed directly into the reservoir. Therefore, as can be seen from these preliminary results, the loadings of the metals (listed in Table 6) may have been significant.

e) A inter-laboratory exercise carried out by the NLS lab (NRA) and Cambridge lab (AWS) to investigate metal concentrations in Rutland sediments was carried out during 1994. Six Jenkin's cores were taken from various sites at Rutland Water on the 12 October 1994. The 0-5cm section of these cores was extruded and sent to the NLS.

These samples were freeze dried, ground and divided into 4 portions by the cone and quarter process. Two portions were sent to the AWS lab, one was retained by the NLS for analysis and the final portion retained for reference. Two standard reference sediments (CRM277 and NIES 2) were also analysed. Analysis of copper, lead, titanium, cadmium and iron from all samples was carried out.

The results (3 replicates) from the reservoir sediments and the certified reference material for the NLS lab are summarised in Figure 33. (The analytical methods used by the two laboratories were different).

CERTIFIED REFERENCE MATERIAL - CRM 277.

All results in mg/kg.

	Cu	Cd	Pb	Ti	Fe
REFERENCE VALUE	101.7	11.9	146	3200	45540
REPLICATE 1	110.2	12.4	130	350	40900
REPLICATE 2	97.1	12.3	137.8	399	39400
REPLICATE 3	98.5	11.7	131.1	380	39300

CERTIFIED REFERENCE MATERIAL - NIES 2.

All results in mg/kg.

	Cu	Cd	Pb	Ti	Fe
REFERENCE VALUE	219	0.82	105	6400	65300
REPLICATE 1	191	<2.0	44	3280	51800
REPLICATE 2	190	<2.0	50	3290	52200
REPLICATE 3	190	<2.0	50	3280	52200

SITE N1.

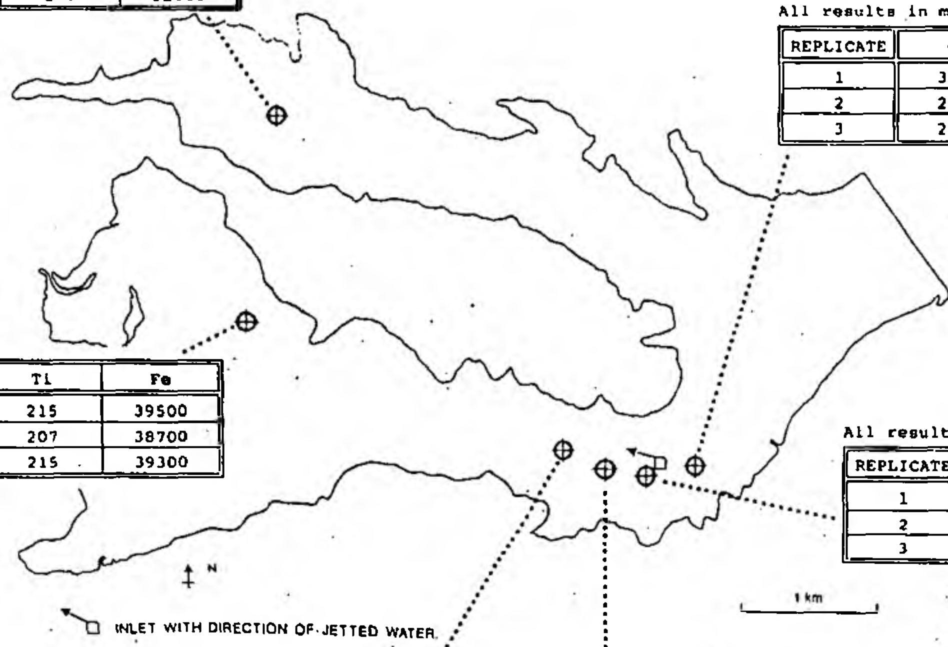
All results in mg/kg.

REPLICATE	Cu	Cd	Pb	Ti	Fe
1	29.0	<3.3	<40	243	50000
2	29.1	<3.3	<40	305	52500
3	29.7	<3	<36	245	52800

SITE AWS 16 BUOY.

All results in mg/kg.

REPLICATE	Cu	Cd	Pb	Ti	Fe
1	30.0	<3.1	41.6	999	139000
2	29.6	<3.4	<41	1057	142000
3	29.2	<2.6		1010	142000



SITE S13 BUOY.

All results in mg/kg.

REPLICATE	Cu	Cd	Pb	Ti	Fe
1	24.3	<2.6	<32	215	39500
2	23.7	<3.1	<37	207	38700
3	25.1	<3.4	<41	215	39300

SITE INLET BUOY.

All results in mg/kg.

REPLICATE	Cu	Cd	Pb	Ti	Fe
1	19.9	<3.4	62.4	4690	466000
2	19.2	<3.0	55.3	4460	445000
3	19.6	<3.3	55.8	4700	467000

SITE AWS 2 BUOY.

All results in mg/kg.

REPLICATE	Cu	Cd	Pb	Ti	Fe
1	34.9	<4.1	<43	889	131000
2	30.3	<3.0	<39	737	114000
3	29.2	<2.7	<41	733	113000

SITE SW2 BUOY.

All results in mg/kg.

REPLICATE	Cu	Cd	Pb	Ti	Fe
1	30.9	<3.3	<40	945	134000
2	29.7	<2.8	<38	918	134000
3	29.8	<2.9	<35	910	133000

FIGURE 33. METAL CONCENTRATIONS IN 0-5cm SEDIMENTS FROM RUTLAND WATER - 12 OCTOBER 1994.

The data set is very limited and further detailed investigations are required before any definite conclusions are made. Comparisons with sediments from other freshwaters can be made by referring to Appendix B.

f) A PhD student at Nottingham Trent University has been investigating the toxicity of ferric sulphate on Brown Trout has now completed his second year and has established a 96 hour LC_{50} and has suggested that the major toxic affect is due to physical clogging of the gills, leading to gill damage and changes in the physiological status of the fish.

Small point.
Range of interpretation.
? Places in the text
Not presented objectively.

6. CONCLUSIONS

? more precise about.

1) The direct ferric dosing carried out at Rutland Water since 1990 has continued to affect the benthic macroinvertebrate communities of the reservoir at both a chronic and acute level:

chironomids. ? as before.

- the fauna of contaminated areas had reduced diversities, densities and altered community structures.

reflected in recent survey.

- there were indications of a sub-lethal effect of the ferric contamination on the chironomids, resulting in delayed development of the larvae.

clarification.

2) It appears that there is little possibility of iron concentrations declining in the inhabited layers of the sediment even if ferric dosing is continued intermittently at very low levels:

- despite the greatly reduced ferric dosing regime since 1993 there were only slight indications of a reduction in sediment iron levels in the worst affected areas.

- sediment iron levels were seen to rapidly increase in response to resumed or increased dosing.

upward trend. balancing

3) There has been an indication of an increase in sediment iron concentrations on the marginal transects suggesting that the floc may be spreading outwards. ~~To date~~ there have been no apparent adverse effects on the benthic communities. Continued dosing or a spread from the most contaminated area may lead to a deterioration in the communities.

4) The littoral sediments of Rutland Water do not appear to be affected by the ferric dosing, and there is no evidence of adverse effects on littoral invertebrate communities.

Rutland

5) At Covenham high sediment iron levels were recorded at sites close to the Inlet, but generally no effect was seen on the fauna. It may be possible that the issue is being complicated by the introduction of animals from the Louth Canal to the reservoir via the Inlet, but this cannot be established from the data available. P. stifled.

6) Annual mean soluble reactive phosphorus levels have continued to remain low at both Rutland Water and Covenham reservoir. However total P has fluctuated at higher levels and at Covenham, as a result of poor P control, total P reached nearly 2 mg/l in the spring.' 94

7) Algal biomass remained relatively low at Rutland Water during 1994, however a large bloom of cyanobacteria (dominated by *Aphanizomenon flos-aquae)* appeared in late August reaching 220ug/l chlorophyll a (the highest concentration ever recorded during routine sampling at Rutland). The annual mean and maximum chlorophyll at Covenham reservoir was considerably higher than in the previous three years and were similar to the levels found in pre-dosing years. The summer peak was dominated by cyanobacteria and reached a chlorophyll concentration of 192ug/l in July. The densities of all cyanobacterial taxa were higher than in any year since dosing began.

8) Direct ferric dosing at both Rutland Water and Covenham reservoir has not been unequivocally successful in controlling eutrophication and reducing cyanobacterial growth.

- at Covenham, the instability of the system has been highlighted in 1994 - the biological system has reverted to pre-dosing biomass levels and algal succession even though the P is still quite low.

- at Rutland Water, a very large bloom of cyanobacteria developed during August.

9) In view of the questionable effectiveness of direct ferric sulphate dosing in controlling cyanobacteria and eutrophication, the deleterious effect on the benthic macroinvertebrates and sediments of these reservoirs, it is suggested that the use of this technique be progressively phased out and alternative eutrophication control measures investigated urgently.

sustain solution

7. RECOMMENDATIONS.

The main recommendations from the Northern Area survey work are as follows:- [illegible]

1. Direct ferric sulphate dosing should be progressively phased out over the next 5 years. Any spread of ferric outside the zone, bounded by Howell's Inlet and Lodge Farm transects, would result in this time scale being reviewed.

2. Direct dosing should continue be minimised during this period by managing the reservoir systems effectively, ie. investigation of alternative eutrophication controls methods (artificial mixing regimes, biomanipulation) as well as pumping in low P water, reducing drawdown etc.

3. Monitoring of the benthos, phytoplankton and sediment chemistry should continue. [illegible]

4. Routine detailed analyses of contaminants in the ferric sulphate and contaminants in the river inputs should be implemented or continued. [illegible] Total SRO data.

5. Further work which requires attention includes:-

   i) Investigations of distribution of heavy metals in deep core taken at the inlet during 1994.

   ii) Toxicity testing of the sediments from contaminated areas.

   iii) Review and change present sampling and analytical protocols if necessary - sediment chemistry requires further attention.

   iv) Detailed identification and mapping of the contaminated sediments around the inlets.

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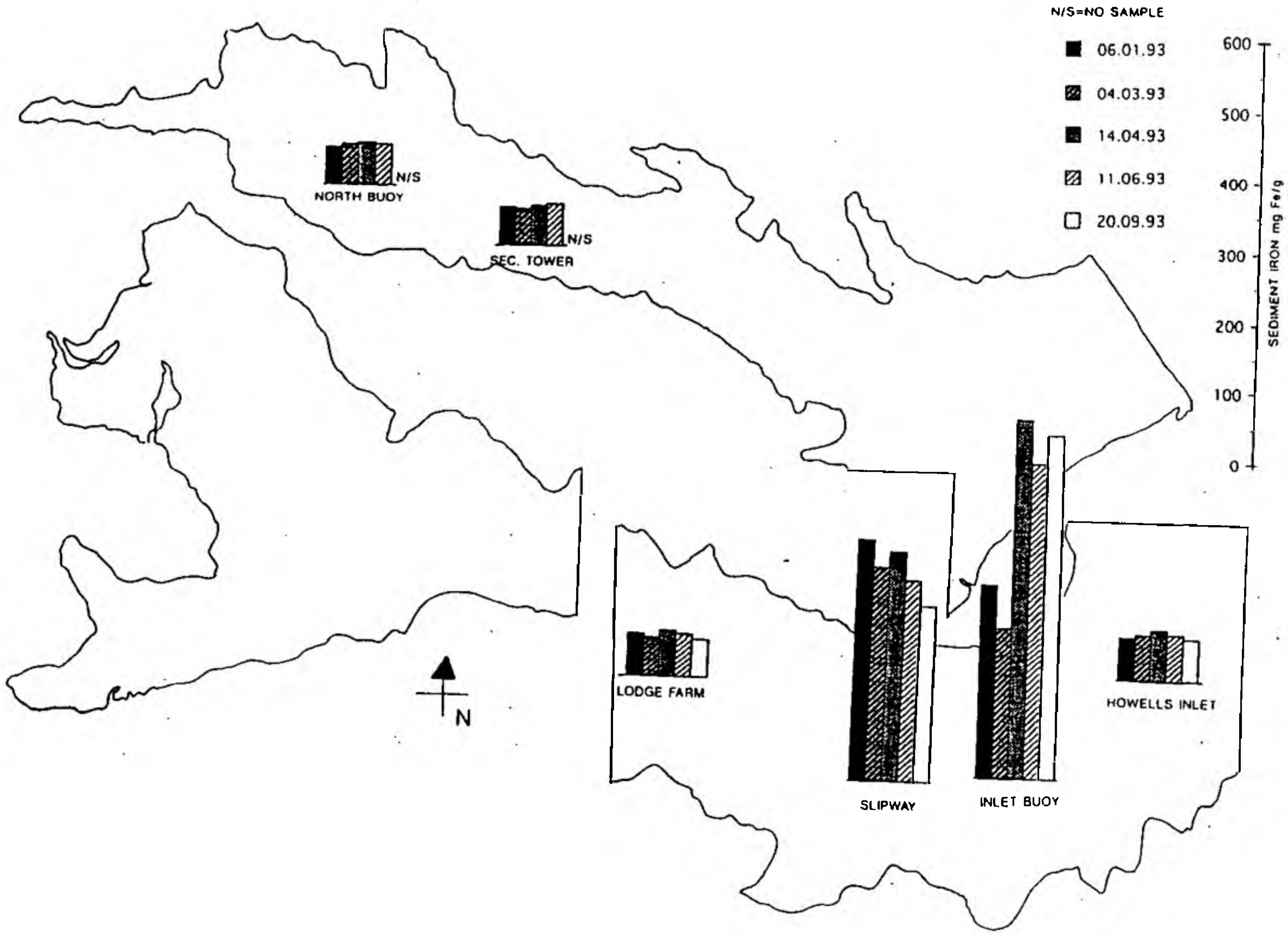
Appendix A.

The following figures are reproduced from "Ferric dosing of reservoirs: A review of dosing and its impact in the Northern Area during 1993", Christmas et. al (1994) to allow comparison with historical trends.

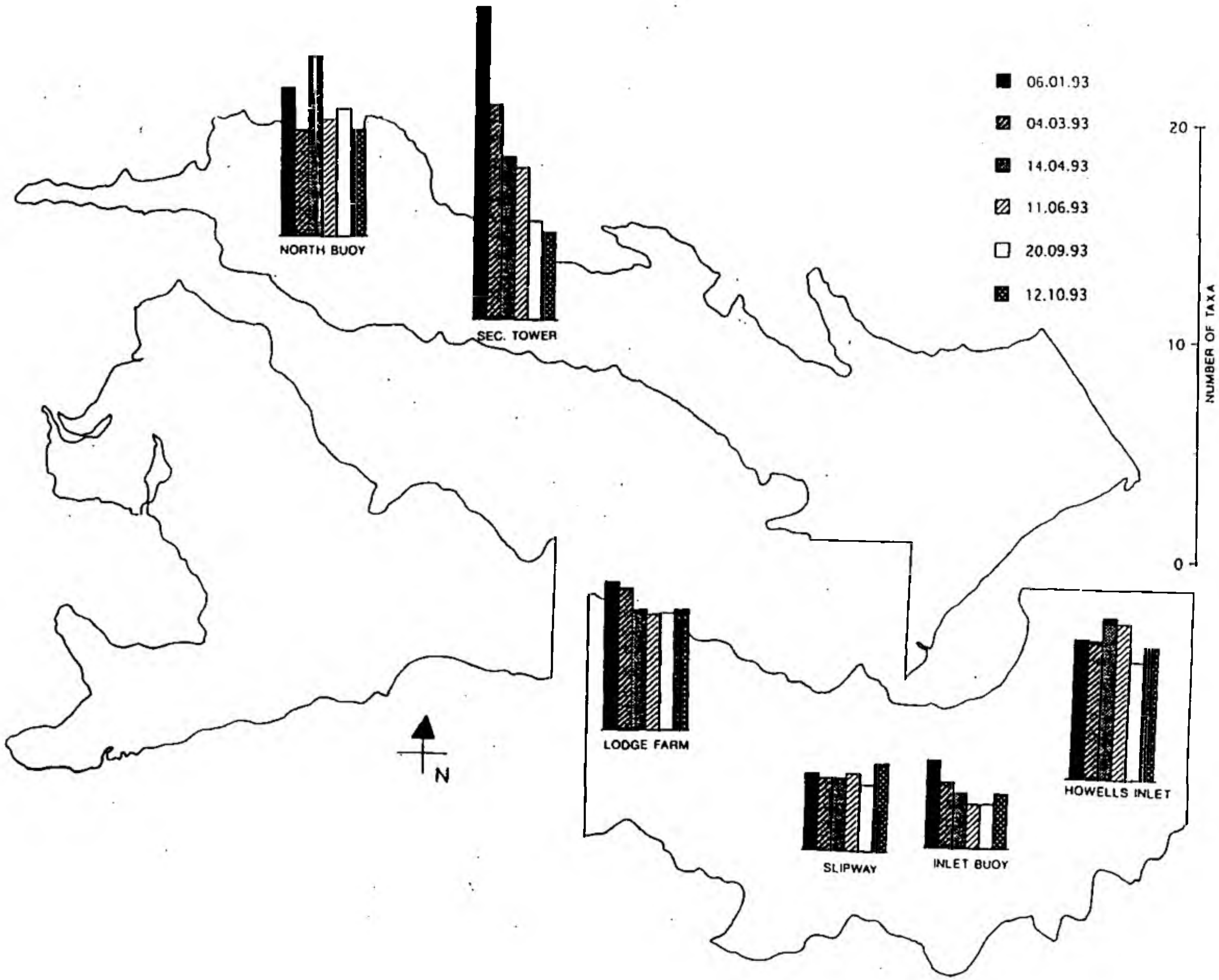
MEAN TOTAL IRON IN SEDIMENT AT RUTLAND WATER 1993.

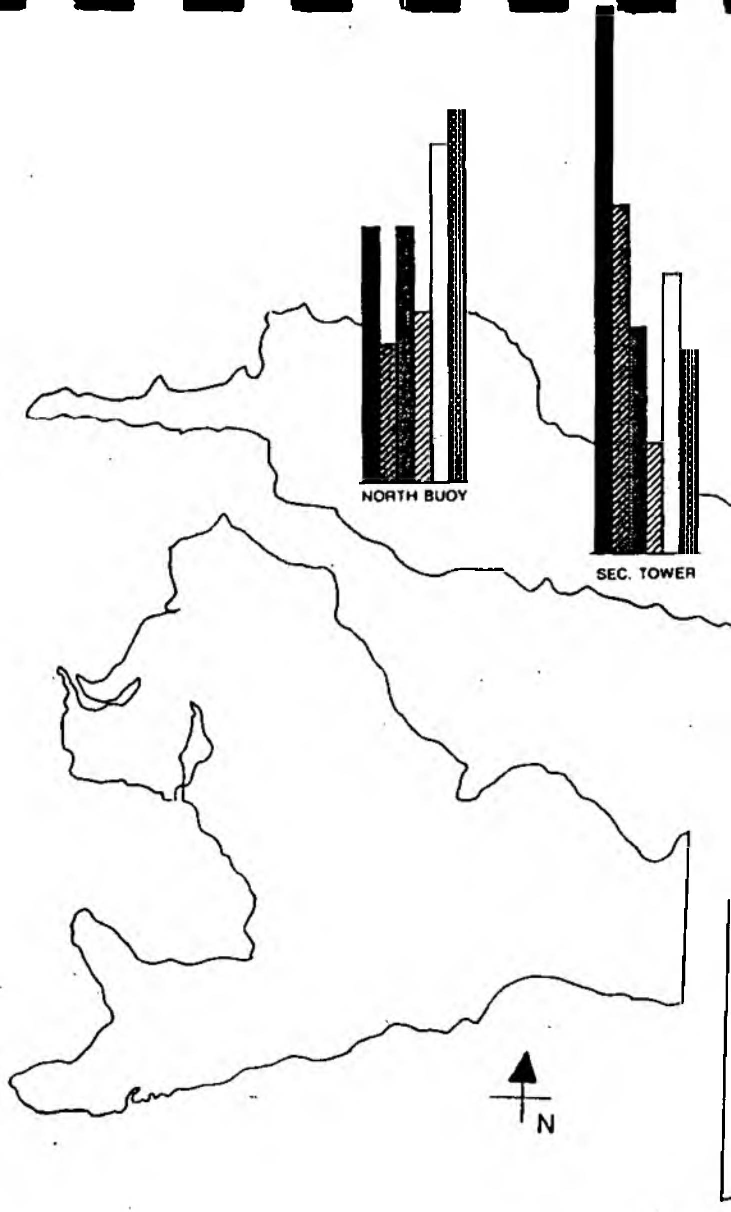


APPENDIX A. FIGURE 1.

APPENDIX A. FIGURE 2.

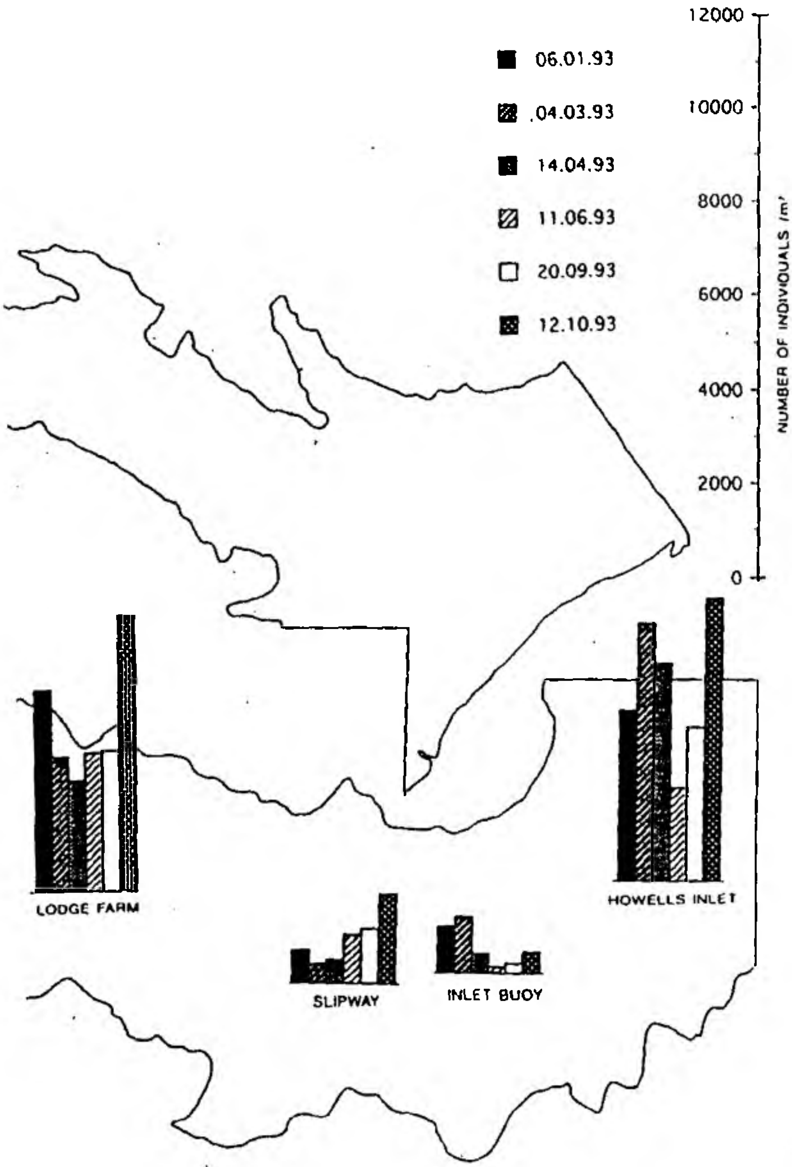
MEAN NUMBER OF TAXA AT RUTLAND WATER 1993.

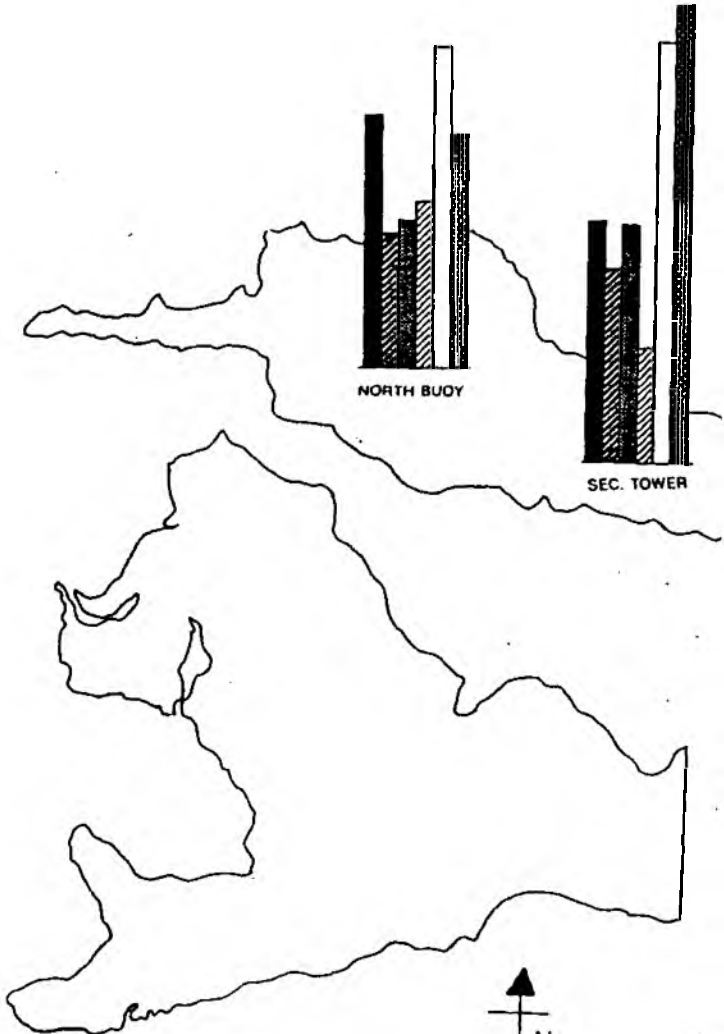



NORTH BUOY
SEC. TOWER
N

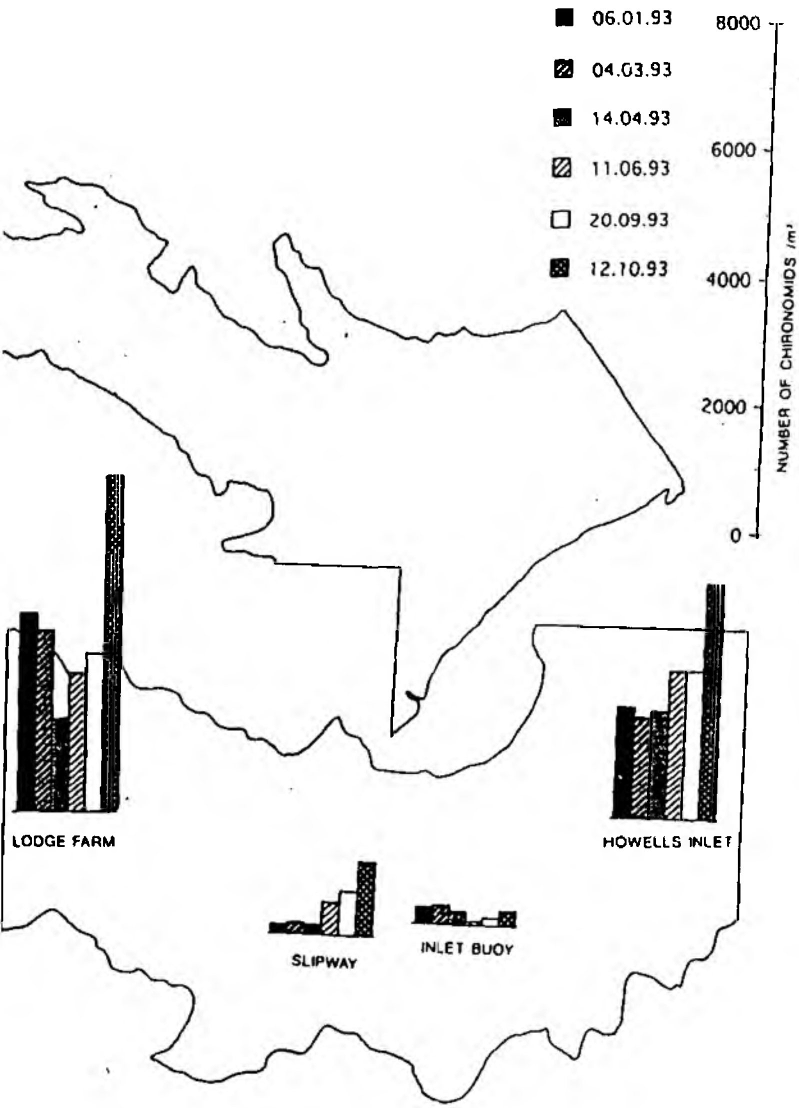
APPENDIX A. FIGURE 3.

MEAN NUMBER OF INDIVIDUALS AT RUTLAND WATER 1993.




NORTH BUOY
SEC. TOWER
N

MEAN NUMBER OF CHIRONOMIDAE AT RUTLAND WATER 1993.



APPENDIX A. FIGURE 4.

APPENDIX B.

RANGES OF METAL CONCENTRATIONS IN SEDIMENTS FROM DIFFERENT FRESHWATER SITES.

SITE	Cu(mg/kg)	Cd(mg/kg)	Pb(mg/kg)	Ti(mg/kg)	Fe (%)
RUTLAND-"CONTROLS"	**23-30**	**<2.6-<3.4**	**<32-<41**	**207-305**	**3.87-5.28**
RUTLAND-"CONTAMINATED"	**19-35**	**<2.7-<4.1**	**<39-62**	**733-4700**	**11.3-46.7**
RIVER YARE (a)	3.2-130	1.9-3.9	136-190	ND	ND
LAKE WINDEMERE (b)	22-140	ND	51-552	ND	3.1-5.5
ESTHWAITE (c)	29-88	-7.7	53-366	ND	ND
LOCH BA, MULL (d)	~45	ND	50-150	ND	~4.5
MANILLA BAY RIVERS (e)	28-189	1.2-15.2	11-220	ND	1.0-4.1
BOLTS BURN u/s MINE DISCHARGE (f)	ND	0.6-3.8	96-120	ND	ND
BOLTS BURN d/s DISCHARGE (f)	ND	3.1-11.4	1600-3120	ND	ND

ND = NO DATA AVAILABLE.

REFERENCES:

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