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Predicted Effects of proposed changes in patterns of water abstraction on the ecosystems of the Lower River Thames and its Tidal Estuary.

FINAL REPORT

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PREDICTED EFFECTS OF PROPOSED CHANGES IN PATTERNS OF WATER ABSTRACTION ON THE ECOSYSTEMS OF THE LOWER RIVER THAMES AND THE THAMES ESTUARY.

CONTENTS.

This report was commissioned by the Thames Water Authority as part of evidence to be submitted to a public inquiry into a proposal to remove the statutory flow constraints which relate to Teddington Weir. The work was the responsibility of the following NERC Institutes and staff.

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1. EXECUTIVE SUMMARY

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1.1. BACKGROUND

Since 1911 the Teddington Statutorily Protected Flow of the Thames has been set at 772 thousand cubic metres per day (tcmd) or 8.93 cubic metres per second (cumecs) except that after May 1 each year, if the quantity of water in the London storage reservoirs is below some target level, it may be further reduced in steps of 45.4 tcmd to an absolute minimum of 227 tcmd (2.62 cumecs). The rules which govern these reductions relate to target levels of reservoir storage between May 1 and January 31. Unfortunately the rules proved particularly restrictive during the drought of 1976 and, as a result, actions were necessary which reduced summer flows well below 227 tcmd for most of July, August and September, back pumping over Teddington and Molesey Weirs was required to maintain water levels. The Thames Water Authority (TWA) now wish to change the rules which determine minimum flows at Teddington to allow more water to be abstracted earlier in the year, as this would be a very cost-effective way of improving the reliability of London's water supply, and they have commissioned the Natural Environment Research Council (NERC) to assess the ecological impact of these changes.

1.2. OBJECTIVES AND METHODS

In order to predict the effects of changes in the pattern of water abstraction on the biota, the following objectives were pursued:

1.2.1. To present mathematical models which simulate the water quality of both the Lower Thames River and the Thames Estuary and to demonstrate their ability to reproduce measured water quality for a number of historic flow sequences. The NERC Institute of Hydrology (IH) has developed a flow, dissolved oxygen (DO) and biochemical oxygen demand (BOD) model for the non-tidal river, whilst the NERC Institute for Marine Environmental Research (IMER) has adapted a Thames Water quality model to predict daily distributions of salinity, ammonia, total BOD, nitrate and DO along the length of the estuary.

1.2.2. To use these models to give predictions of the changes in water quality resulting from current water demand under the present statutorily enforced abstraction rules and contemporary pollution loads, and to demonstrate the effect of changing the rules to a scheme designed to provide a more reliable water supply.

1.2.3. To discuss the implications of these predicted changes in water quality for the ecology of the river and estuary. This assessment has been made by a number of ecologists specialising in various aspects of river and estuarine biology at the Freshwater Biological Association (FBA) and at IMER.

1.3. ENVIRONMENTAL QUALITY

Flow, velocity and water quality conditions under current operating strategies have been compared with a new Maximum Resource Benefit (MRB) policy assuming 1984 demands.

1.3.1. <u>The non-tidal river above Teddington Weir.</u>

1.3.1.1. The flows and velocities during drought conditions are likely to be very similar under the MRB and current strategies, but the MRB strategy will slightly improve the situation since it should be possible to avoid extreme low flow conditions and necessity to back pump over Teddington and Molesey Weirs. In general the flows over Molesey Weir are lower than those at Teddington because of the major abstractions upstream of Molesey and the inflow of the Hogsmill and Mole rivers below it.

1.3.1.2. Flow and velocity in non-drought years would be considerably reduced by the MRB strategy, increasing the reach residence time. However, non-drought spring condition velocities will not be affected significantly, suggesting that abstracting additional water earlier in the year would certainly be preferable to making up losses through lower flow in the summer.

1.3.1.3. DO levels are generally close to saturation for most of the year. In drought years the BOD concentrations are similar for both operating strategies and it is only in non-drought summers that any increase in BOD levels can be detected, and even these are insignificant. The MRB rules would usually improve DO concentrations slightly in drought summers since extreme low flows would be avoided.

1.3.1.4. Ammonia levels are generally less than 0.5mg per litre. Occasional high mean values of ammonia were recorded in the Molesey to Teddington reach in the drought of 1976 due to back pumping, but recent improvements to treatment facilities and the avoidance of back pumping with the MRB strategy would improve the situation. Under drought conditions, the highest mean levels of ammonia would be approximately a third of the peak 1976 concentrations.

1.3.2. <u>The tidal river below Teddington Weir.</u>

1.3.2.1. Comparison of the current and MRB policies on salinity, ammonia, BOD, total nitrogen and dissolved oxygen during the drought sequence 1975/76 shows a very small change which is only detectable upstream of London Bridge. This is also true of the seasonal variability of these concentrations. In non-drought years there would be a slight increase in ammonia, BOD and nitrate with an associated small drop in DO upstream of London Bridge, but absolute levels would not approach those currently experienced under drought conditions.

1.3.2.2. At Teddington and Syon Park salinities would not increase by more than 0.4 parts per thousand in the 1976 drought, and on occasion the MRB policy reduces salinity by as much as 1.5 parts per thousand. At London Bridge there is no consistent effect of the MRB policy but the day to day variation is reduced slightly. In non-drought years the water above Richmond Half-tide Lock rarely becomes saline under either management system. However, at Syon Park the MRB policy results in higher salinities, but the maximum values of 1 part per thousand are still much less than equivalent drought values (3 parts per thousand) under either system. The same is true at London Bridge, where the relative increase in salinity has a maximum of less than 5 parts per thousand at a time when the absolute salinity is about 10 parts per thousand.

1.3.2.3. In the drought sequence 1975/76 oxygen levels are not significantly reduced, and in some cases are improved slightly. In non-drought years there is no measurable effect on the oxygen regime above Richmond Half-tide Lock or at London Bridge, the only significant reduction occurring at Syon Park Reach where the oxygen concentrations are highest, but even here DO is never lower than currently experienced in drought conditions.

1.3.2.4. Similar trivial effects were observed in simulations of other classical drought sequences. Daily time series results for London Bridge do in general show how the salinity there increases slightly earlier in the drought under MRB operation, sometimes reaching a higher maximum value, but these differences are small (less than 5 parts per thousand) and of short duration.

1.4. ECOLOGICAL CONSEQUENCES WHICH HAVE BEEN CONSIDERED

There is a scarcity of good information about the flora and fauna of the lower river and estuary, which makes accurate predictions about the likely effects of the small changes in environmental conditions indicated above difficult. Unfortunately, many aspects of the ecological effect of the severe 1975/76 drought are rather poorly documented, but some information exists which can be used as an empirical yardstick against which possible changes due to the MRB policy can be assessed. The biological predictions based on current levels of demand for water should not be considered applicable under increased levels of demand. However, it is possible in this section to give some general indication of the consequences of very reduced flows, but as will be indicated in the next section (1.5.), the majority of these is unlikely to be realised under MRB policy.

1.4.1. Diurnal variations in DO in the river during the summer can be significant and are related to solar radiation levels and the growth of algae within the water column. During the summer months extremely high concentrations up to 180 percent saturation have been measured, but low concentrations of 40-50 percent have occasionally been recorded, and lower values are likely at night. Flow also has an important effect on DO levels because of reaeration processes and the residence time of water in a reach. Reaeration increases with increasing flow rate and effluents are also more diluted. Under low flow conditions residence times are high, which allows more time for decay of organic material and contributes to favourable conditions for the growth of algae. In non-drought summers there could be a possibility of increased algal growth and this might have a detrimental effect on DO levels when the bloom collapsed. Any problem of this kind which may arise is likely to be particularly acute in the reach above Molesey Weir because of the generally low flows there.

1.4.2. Reduced flows in spring and early summer could lead to increased sedimentation of suspended particles which would allow increased light penetration and this could give rise to large growths of filamentous algae. Enteromorpha forms floating growths in some backwaters and these could invade the main river, where the growths would be unsightly and could exacerbate the dissolved oxygen levels.

1.4.3. Large growths of planktonic algae during the spring and early summer are controlled by light and turbidity, and whilst reduced flows may affect the timing and duration of the peak periods they should not affect the quantity or species composition. Changes in summer, perhaps favouring greater quantities of bloom-forming cyanobacteria (blue-green algae) cannot be discounted.

1.4.4. Changes in higher plants could occur if there were long periods with little or no flow. A floating cover of duckweed is probably the most undesirable change that can be envisaged. This would be unsightly and restrict gaseous exchanges and light transmission at the surface, leading to low oxygen and high carbon dioxide concentrations in the water.

1.4.5. The factors most likely to affect the freshwater invertebrate fauna of the river, on which most of the fish feed, are sudden changes in flow conditions and long periods of very low flows in summer. Computation of biotic indices for the available data suggest that the reach most sensitive to sustained low flow is between the Hogsmill and Teddington Weir. In the tideway salinity is the 'master factor' governing esturine distributions provided other conditions such as oxygen concentration remain favourable, which the models show to be the case. Salinities between 5 and 7 parts per thousand constitute a significant ecological boundary: salinities above London Bridge are generally lower then this and the fauna is predominantly a freshwater one. However, marked fluctuations in the invertebrate fauna occur at this boundary under current operating conditions, both seasonally and between years, and very low flows would enhance the magnitude of such fluctuations.

1.4.6. Estuarine fish will move up and down the estuary following their preferred salinity. Reduction in flow in the spring could however affect anadromous and freshwater fish in several ways. There would be less flow to stimulate the upstream migration of salmon. Silt deposition would be increased and this could affect the eggs of fish such as salmon and dace which spawn on the bed of the river. If salmon become re-established in the river there is a danger of smolts being diverted into the reservoirs during their downstream migration. Similarly the spawning areas of dace and smelt in the upper part of the estuary could be restricted if salt water penetrated far into the estuary in spring.

1.5. CONCLUSIONS

In general the environmental effects of the MRB compared with the current policy are small: any detrimental effects are more noticeable in non-drought years, whereas in drought years conditions appear to show a general improvement. Since acute environmental problems are only likely to occur during droughts, the overall effect of the policy would therefore seem to be beneficial. Slight deterioration in non-drought years would certainly not create conditions as severe as those currently experienced in droughts such as 1975/76.

1.5.1. No serious ecological problems arose resulting from algal growth in the river and subsequent reduction in DO in 1975/76, and seem unlikely to do so under the proposed changes in abstraction rules. The slight deterioration in dissolved oxygen in the reach above Molesey Weir under MRB policy may be more than offset by improvements in the quality of effluents discharged to the river in this area, but nocturnal levels of oxygen will be lower than the daily averages predicted by the models.

1.5.2. There are no records of large growths of filamentous algae such as *Enteromorpha* invading the main river during 1975/76, and it is unlikely that the small changes resulting from MRB policy will produce such an effect.

1.5.3. The overall effect on the bacteriology of the river is likely to be negligible. Although there was some growth of the bloom-forming cyanobacterium Microcystis in 1976 it did not become a serious ecological problem and seems unlikely to do so under MRB policy.

1.5.4. There was apparently some development of duckweed in backwaters during 1976, but the plant did not invade the main river and seems unlikely to do so under MRB policy. Wind action and disturbance by boats both inhibit the spread of this plant.

1.5.5. Provided sudden changes in flow conditions are avoided it seems unlikely that the proposed changes will have any significant effect on freshwater invertebrates. During drought years conditions are more likely to improve than to deteriorate under MRB regulations. During non-drought years the predictions of BOD and DO suggest that conditions will deteriorate slightly, but a prudent extraction policy in spring should avoid any detectable adverse effects on macroinvertebrate communities. In the tideway, differences in salinity between the current and MRB systems of management are trivial when measured against the present variability in the system. Resilience of the fauna is a feature of such dynamic regions.

1.5.6. Effects on fish spawning and migration are unlikely to be significant given the small changes in flow proposed.

1.5.7. As the future abstraction demand increases, the reliability of meeting that demand will reduce until, at 117% of the 1984 demand (i.e. by about the year 2006) the failure rate would be intolerable. Under the worst possible assumptions the absolute quality of the estuary would deteriorate in direct proportion to demand, but with the increased volume of fresh water in the effluent return, current salinity levels would be maintained. The relative effect of the MRB policy as opposed to the current scheme is similar to that predicted for current demands, with no severe detrimental effects.

2. BACKGROUND AND SCOPE OF STUDY

P.G. Whitehead and P.J. Radford

2.1 BACKGROUND

2.1.1. Since 1911 the Teddington Statutorily Protected Flow of the Thames has been set at 772 tcmd (thousand cubic metres per day) or 8.93 cumecs (cubic metres per second) except that after May 1 each year, if the receiving reservoir is below some target level, it may be further reduced in steps of 45.4 tcmd to an absolute minimum of 227 tcmd(2.62 cumecs). The rules which govern these reductions relate to target levels of reservoir storage between May 1 and January 31. Unfortunately the rules proved particularly restrictive during the drought of 1976 and resulted in summer flows much lower than 227 tcmd for most of July, August and September: back pumping over Teddington and Molesey Weirs was required to maintain water levels. Such low flows, together with high ambient temperature, provided ideal conditions for algal growth, creating water supply problems and also the possibility of environmental problems. Water quality conditions were affected in the non-tidal part of the river with possible adverse affects on fish stocks, whilst increased salinity below Teddington Weir resulted in an upstream movement of estuarine invertebrate and fish faunas.

2.1.2. The Thames Water Authority (TWA) consequently wish to change the rules which determine minimum flows at Teddington to allow more water to be abstracted earlier in the year, hopefully preventing a recurrence of 1976 conditions and improving water quality, and have commissioned The Natural Environment Research Council (NERC) to assess the ecological impact of these changes.

2.2 SCOPE OF THE STUDY

2.2.1. The overall objective of the NERC study has been to assess the ecological impact of changes that could follow from a new operating strategy. In order to investigate possible biological changes it is first necessary to study the physical and chemical behaviour of the river and estuarine systems and to provide a means of predicting changes induced by a new operating strategy.

2.2.2. The NERC Institute of Hydrology (IH) and Institute for Marine Environmental Research (IMER) have employed mathematical modelling techniques to simulate physical and chemical variables such as river flow, dissolved oxygen (DO) and biochemical oxygen demand (BOD). The resultant mathematical model may be considered as a computer programme consisting of a set of equations which relate variables of interest and predict, to within satisfactory limits, the behaviour of the natural or modified system.

2.2.3. The I.H. has developed a daily flow, DO and BOD model for the non-tidal river upstream of Teddington Weir based on an earlier dynamic nitrogen balance model for river systems,a product of an earlier collaborative study with Thames Water (Whitehead and Williams 1982). It takes as its main driving variable the flow of the Thames at Datchet; a time series generated by the Thames Water Resources Model (Sexton et.al 1979). The main output of the model is a numerical assessment of water quality in the five lower reaches of the river Thames and the quality and quantity of the resultant flow over Teddington Weir. (See section 3 for further details).

2.2.4. The IMER have adapted the Thames Water Quality Model (Barrett et.al 1978) to simulate the daily residual flow, salinity, ammonia, BOD nitrate and DO distributions in the tidal estuary below Teddington. The model is driven by the output of the IH model of water quantity and quality predicted for Teddington Weir. In the Thames Estuary water quality is largly determined by the lateral discharges into the estuary and these are represented in the model as constant daily inputs based on annual average inputs, measured or proposed. The main output of the IMER model is a numerical assessment of water quality in 36 two-mile reaches of the estuary from Teddington to Shoebury Ness. (See Section 4 for fuller details).

2.2.5. Both of these models have been tested against observed data to demonstrate their ability to reproduce measured flow and water quality for a number of historic flow sequences. The models have then been used to predict changes in flow and water quality for two strategies of water abstraction above Teddington.

2.2.6. The existing strategy is based on the current rules as determined by the Chart and the new strategy is termed maximum resource benefit (MRB). In the latter case it is assumed that as much water as possible is abstracted from the Thames so that the reservoirs are maintained at their highest level possible at all times. The limit of abstraction would be when the flow over Molesley Weir is reduced to zero leaving only the discharges from the Mole and the Hogsmill to maintain a flow over Teddington Weir. This policy would ensure the most reliable water supply possible, given the present reservoir system and hopefully avoid the need to extract large quantities of water from the Thames during very low flow conditions. This in turn should alleviate ecological problems experienced during notable historic drought sequences.

2.2.7. An assessment of possible ecological changes has been undertaken by a number of ecologists from the Natural Environment Research Council (NERC); in particular from the Freshwater Biological Association (FBA) and the Institute for Marine Environmental Research (IMER). They have reviewed all the published literature for the Thames and produced a comprehensive bibliography which is available for inspection at the Thames Water Authority. The information generated by the models on flow and water quality has been used by the ecologists to assess all the possible effects on the natural ecosystem of possible changes in operating strategies.

2.2.7. In addition the effects of increased demands for water by the year 2006 have been considered. Simulations under the Chart and the MRB strategies have been obtained and again conclusions have been presented to the ecologists from which they have deduced the possible impact on the Lower Thames Biological System.

2.3 REFERENCES

Barrett, M.J., Mallowney, B.M. and Casaperii P (1978) The Thames Model: An assessment. Prog. Wat. Tech. 10,Nos 5/6, 409-416.

Sexton, J.R., Cook, D.J. and Jones A.E. (1979), Thames Water Resources Model; An Introduction publ. Thames Water. pp.8.

Whitehead, P.G. and Williams R.J. (1982), A dynamic nitrogen balance model for river systems In Effects of Waste Desposal on Groundwater and Surface Water. Proc. Symp. IAHS p.89-99.

3. EFFECTS ON WATER QUALITY OF THE LOWER THAMES

P.G. Whitehead and R.J. Williams

3. EFFECTS ON WATER QUALITY OF THE LOWER THAMES

3.1. BACKGROUND AND PROGRESS

3.1.1. As part of the NERC study the Institute of Hydrology (IH) has developed a model of flow, dissolved oxygen and biochemical oxygen demand (BOD) for the non-tidal section of the river, upstream of Teddington Weir. This research has followed other extensive modelling studies on the non-tidal river by IH in collaboration with Thames Water Authority.

3.1.2. Of particular relevance are two studies initiated in 1980 to develop a nitrate management model for the Thames and a model of algal growth, death and transport processes. In the case of the nitrate model a multi-reach, multi-tributary flow and water quality model was developed. The model could account for all inputs of flow and nitrate from tributaries, surface runoff, groundwater and effluents and allow for abstractions and losses of nitrate via denitrification processes. The model has been used extensively by Thames Water Authority to simulate nitrate behaviour given different agricultural trends and population levels. The main purpose of these predictions have been to assess the implications of WHO and EEC nitrate standards on water supply. Since nitrate levels are rising considerable investment in treatment plant or new reservoirs may be necessary to avoid exceeding WHO and EEC standards.

3.1.3. The algal model has been developed to investigate the key processes controlling algal growth, death and transport along the river. Major algal blooms occur during spring, summer and autumn and these can present operational problems for the water authority or water companies. Abstracted water is pumped into reservoirs or directly into water treatment plants. Algae affect water taste and smell and may block filtration equipment. Relatively little is known about the processes determining algal behaviour and the IH modelling study was designed to provide a predictive model that might be used eventually for operational management. A description of the study is given elsewhere (Whitehead and Hornberger, 1984)

3.1.4. The progress on the DO-BOD modelling study has been aided considerably by the previous studies mentioned since information on sources of water has been available togther with information on physical characteristics and the dynamic (day to day) behaviour of the river.

Basic flow data for inputs such as tributaries, effluents, surface run-off and groundwater have been obtained from the Thames Water Resources Model together with abstraction data. A total period of 61 years data have been obtained and flows from 1920 to 1981 have been simulated

Thus seven major drought episodes have been investigated together with over fifty years of non-drought flows.

3.1.5. The DO-BOD model has been developed for the lower reaches of the Thames from Romney Weir to Teddington Weir. The model accounts for the effects on reach quality of upstream inputs and effluent discharges and other processes such as BOD decay, mud respiration, reaeration, photosynthetic oxygen production and respiration by phytoplankton. The model has been calibrated using 1974/5/6 flow and quality data and has been used to simulate severe drought periods and non-drought periods.

3.1.6. The flow and quality models have been run under both the chart strategy and the MRB strategy for current demands and demands expected in the year 2006 so that changes could be investigated. Conclusions are drawn on flow and quality changes and information and data made available to IMER and FBA scientists.

3.2 DESCRIPTION OF THE RIVER THAMES SYSTEM

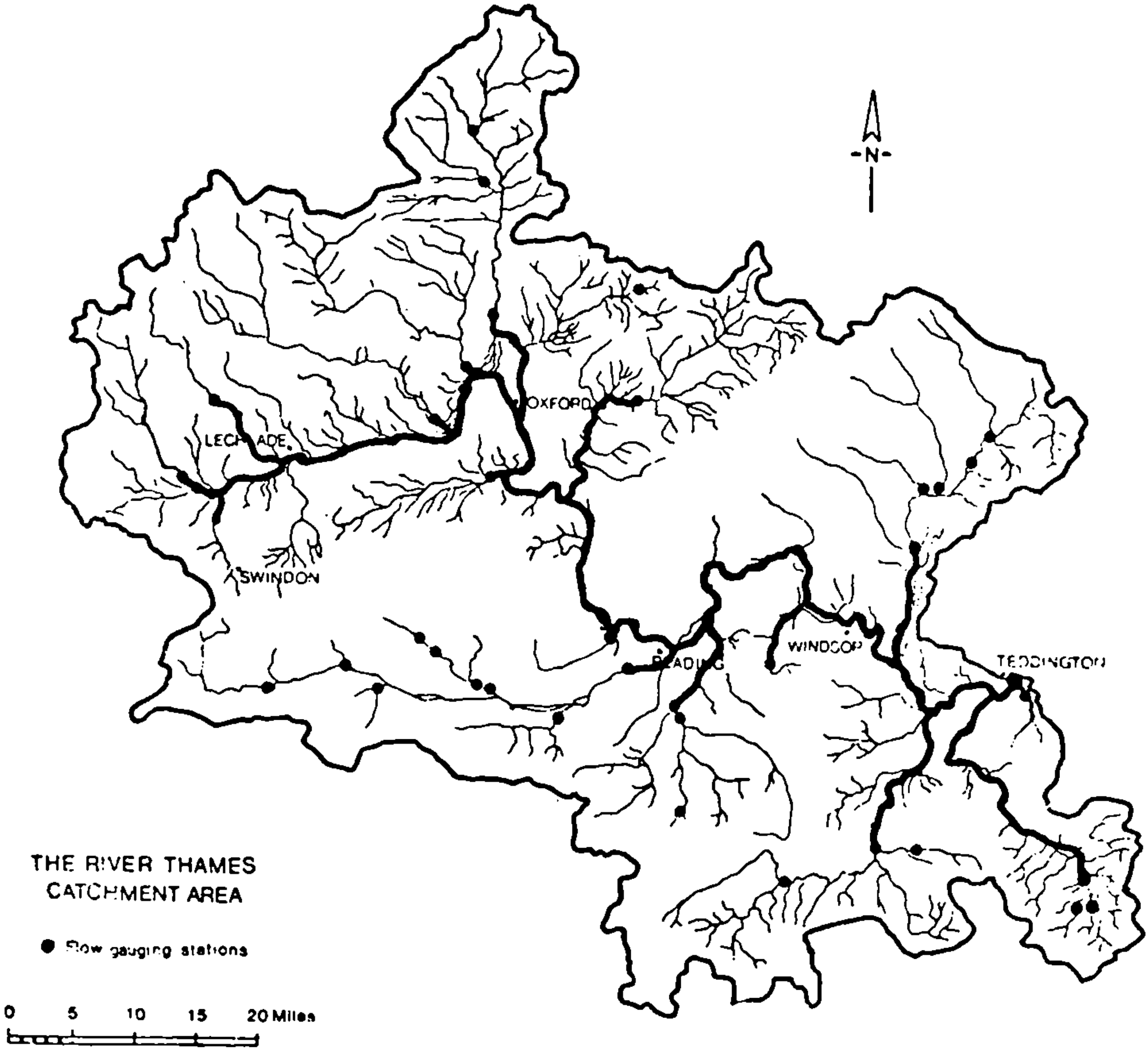
3.2.1. Figure 3.1. shows the Thames with its major tributaries and the location of flow gauging stations. The length of the main river is 236 kilometres with a fall of 108 metres. There are many locks and weirs on the river and these are shown in Figure 3.2. together with the geological strata underlying the alluvial bed.

3.2.2. The weirs have a large effect on water quality since in addition to regulating depth, they largely control mixing and aeration in the reaches. Except in times of high flow the depth regulation results in low velocity of flow, long retention times, a lower rate of aeration these are the ideal conditions for the growth of algae. The river is navigable and is used by numerous small boats in the spring, summer and early autumn. This tends to increase the turbidity of the river water as a result of scouring of bottom silts. The increased residence times associated with the regulating weirs is an important factor when considering water quality.

3.2.3. The quantities of sewage and trade effluent discharged to the Thames catchment above Teddington forms a high proportion of the total flow of the Thames in times of low flow. This is an important consideration in defining river reaches and Figure 3.3 shows the location of principal sewage outfalls.

3.2.4. Whilst some water is abstracted for public water supplies from the River Thames above Oxford, from the River Kennet at Reading,and from a number of chalk and limestone springs the largest abstractions are made from the reaches of the Thames between Windsor and Hampton. The main abstractors are the Thames Water Authority the North Surrey Water Company (NSWC) and

Figure 3.1. Location of gauging stations within the Thames catchment



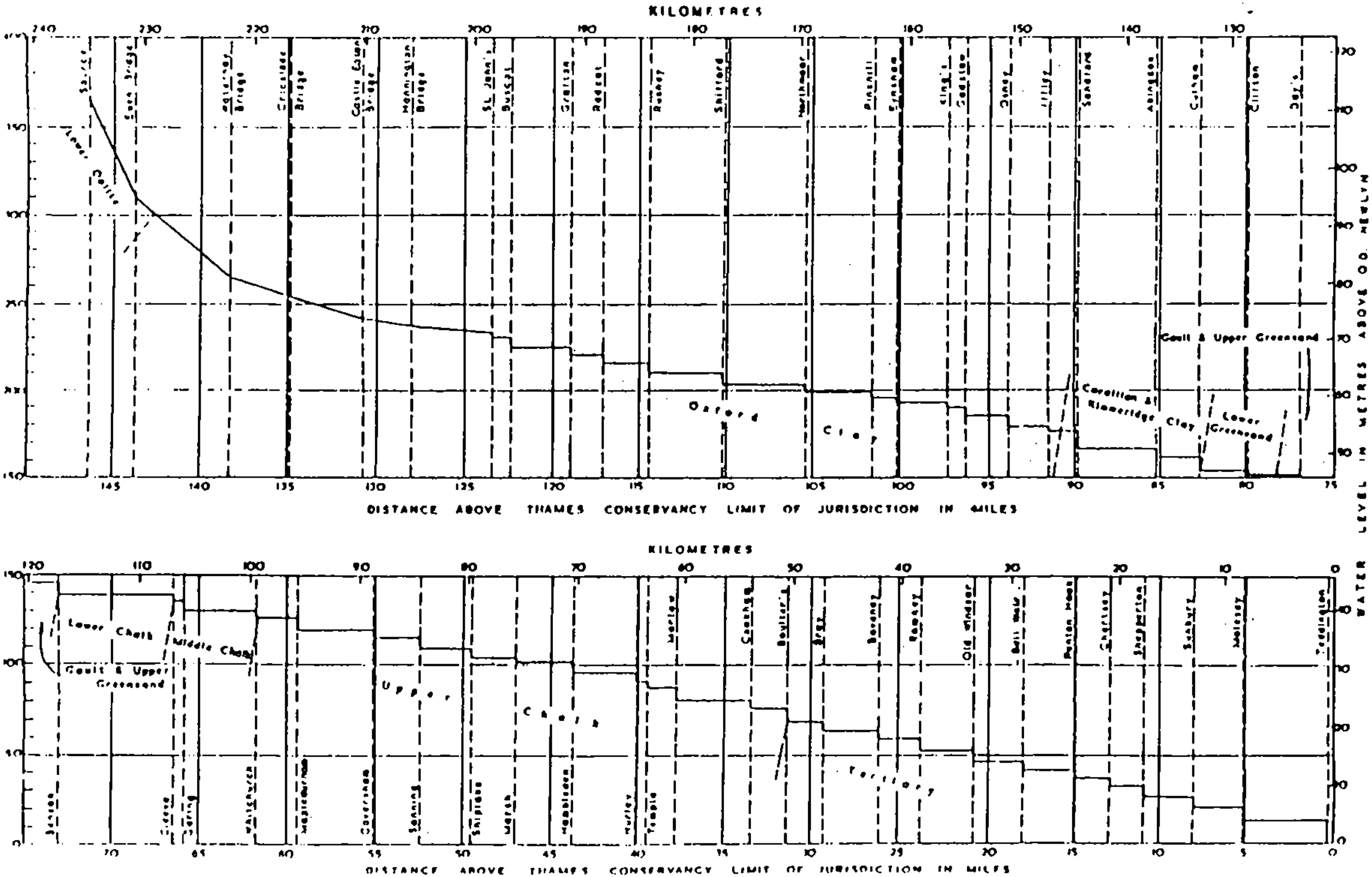


Figure 3.2 Longitudinal profile of River Thames showing locations of weirs and geological strata underlying the river bed.

Figure 3.3. Location of sewage outfalls within the Thames catchment

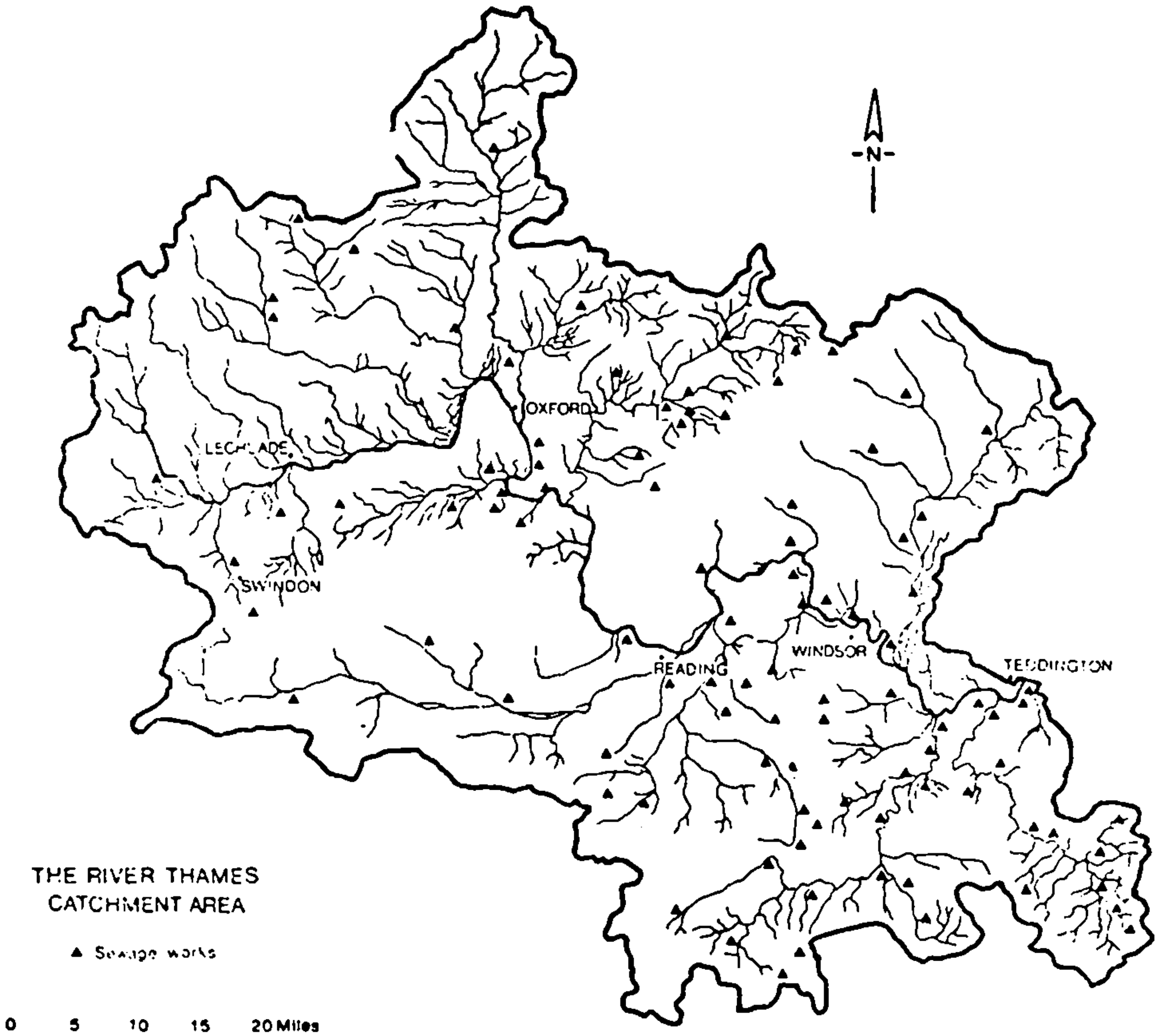
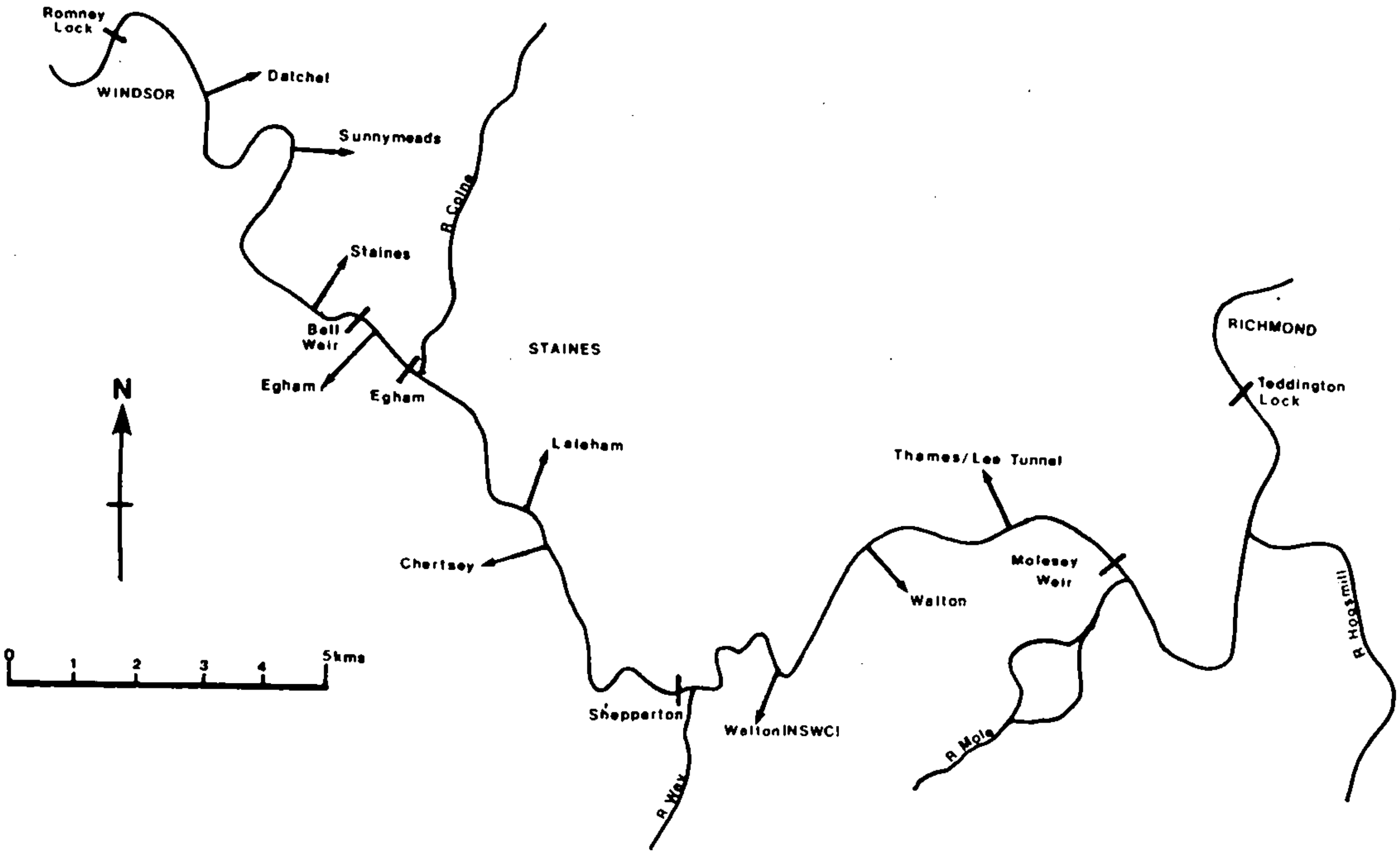


Figure 3.4. Location of reaches, abstraction sites and tributaries on the Lower Thames



Three Valleys Water Company all of whom have intakes along these lower reaches.

3.2.5. Table 3.1. provides a summary of the river reaches between the upstream site of Cricklade and the tidal limit at Teddington. Information is provided on major tributaries and effluents entering the river, principal abstractions, flow gauging stations, water quality monitoring sites and physical characteristics of the reaches. The principal reaches of concern in this study are shown in Figure 3.4 together with abstraction sites and tributaries.

3.3 A REVIEW OF EXISTING WATER QUALITY DATA

3.3.1. Routinely Monitored Data (1974-1982)

3.3.1.1. Water samples have been taken by Thames Water Authority on a regular basis since 1974 at a large number of sites on the Thames (see Table 3.1.) Sample frequency varies from weekly to monthly with samples being taken on average on a fortnightly basis. The sites of particular interest in this study are located in the lower reaches of the Thames which are significantly affected by abstractions. Dissolved oxygen, biochemical oxygen demand, ammonia and nitrate data for the period 1974-1982 at Teddington Weir are presented graphically in Appendix 3.1 and the data for BOD and DO are summarised in Table 3.2. for the sites of interest in the lower Thames.

3.3.1.2. The Table shows the extreme fluctuations in DO levels with concentrations reaching 18 or 19 mg l^{-1} at times and falling to 3.35 mg l^{-1} on one occasion. These fluctuations are typical for UK lowland river systems and are often associated with the growth and subsequent death of algae. The diurnal variations in DO levels will be discussed in the next section of the report. In general, mean DO levels are high, lying between 9 and 11 mg l^{-1} or close to the saturation level. Figure 3.5 shows the profile of the mean DO levels along the river system together with a measure of the variability given as plus and minus 1 standard deviation. The DO concentrations show very little variation along the river although the variability increases significantly in the reach above Molesey Weir. The data available for the Molesey site were collected only during 1976 a severe drought year, and it is not surprising that more variability is observed.

3.3.1.3. BOD concentration on the other hand as shown in Figure 3.6 attain higher levels in the reach below Romney, this behaviour is associated with the effluents entering this reach of the river. BOD thereafter declines

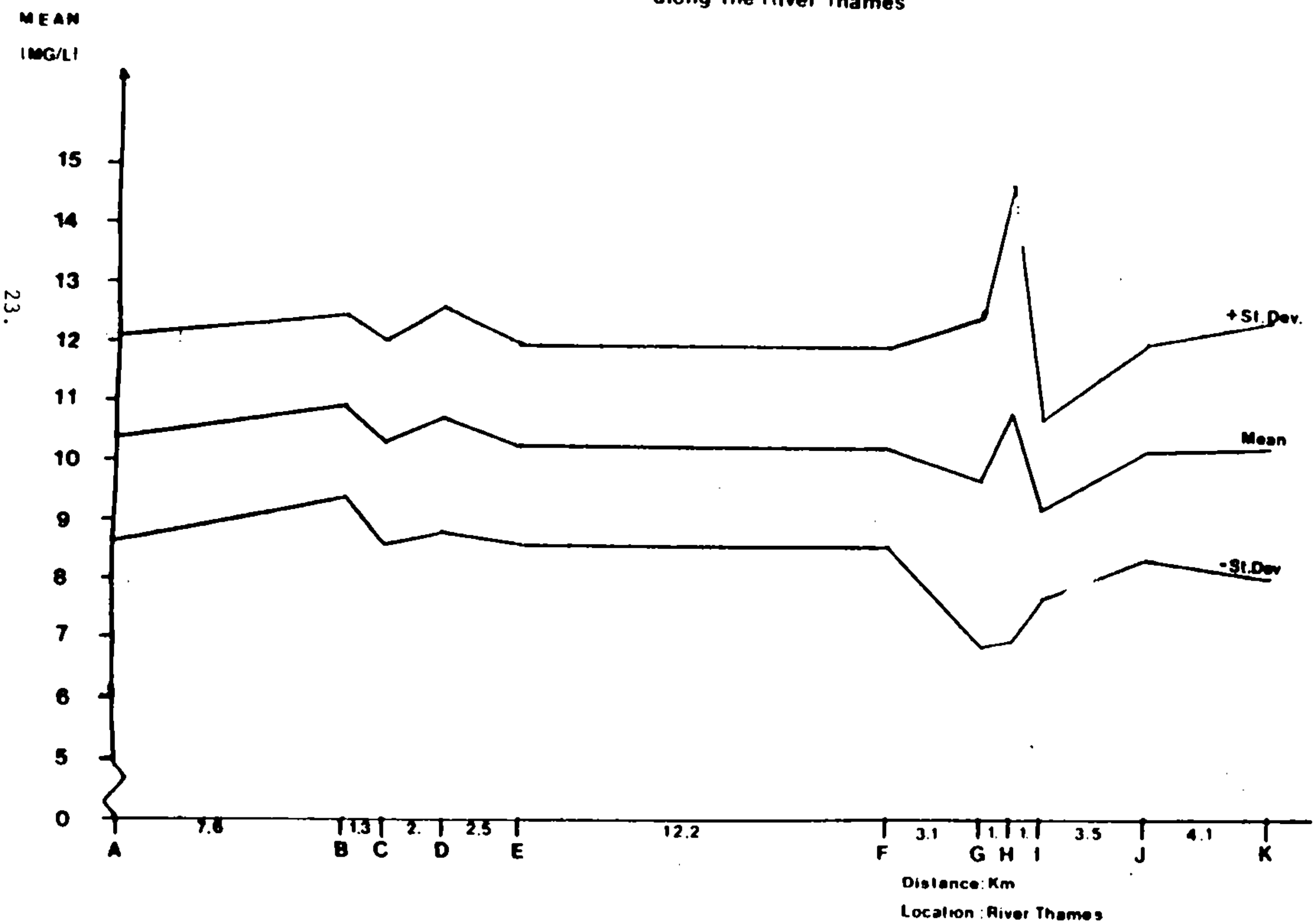
Reach	Rivers Entering Reach	Effluents	Abstractions	Main River Flow Gauging Station	Water Quality Monitoring	Length km	Mean Width m	Mean Depth m	Linearised Velocity $ms^{-1}\ cumec^{-1}$
1 Cricklade - Castle Eaton	Thames, Churn Ampney Brook and Ray	-	Castle Easton	Cricklade	H. Bridge	8.0	19.0	1.2	0.0414
2 Castle Eaton - Buscot	Coln, Leach, Cole	Highworth	Buscot	-	-	12.8	19.8	1.22	0.0413
3 Buscot - Rushey	-	Faringdon	Brampton	-	-	12.6	20.3	1.30	0.0397
4 Rushey - Pinkhill	Windrush	Carterton	Farmoor	Eynsham	Swinford	20.7	25.3	1.27	0.0402
5 Pinkhill - Kings Weir	Evenlode		Kings Weir	-	-	6.75	45.5	1.77	0.0237
6 Kings Weir - Osney	-	-	-	-	-	5.67	51.1	1.59	0.0199
7 Osney - Abingdon	Cherwell	Oxford	-	-	-	14.82	85.6	2.11	0.0120
8 Abingdon - Culham	Ock		Culham	-	Culham	4.14	85.7	2.01	0.0116
9 Culham - Days Weir	Ginge Brook, Moor Ditch	Abingdon, Didcot, Culham	Didcot Power station and return	Days	Days	9.32	64.5	1.59	0.0152
10 Days Weir - Benson	Thame	-	Warborough	-	-	6.38	94.0	2.29	0.0106
11 Benson - Whitchurch	-	Benson, Cholsey, Goring	Pangbourne	-	-	18.01	109.0	2.30	0.0092
12 Whitchurch - Caversham	Pang	Pangbourne	Playhatch	-	Caversham	10.76	128.8	2.25	0.0077
13 Caversham - Shiplake	Kennet	-	Sheeplands	-	Sonning	8.91	124.0	2.24	0.0081
14 Shiplake - Marlow	Loddon	Henley	-	-	-	18.99	159.4	2.52	0.0064
15 Marlow - Bray	Wye	Marlow, Little Marlow	-	Bray	-	11.66	130.1	1.83	0.0077
16 Bray - Romney	-	Slough Effluent		-	Datchet	8.85	105.3	1.77	0.0095
17 Romney - Bell	-	Windsor	Staines, Datchet and Sunnymeads	-	-	9.54	88.5	1.70	0.0115
18 Bell - Egham	-	Iver effluent	Egham	-	Egham	2.00	102.5	2.04	0.0097
19 Egham - Shepperton	Colne	-	Laleham and Chertsey	-	Laleham	8.46	104.5	1.92	0.0095
20 Shepperton - Molesey	Wey	Chertsey	Walton (NSWC) Walton (TWA) and Thames-Lee Tunnel	-	Walton	9.54	151	2.12	0.0067
21 Molesey - Teddington	Mole, Hogsmill	Surbiton, Hogsmill	-	Teddington	Teddington	7.74	185.9	2.44	0.0054

TABLE 3.1 Reach characteristics for the River Thames

TABLE 3.2 RIVER THAMES DATA SUMMARY

Location	BOD (mgl^{-1})				DO (mgl^{-1})			
River Thames	Min	Max	Mean	Standard Deviation	Min	Max	Mean	Standard Diviation
at M.W.D. Intake, Datchet	0.9	9.7	3.65	1.95	4.3	16.65	10.40	1.71
Above M.W.D. Intake, Bell Weir	1.2	13.3	4.25	2.96	8.6	15.35	10.92	1.55
Above N.S.W.C. Intake, Egham	0.6	14.2	3.89	2.18	5.0	19.9	10.31	1.72
800m below Colne	0.6	11.5	4.02	2.34	7.85	15.6	10.70	1.89
Above M.W.D. Intake, Littleton	1.2	9.7	3.55	1.86	4.8	15.8	10.28	1.69
At M.W.D. Intake, Walton	0.9	9.6	3.13	1.75	5.9	15.25	10.22	1.69
Below M.W.D. Intake, Hampton	1.9	7.1	3.5	1.61	6.6	16.1	9.65	2.81
At Garrick's Ait	1.8	7.2	3.87	1.71	7.45	18.25	10.78	3.83
Above Molesey Weir	1.0	9.8	4.09	2.45	7.7	12.85	9.15	1.5
Above Ravens Ait, Surbiton	0.8	9.2	3.57	1.8	5.4	15.55	10.14	1.81
At Teddington Weir	1.2	16.3	4.49	2.46	3.35	18.75	10.17	2.15

Figure 3.5 Mean and Standard Deviation of D.O. along the River Thames



KEY

A T.W.A Intake Datchet
8/1/74 - 29/12/82

B Above T.W.A Intake Bell Weir
24/1/74 - 29/12/82

C Above N.S.W.C. Intake Egham
2/1/74 - 29/12/82

D 800m below Colne
24/1/74 - 29/3/76

E Above T.W.A. Intake Littleton
8/1/74 - 29/12/82

F At T.W.A. Intake Walton
8/1/74 - 21/12/82

G Below T.W.A. Intake Hampton
26/7/76 - 1/11/76

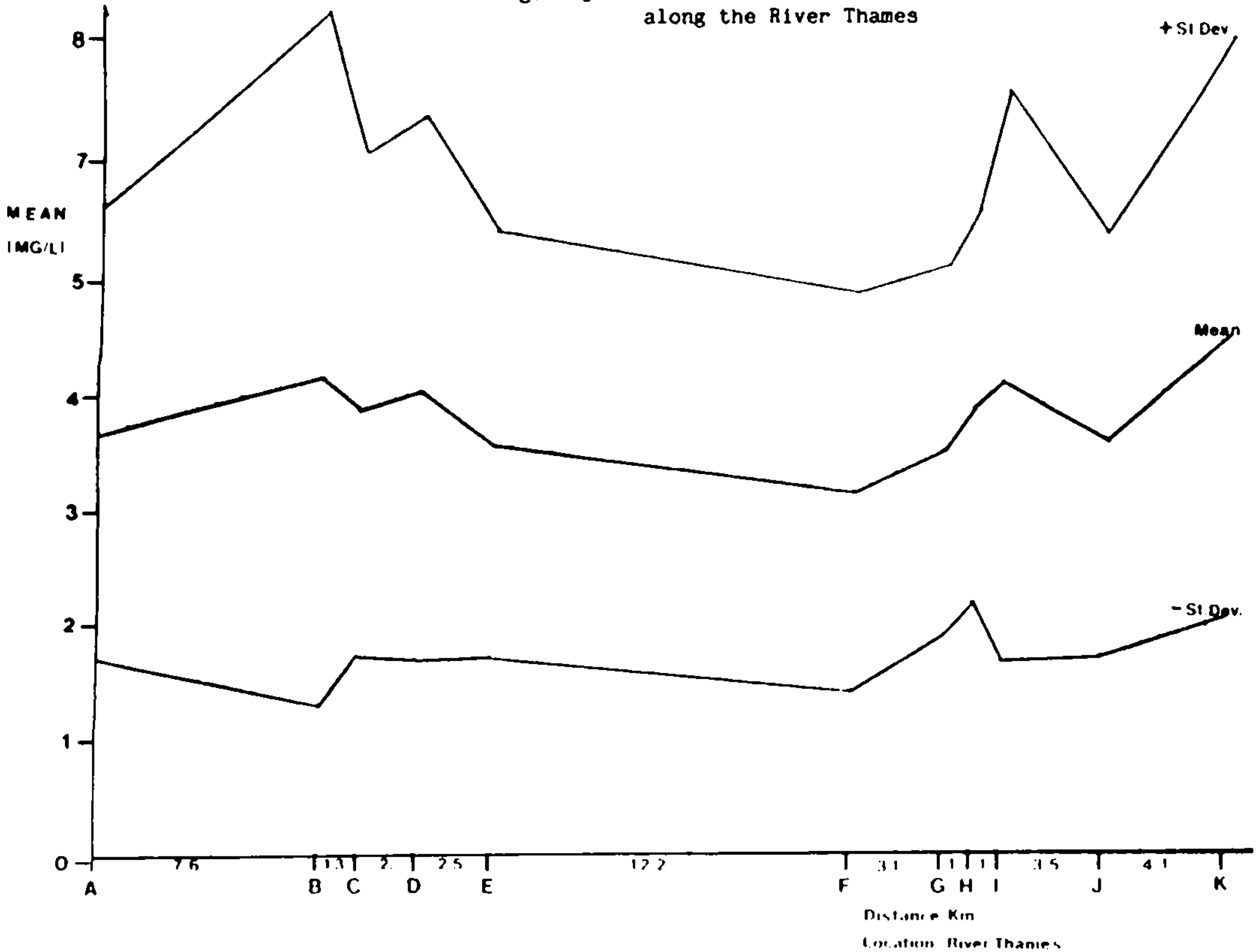
H At Garrick's AIT
26/7/76 - 5/9/77

I Above Molesey Weir
26/7/76 - 1/11/76

J Above Ravens AIT Surbiton
5/4/76 - 21/12/82

K At Teddington Weir
2/1/74 - 29/11/82

Figure 3.6 Mean and standard deviation of B.O.D. along the River Thames



KEY

A T.W.A Intake Datchet
8/1/74 - 29/12/82

B Above T.W.A Intake Bell Weir
24/1/74 - 29/12/82

C Above N.S.W.C. Intake Egham
2/1/74 - 29/12/82

D 800m below Colne
24/1/74 - 29/3/76

E Above T.W.A. Intake Littleton
8/1/74 - 29/12/82

F At T.W.A. Intake Walton
8/1/74 - 21/12/82

G Below T.W.A. Intake Hampton
26/7/76 - 1/11/76

H At Garrick's AIT
26/7/76 - 5/9/77

I Above Molesey Weir
25/7/76 - 1/11/76

J Above Ravens AIT Surbiton
5/4/76 - 21/12/82

K At Teddington Weir
2/1/74 - 29/11/82

until Molesey Weir where additional effluents increase BOD concentrations. The BOD decline is probably due to sedimentation and decay of organic material.

3.3.1.4. Ammonia concentrations are generally low in the Thames being less than 0.5 mg l^{-1} above Molesey Weir. However discharges from effluent treatment works into the Mole and the Hogsmill Rivers can have a significant effect and back pumping over Teddington Weir in 1976 (see Appendix 3.1.) produced ammonia concentrations of 6 mg l^{-1} on occasions. Recent improvements to the treatment plants have improved this situation and predicted peak concentrations under low flow conditions are given later in the report.

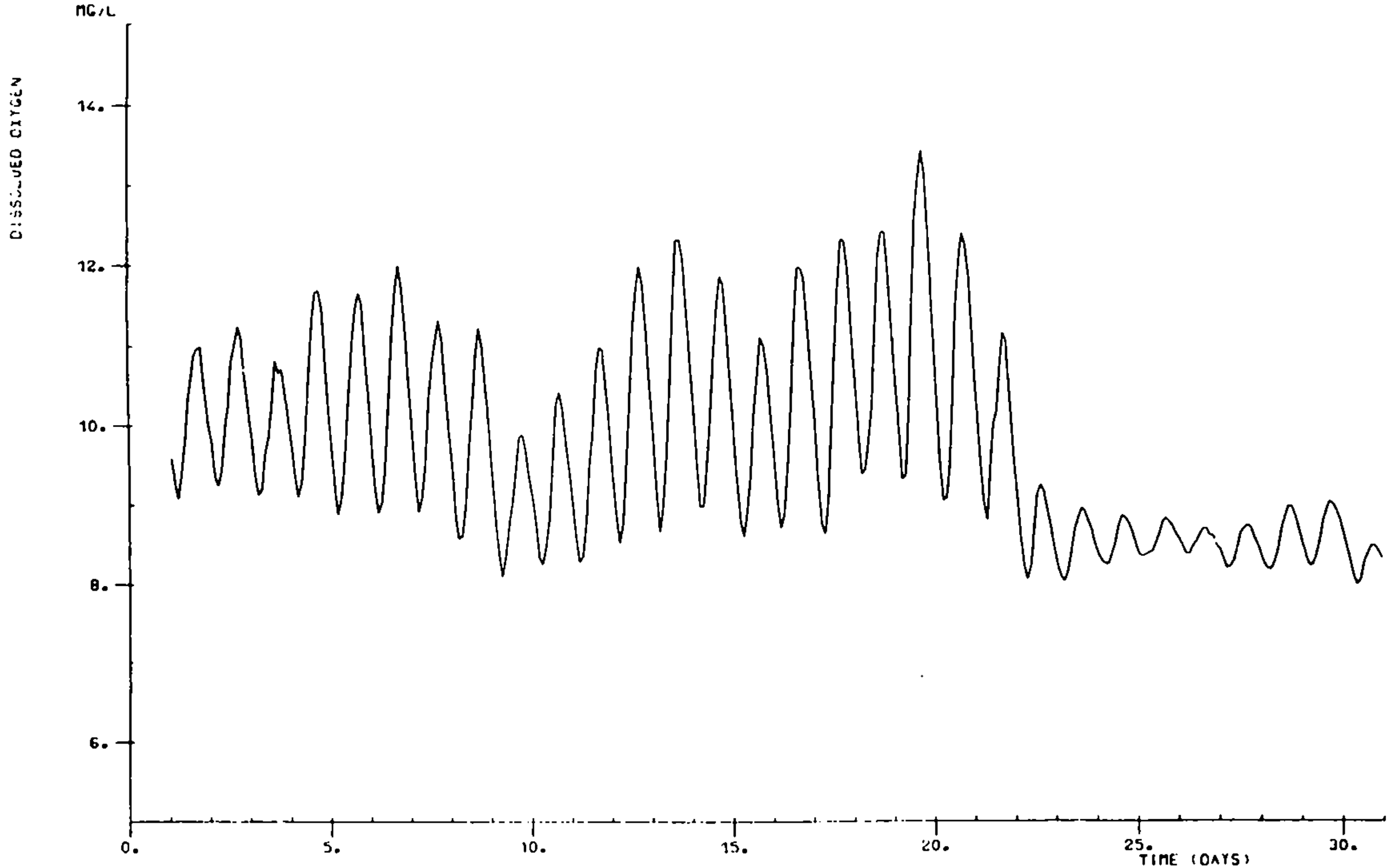
3.3.2. Continuously Monitored Data

3.3.2.1. Dissolved Oxygen concentrations have been monitored by Thames Water Authority on a continuous basis using a probe located at Romney Lock. A plot of the DO concentrations over the month of July 1981 is shown in Figure 3.7 and represents a typical summer pattern in the river. A clear variation through the day is observed with oxygen levels increasing during the morning and early afternoon and decreasing at night, falling to a minimum at about dawn. These fluctuations are caused by oxygen being produced by photosynthetic activity of algae and plants and oxygen being consumed by respiration of plants, algae and the benthos. Since photosynthesis only takes place during daylight hours a diurnal pattern is obtained and this pattern will also be modified if the intensity of solar radiation varies. For example, on day 22 onwards solar radiation levels are relatively low because of cloud cover and this dampens the daily oscillation. Moreover, many factors can affect the extent of the daily oscillation such as the growth rate of algae, the species of algae and the relative importance of mud or benthic respiration.

3.3.2.2. A major problem in obtaining representative samples for a river is to know when to actually take a sample. For example, on day 19 a sample taken at 6.00 would have a DO of approximately 9 mg l^{-1} whereas a sample taken at 2.00 would have a DO of 13 mg l^{-1}. Thus interpreting spot daily samples can be very misleading.

3.3.2.3. In the study the continuous DO data have been used to investigate the relationship between oxygen production, algal levels and solar radiation and this will be described later in the report. It is important however to bear in mind the daily oscillation of DO since it is the minimum DO levels which create particular problems for fish and the general ecology of the river. A number of other months of continuous DO data and solar radiation information is given in Appendix 3.2.

Figure 3.7 ROMNEY LOCK AUTOMATIC QUALITY MONITORING STATION
JULY 1981



Continuous DO recording at Romney for July, 1981

3.4 THE RIVER THAMES FLOW MODEL

3.4.1. In order to model any water quality variable it is necessary to first simulate streamflow in all key reaches of the river. A streamflow model for all the reaches shown in Table 3.1 has been developed (Whitehead and Williams, 1982) In this model each reach is characterised by a number of cells and the model for flow variations in each cell is based on an analogy with the lumped parameter equations for the variations in concentration of a conservative pollutant under the assumption of uniform mixing over the cell (Whitehead et al 1979) The model may be viewed in hydrological flow routing terms as one in which the relationship between inflow I and outflow Q and storage S in each cell is represented by the differential equation:

$$\frac{dS}{dt} = (I - Q) \tag{3.1.}$$

where $S = \tau Q$ and where τ is a travel time or residence time parameter. It is necessary to allow the residence time to vary with flow and in order to achieve this τ is expressed as

$$\tau = \frac{L}{un} \tag{3.2.}$$

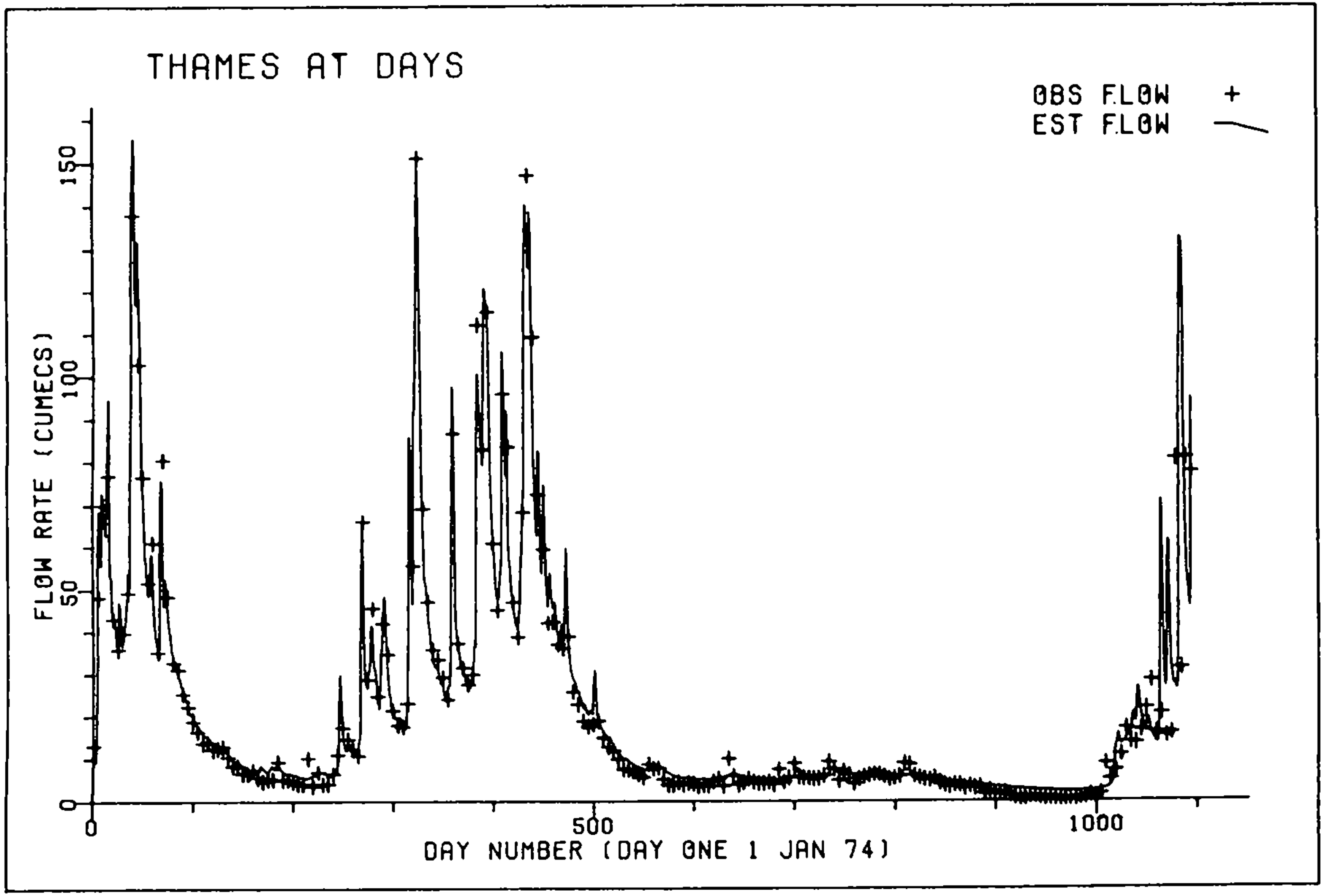
where n is the number of compartments in the reach, L is the reach length, and u is the mean flow velocity in the reach, which is related to discharge through

$$u = a\, Q_m^{\,b} \tag{3.3.}$$

Where Qm is the mean reach flow and where a and b are coefficients to be estimated. The value of n controls the amount of dispersion in a reach, and can be determined through tracer experiments which are also used as a basis for estimating the coefficients a and b. In the absence of information from tracer experiments, values of n, a and b can be obtained through calibration on an observed record of down-stream flow.

3.4.2. Given information on upstream and tributary inputs, the flow routing model can be used to derive simulations of downstream flow by solving the differential equation (3.1.) with τ defined through equations (3.2.) and (3.3.) The equation is solved using a numerical integration technique which contains an automatic adjustment to the integration step length. This is particularly useful since during periods of low flow and high residence times, the integration step length can be increased

Figure 3.8. Observed and simulated daily flows at Days Weir for 1974,75 and 76



thereby saving computer time. Under high flow conditions, however, residence times are reduced and in order to solve the equation to the same accuracy, it is necessary to reduce the integration step length. Since this is achieved automatically, there are relatively few numerical integration problems. Figure 3.8 shows simulated flows compared to observations for the years 1974, 1975 and 1976 allowing for all the inputs shown in Table 3.1.; 94% of the variance is explained and the model provides a sound basis for subsequent water quality studies.

3.5 A REVIEW OF DO-BOD MODELS FOR RIVER SYSTEMS

3.5.1. In order to support water-borne life, water must contain dissolved oxygen in sufficient quantities and also be largely free from toxic and harmful compounds. The principal aim of most oxygen balance studies is to assess the impact of effluent discharges on water quality and to quantify the extent of biological self-purification.

3.5.2. Biological self-purification is the process by which organic wastes are broken down by the respiration of micro-organisms into stable end products. It is a biochemical oxidation process through which organic wastes are consumed leaving behind end products such as carbon dioxide, water phosphates and nitrates. The water is 'purified' in the sense that the concentration of waste material has been reduced. Organic materials which can be broken down (i.e. are biodegradable) include natural materials such as simple sugars, starch, fats, proteins as well as more complex natural or synthetic compounds which are found in sewage or other wastes.

3.5.3. The interaction between BOD and DO downstream of a discharge may be illustrated in a simple manner as shown in Figure 3.9 A high initial BOD exerts a large demand for oxygen and as the instream organic matter decays the oxygen level becomes depressed. The river, however, has the capacity to recover naturally from this situation by the transfer of oxygen from the atmosphere to the water. This process of re-aeration is enhanced by low temperatures and a large degree of turbulence in the river but is primarily controlled by difference between the saturated oxygen level and the actual river oxygen level. At the minimum of the oxygen sag curve a high level of re-aeration occurs and as the BOD level declines the DO level increases. The minimum is influenced by various factors including the type of effluent and the rate of discharge, a higher organic load producing a lower minimum. In addition, at high river flows the location of the minimum shifts downstream and an important aspect of any water quality study is therefore a satisfactory streamflow model.

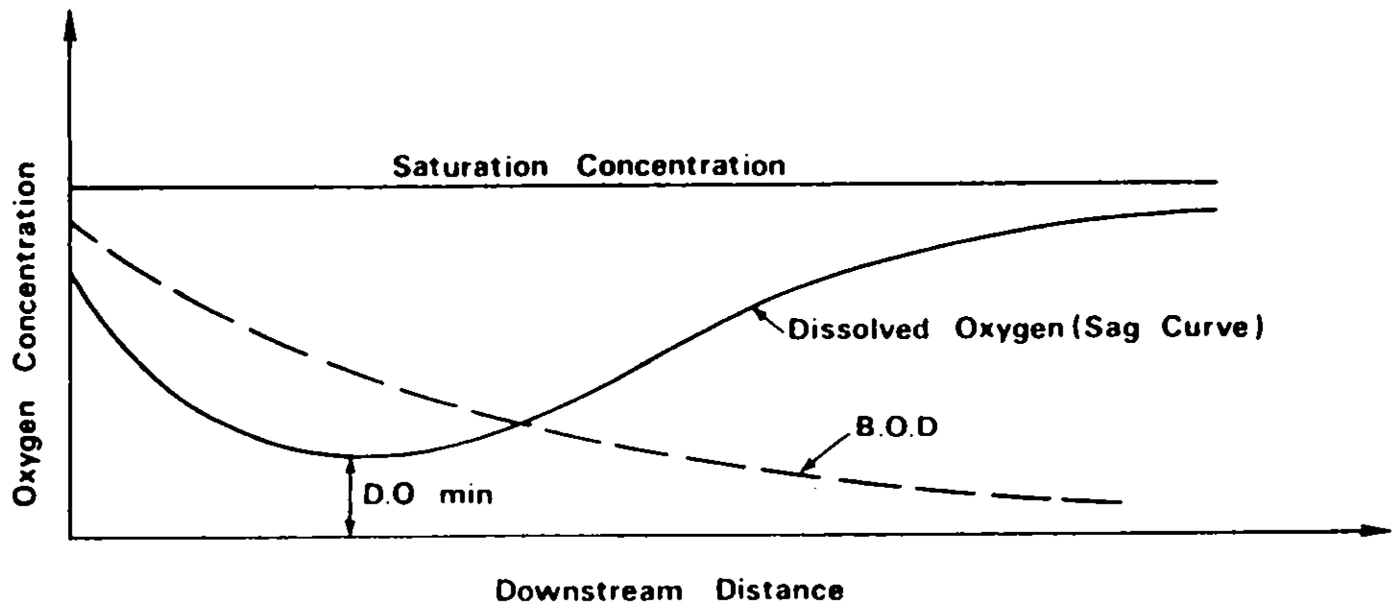


Figure 3.9 The dissolved oxygen sag curve

3.5.4. Research on the modelling of DO-BOD interactions in a river system has been dominated by the classical model of Streeter and Phelps (1925) where the equations take the form:-

$$\text{DO:} \quad \frac{dD}{dt} = k_1 L + k_2 D \qquad (3.4.)$$

$$\text{BOD:} \quad \frac{dL}{dt} = k_1 L \qquad (3.5.)$$

where L represents the BOD concentration mgl^{-1}; D represents the DO deficit, where the deficit is determined as the difference between the DO saturation concentration and the DO concentration in the stream, mgl^{-1}.

k_1 and k_2 are rate constants describing respectively the BOD decay process and the re-aeration process

3.5.5. It should be noted that this system of two coupled differential equations does not describe the transient behaviour of the DO at a fixed point in the stream but represents an element of fluid moving downstream at the streamflow velocity; in other words it describes a profile or steady state solution along the stream length. In addition, the equations do not reflect the effects of the aquatic environment on the DO distribution. The magnitude of these effects vary from stream to stream according to the physical, chemical and biological conditions and a number of investigations, such as those of Dobbins (1964), Owens (1969), Camp (1965), O'Connor (1967) and Thomann (1972) have extended the basic model to include such phenomena as,

i) the removal of BOD by sedimentation and adsorption;

ii) the addition of BOD along the reach by the scour of bottom deposits or by diffusion of partly decomposed organic matter from the benthal (mud) layer into the water above;

iii) The removal of oxygen from the water by diffusion into the benthal layer to satisfy oxygen demand in the aerobic zone;

iv) the addition of oxygen by photosynthetic action of algae and fixed plants;

v) the removal of oxygen by respiration of algae and fixed plants.

3.5.6. All of these processes affect the distribution of DO and may therefore need to be considered in a water quality study. A comprehensive review of the mechanistic terms and model variations is given by Beck (1978) and, as emphasised by Beck, these modifications have in the main been applied to the Streeter-Phelps equations; equations that provide a steady state profile along the stream length.

3.5.7. The steady state model is not, however, particularly well suited to the Thames DO-BOD study. This is because the river is rarely, if ever, in steady state. Flows and, thereby, river velocities change significantly from day to day, as do a number of other variables, such as temperature and solar radiation levels. A dynamic model is required that is capable of accepting time varying inputs and operating upon them to give time varying output responses. Only by comparing the synamic responses of the DO and BOD behaviour can a realistic assessment of the proposed operating strategies be made.

3.5.8. One possible dynamic characterisation of water quality that appears, on a priori considerations, well suited to this kind of problem, consists of a lumped parameter differential equation model obtained from a mass balance over each reach of the river. The reach is assumed to be uniform throughout and analogous in chemical engineering terms to a continuous stirred tank reactor (CSTR) in which the output concentrations are equal to those in reach. This approach has been applied by Beck (1978) over a single reach of the River Cam in South East England and by Whitehead et al (1981) over several reaches in the Bedford Ouse. The dynamic model resulting from the mass balance is of the following form:-

$$\text{DO:} \quad \frac{dx_1(t)}{dt} = -\left(k_1 + \frac{Q}{V_m}\right)x_1(t) - k_2x_2(t) + \frac{Q}{V_m}C(t) + k_1C_s + D_B \tag{3.6.}$$

$$\text{BOD:} \quad \frac{dx_2(t)}{dt} = -\left(k_2 + \frac{Q}{V_m}\right)x_2(t) + \frac{Q}{V_m}L(t) + L_A \tag{3.7.}$$

where:-

x_1 is the output (i.e. downstream) DO in mgl^{-1}

x_2 is the output BOD in mgl^{-1}

L is the input (i.e. upstream) BOD in mgl^{-1}

C is the input DO in mgl^{-1}

C_s is the saturation concentration of DO in mgl^{-1}

Q is the volumetric flow rate in m^3 $days^{-1}$

V_m is the mean volume of water held in the reach in m^3

k_1 is the reaeration rate constant $days^{-1}$

k_2 is the BOD decay rate constant $days^{-1}$

L_A is the mean rate of addition of BOD to the reach by local run-off in $mgl^{-1}day^{-1}$

D_B is the net rate of addition of DO to the reach by the combined effects of photosyntheses, respiration and mud deposits in $mgl^{-1}day^{-1}$

t is time in days

The saturation concentration for DO is determined as,

$$C_s = 14.652 - 0.41022T + 0.0079910\ T^2 - 0.000077774\ T^3$$

where T is the stream temperature °C.

3.5.9. A common problem with water quality models is to determine parameter values such as the BOD decay coefficient and reaeration rate coefficients. The standard approach is to select parameter values from the literature or from experimental measurements. Knowles and Wakeford (1978) describe a number of relationships and parameter values which can be used in situations where little information is available and this approach has been applied by Casapieri _et al_ (1978) in a study of the Blackwater Catchment of the Thames.

3.5.10. A more sophisticated approach was developed by Beck and Young (1976) in which the parameters of a dynamic water quality model were estimated directly from field data using the extended Kalman filter. The EKF is essentially a statistical technique which accounts for measurements errors and system noise both of which are highly significant in water quality studies. Whitehead (1978, 80, 81) applied the EKF technique and the instrumental variable (IV) technique to estimate water quality parameters in the dynamic models developed for the Bedford Ouse. However, a requirement of these techniques is that an extensive record of daily or continuous data is available. In the case of the Cam study, 80 days of data were available and in the case of the Bedford Ouse over two years of daily data were analysed. In both studies the estimation techniques proved to be extremely useful in identifying model structures and determining model parameters.

3.5.11. In the River Thames study daily data have not been available for such sophisticated analysis and therefore it has been necessary to make use of existing data and information on processes and parameter values. However, rather than develop a steady state model, the nature of the study demands a dynamic model so that the effect of changes in flow, temperature, solar radiation and algal levels can be incorporated. Similar dynamic models for nitrate and algal growth and transport in the Thames have already been developed using routinely collected water authority data (see Whitehead _et al_ 1982, 84) and the approach is used in the current study of DO-BOD variations.

3.6. THE THAMES DO-BOD MODEL

3.6.1. The dynamic DO-BOD model for the River Thames is based on equations (3.6.) and (3.7.) described in the previous section but with modifications to account for additional sources and sinks. In particular, the mass balance model includes the effects of tributaries and effluents entering the system and losses of water via abstractions. Processes describing the addition of DO and BOD via photosynthetic oxygen production and phytoplankton death and decay are included together with losses of DO and BOD via respiration of mud or benthic deposits, respiration of phytoplankton and sedimentation processes. As in equations (3.6) and (3.7) reaeration and BOD decay processes are incorporated in the model. The following equations apply for each reach.

$$\text{DO:}\quad \frac{dx_1(t)}{dt} = \frac{Q_i(t)}{V_m} C_i(t) + \frac{Q_e(t)}{V_m} C_e(t) + \frac{Q_t(t)}{V_m} C_t(t) - \frac{Q_a(t)}{V_m} x_1(t) - \frac{Q_o(t)}{V_m} x_1(t) - k_1 x_2(t) + k_2\,(C_s(t) - x_1(t)) + P - R - M \quad (3.8.)$$

$$\text{BOD:}\quad \frac{dx_2(t)}{dt} = \frac{Q_i(t)}{V_m} L_i(t) + \frac{Q_e(t)}{V_m} L_e(t) + \frac{Q_t(t)}{V_m} L_t(t) - \frac{Q_a(t)}{V_m} x_2(t) - \frac{Q_o(t)}{V_m} x_2(t) - (k_1 + k_3) x_2(t) + A \quad (3.9.)$$

3.6.2. Here subscripts i, e, t, a, o refer to input (upstream), effluent, tributary abstraction and output (downstream) respectively for the reach.

Parameters k_1, k_2 and k_3 are respectively BOD decay rate, reaeration rate and sedimentation rate parameters (day^{-1}) defined as:-

$$k_1 = 0.2 * 1.047^{(T-20)} \text{ days}^{-1}$$ (Wakeford and Knowles (1978))

where T is temperature °C and $1.047^{(T-20)}$ is a temperature correction term.

$$k_2 = 5.316 \frac{u^{0.67}}{d^{1.85}} \; 1.024^{(T-20)} \text{day}^{-1}$$ (Owens _et al_ (1969))

where u is stream velocity m sec^{-1} and d is reach depth, m

$$k_3 = 0.1 \text{ days}^{-1}$$

C_s in equation (3.8) is the saturation concentration for DO defined as:-

$$C_s = 14.652 - 0.41022T + 0.0079910\,T^2 - 0.000077774\,T^3$$

3.6.3. P in equation(3.8) refers to the addition of oxygen by photosynthesis of plants and algae. In the Thames the river does not support a large biomass of rooted plants because of both depth and high water turbidity. However, algal blooms in spring, summer and autumn are a common occurrance and these have a considerable effect on DO levels, as discussed in section 3.2. Modelling the photosynthetic production of oxygen by algae has been undertaken by many researchers. Steel (1978) describes in detail the processes controlling oxygen production in reservoirs and similar processes are known to occur in rivers. Owens _et al_ (1969) considered a simplified model in which oxygen production is related to light intensity and plant biomass or algal levels. Whitehead _et al_ (1981) used a modified version of the Owens model and estimated the relevant parameters for the Bedford Ouse. A similar approach has been adopted for the Thames and the following relationship developed

$$P = \frac{8.6}{10^5} Cl_a \; I^{0.79} \; 1.08^{(T-20)}$$

Here Cl_a is the chlorophyll-a concentration mg m^{-3} I is the solar radiation level watt hours m^{-2} per day. The coefficient 8.6 was determined from a linear regression analysis using as variates the observed oxygen production, obtained from the continuous data shown in Appendix 2 and $\frac{1}{10^5} Cl_a \; I^{0.79} \; 1.08^{(T-20)}$ obtained for the same period.

3.6.4. R in equation (3.8.) refers to the loss of oxygen via algal respiration. Kowalczewski and Lack (1971) developed a relationship between algal concentration measured as chlorophyll a and respiration rate for the River Thames, where

$$R = (0.14 + 0.013\ Cl_a)\ 1.08^{(T-20)}$$

and this relationship has been incorporated into the model.

3.6.5. M in equation (3.8.) refers to the respiration of the river bed or mud. There has been considerable research into this process (Edwards and Rolley, 1965) and the following equation has been used,

$$M = \frac{k_4}{d}\ x_1^{0.45}\ 1.08^{(T-20)}$$

where x_1 is the DO concentration mgl^{-1}, d is depth, m, and k_4 is a parameter to be determined. The original work of Edward and Rolley was conducted on the highly polluted muds of the River Ivel and later studies by Rolley and Edwards (1967) showed that the parameter k_4 varied considerably from river to river. In the Thames study a value for k_4 of 0.15 $days^{-1}$ was found to provide the best fit to the observed DO data.

3.6.6. Finally A in equation (3.9) refers to the conversion of algae to decaying organic matter. In previous algal modelling studies on the Thames (see Whitehead 1984) the concentration of dead algae is assumed proportional to the concentration of live algae. Thus A can be expressed mathematically as

$$A = k_5\ Cl_a\ .1.047^{(T-20)}$$

Where Cl_a is the chlorophyll a concentration $mg.m^{-3}$ and k_5 is a parameter. From simulation studies on the Thames k_5 was found to be 0.01

3.6.7. The complex interactions between DO, BOD, solar radiation, chlorophyll a, flow and temperature are thus incorporated into the model. The model has been set up to simulate the lowest five reaches on the Thames, shown in Table 3.1. These reaches are most significant in terms of DO and BOD and are affected by the different flow strategies proposed by Thames Water Authority Temperature has also been simulated since this is a key variable controlling rate processes.

The model has been run on a daily timescale using flow and quality data for 1974, 75 and 76. Figures 3.10, 3.11 and 3.12 show model simulations compared with the observed data at Teddington. In general, there is a reasonable fit to the data, although the scatter of the DO and BOD data in the summer months, particularly in 1976, limits the degree to which further model calibration is possible. The scatter reflects the difficulties of sampling and analysis and, in particular, the problems created by variations in the time of sampling.

Figure 3.10 THAMES AT TEDDINGTON
CALIBRATION RUN 1974 - 1976

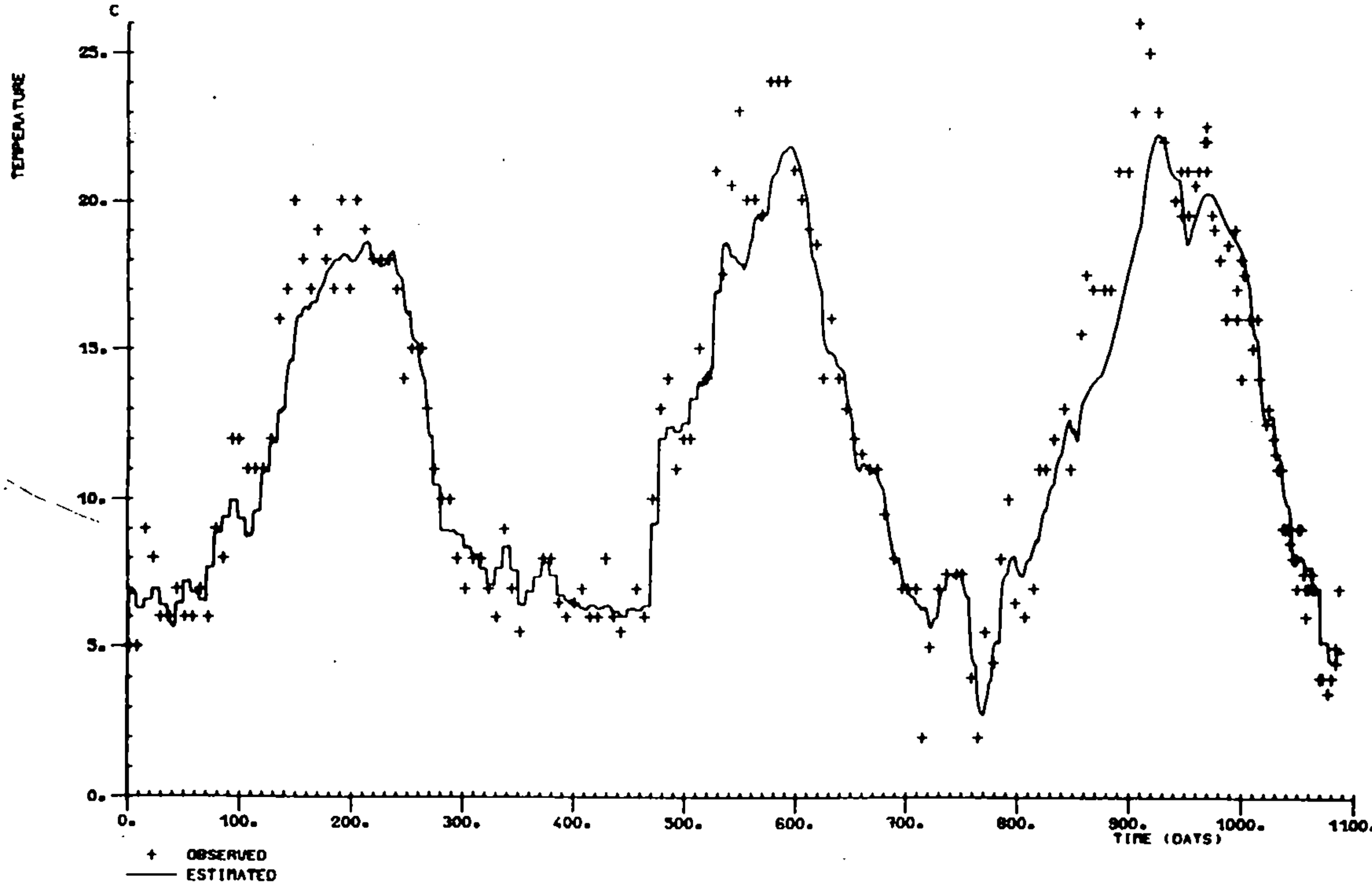


Figure 3.11 THAMES AT TEDDINGTON
CALIBRATION RUN 1974 - 1976

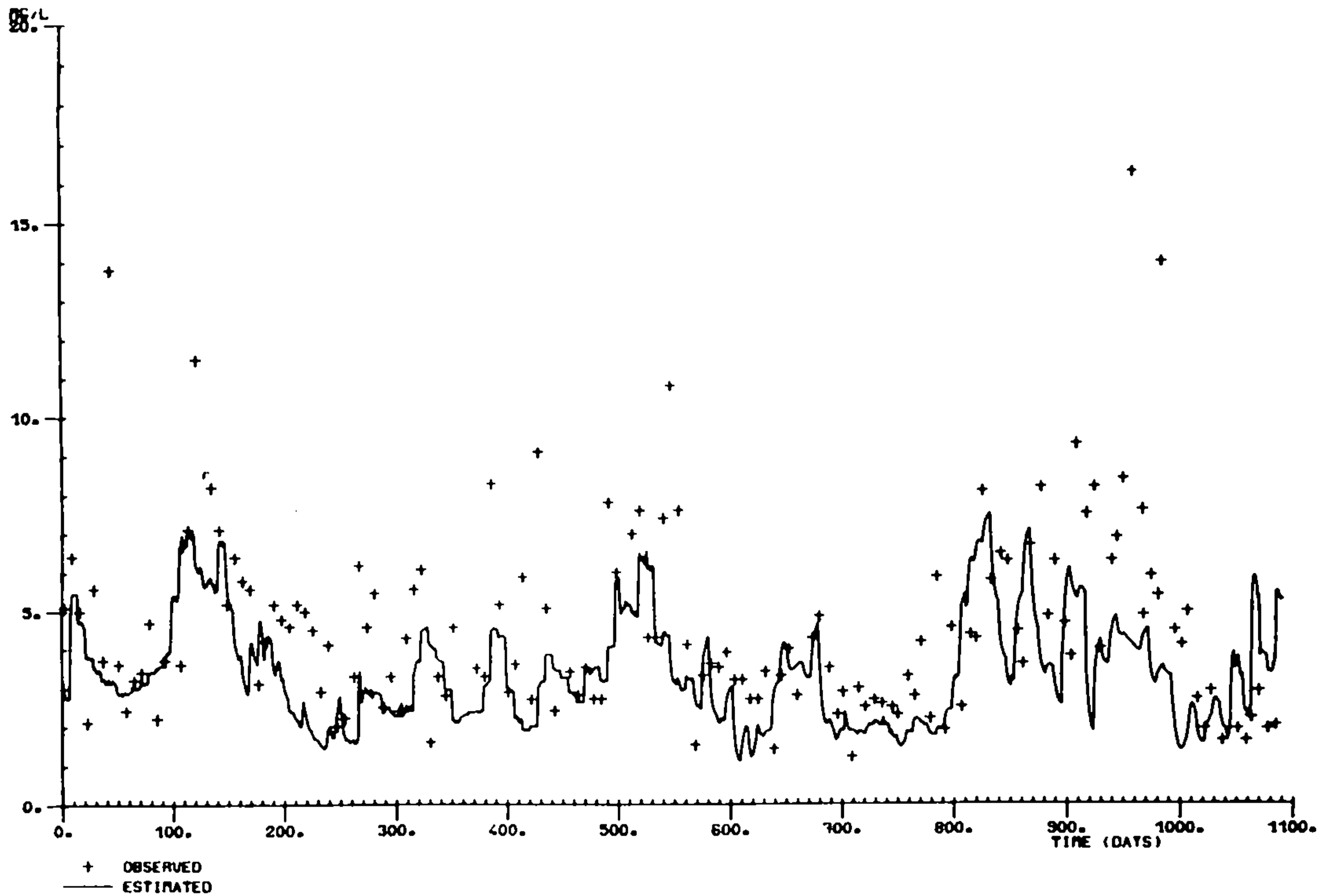
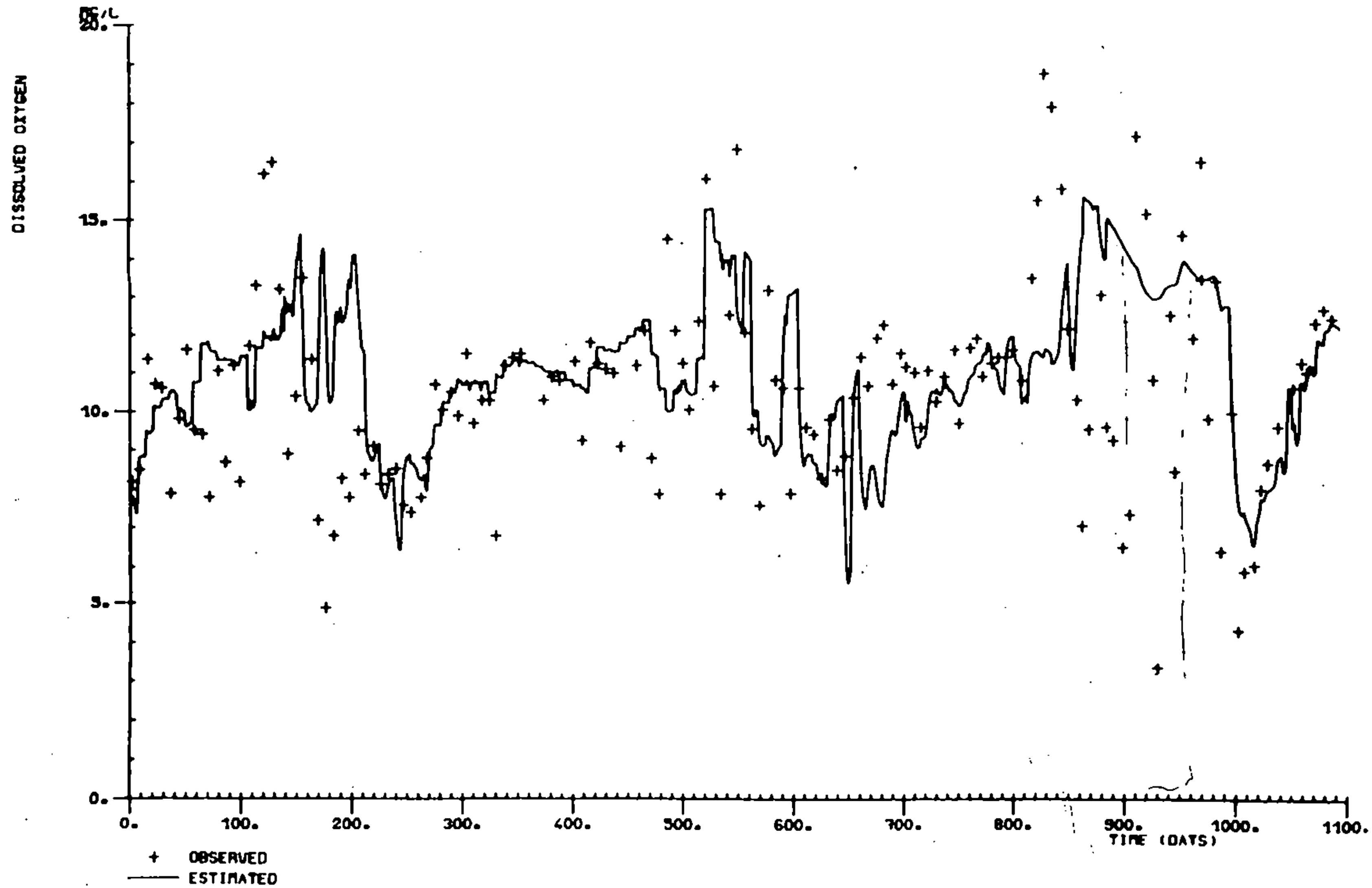


Figure 3.12 THAMES AT TEDDINGTON
CALIBRATION RUN 1974 - 1976



3.6.8. The weekly chlorophyll a data at Teddington used in the simulation is plotted in Appendix 3.3 together with algal data for other sites on the River Thames for 1974,75 and 76.

3.7 EFFECTS OF FLOW CHANGES ON RIVER WATER QUALITY

3.7.1. Flow Changes

3.7.1.1. The hydrological effects of the proposed operating strategy have been investigated using the flow model for the five reaches of the river between Romney and Teddington. Daily flow inputs and abstractions for drought and non-drought years have been generated by the Thames Water Authority water resource model assuming 1984 demand conditions. The water resources model has been run twice, firstly to reproduce the current operating strategy given "The Chart" for all years since 1920 and secondly, assuming the proposed operating strategy given a MRB policy. The MRB policy assumes that the pumps will be operated at their maximum capacity whenever possible. Figures 3.13 to 3.20 show the two contrasting flow strategies for the Teddington Molesey reaches for the drought periods of 1975-1976 and 1944-1945, and the non-drought periods of 1952-53 and 1957-58.

3.7.1.2. In Figure 3.13 the continuous line shows the flows at Teddington given the current "Chart" rules and the dotted line represents the flows given the proposed MRB strategy. Only the flows below 23 cumecs are plotted. In the summer of 1975 (days 210-270) the MRB strategy would reduce the flows at Teddington considerably from an average of 9 cumecs to below 3 cumecs. However in the summer of 76 the flows under proposed and current rules are similar but at very low levels. If anything the flows under current rules are worse than those under the MRB rules, particularly in the critical period between days 600-620 and this is due to the fact that back pumping was necessary at Molesey and Teddington Weirs to maintain sufficient water for abstraction. The major abstractions in the Lower Thames are, in fact, located upstream of Molesey Weir and the flow in this reach is therefore particularly important. As shown in Figure 3.14 the flows over Molesey Weir are lower than those at Teddington. This is because under normal conditions the low flow over Molesey Weir is supplemented by the River Mole and the River Hogsmill which enter the river downstream of the weir.

3.7.1.3. This pattern of behaviour is repeated in all of the drought and non-drought years shown in Figures 3.15 - 3.20 In the drought of 1944, 45 (Figures 3.15 and 3.16) the lowest flows would be avoided by the

Figure 3.13 THAMES AT TEDDINGTON
1975 - 1976

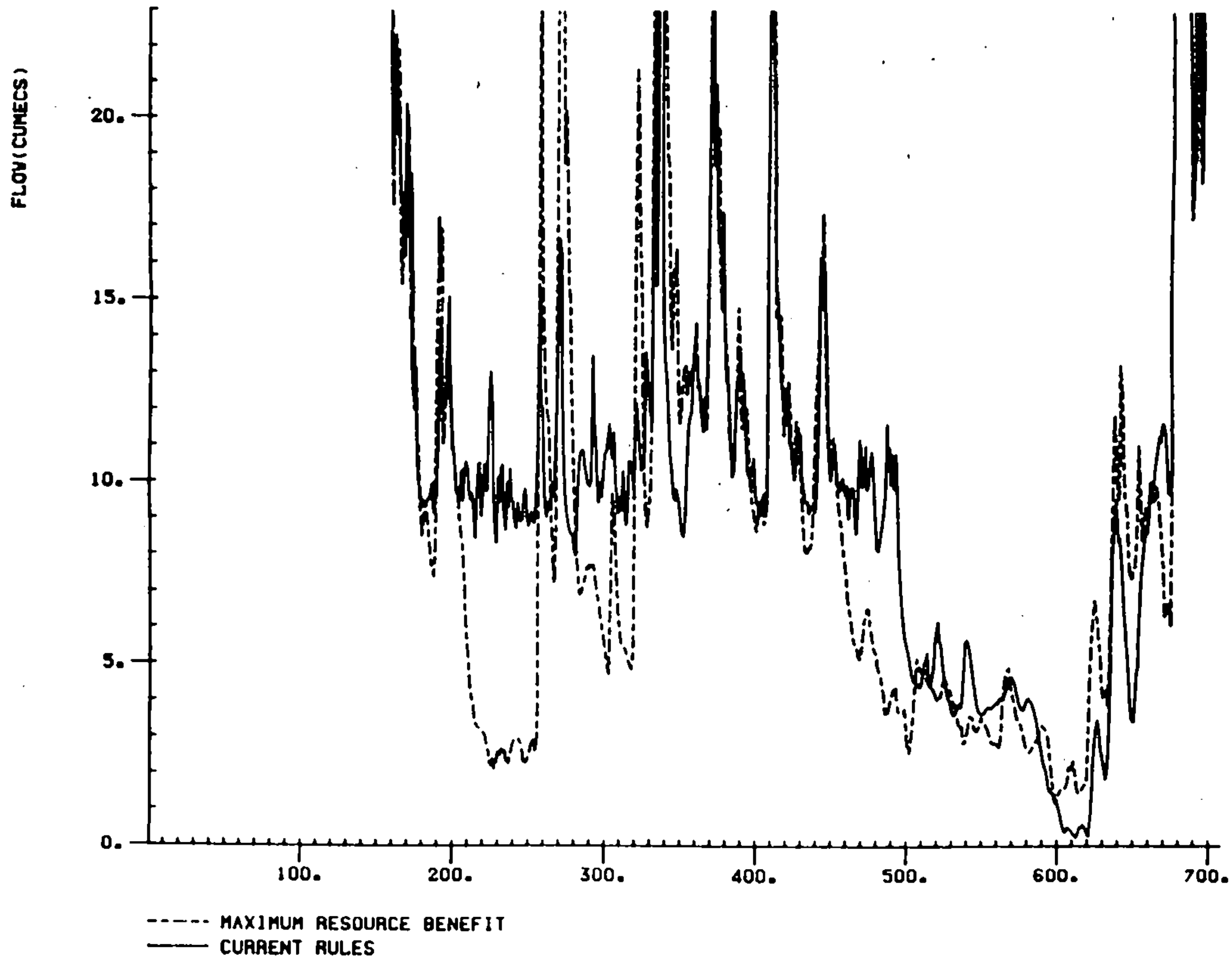


Figure 3.14 THAMES AT MOLESEY
1975 - 1976

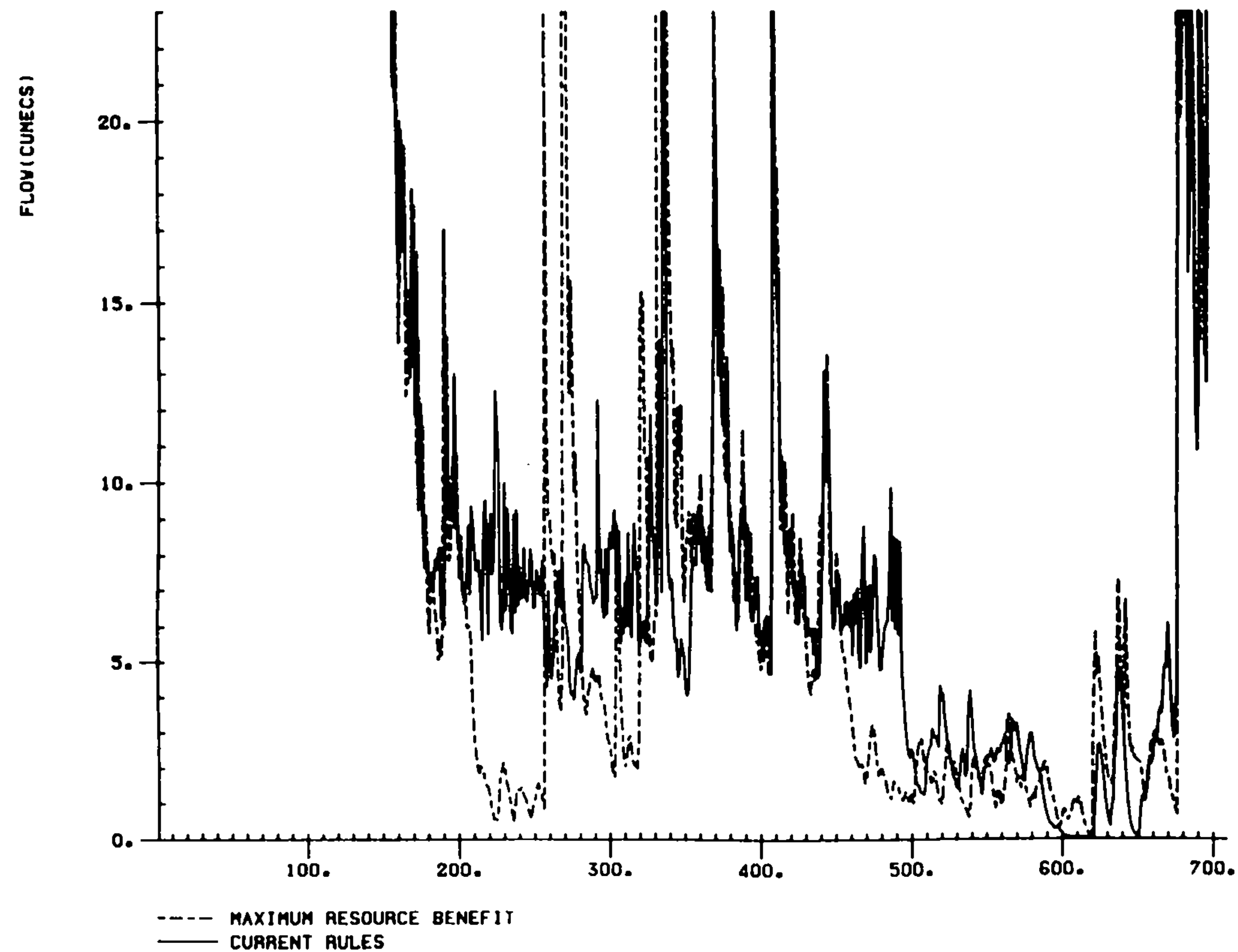


Figure 3.15 THAMES AT TEDDINGTON
1944 - 1945

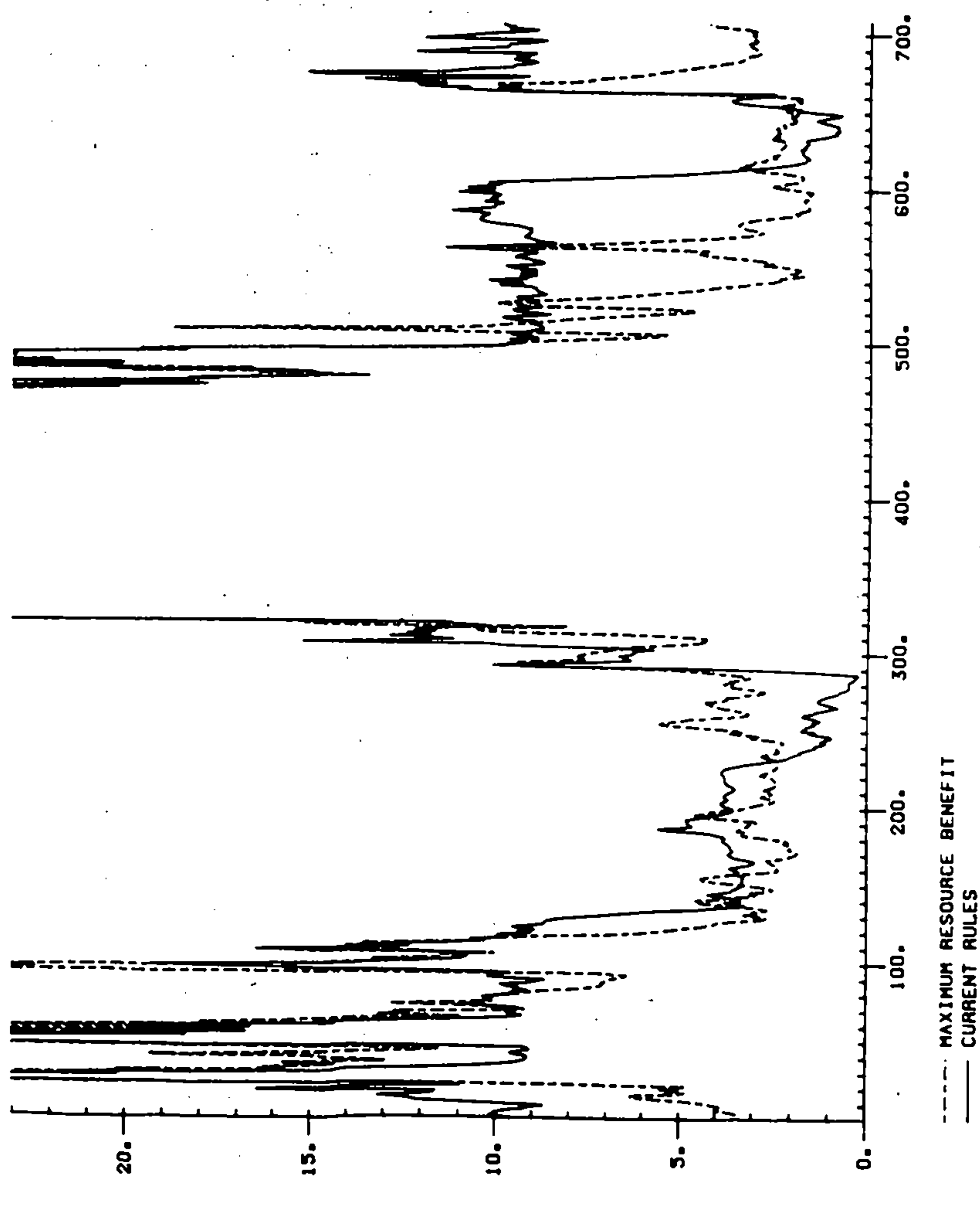


Figure 3.16 THAMES AT MOLESEY
1944 - 1945

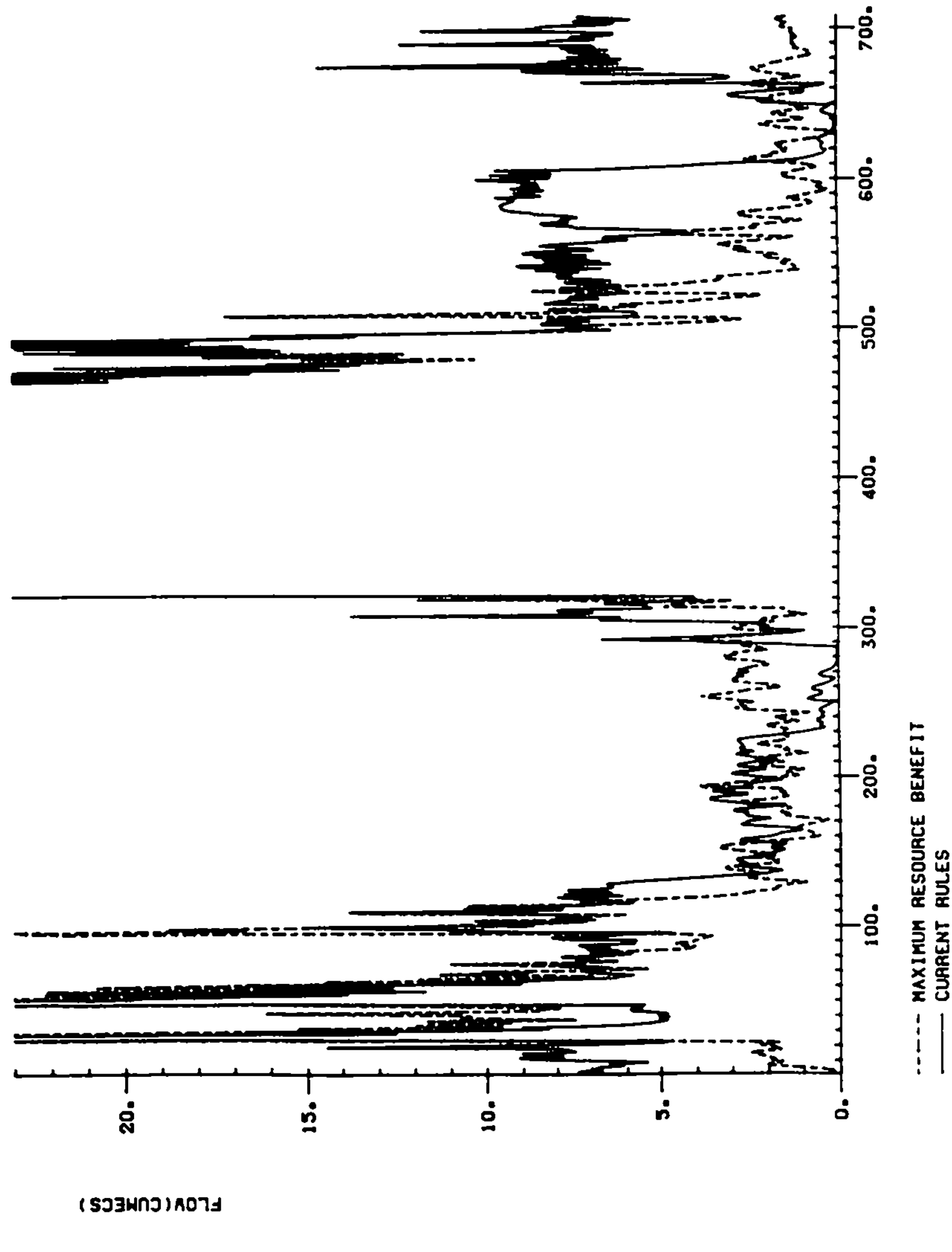


Figure 3.17 THAMES AT TEDDINGTON
1952 - 1953

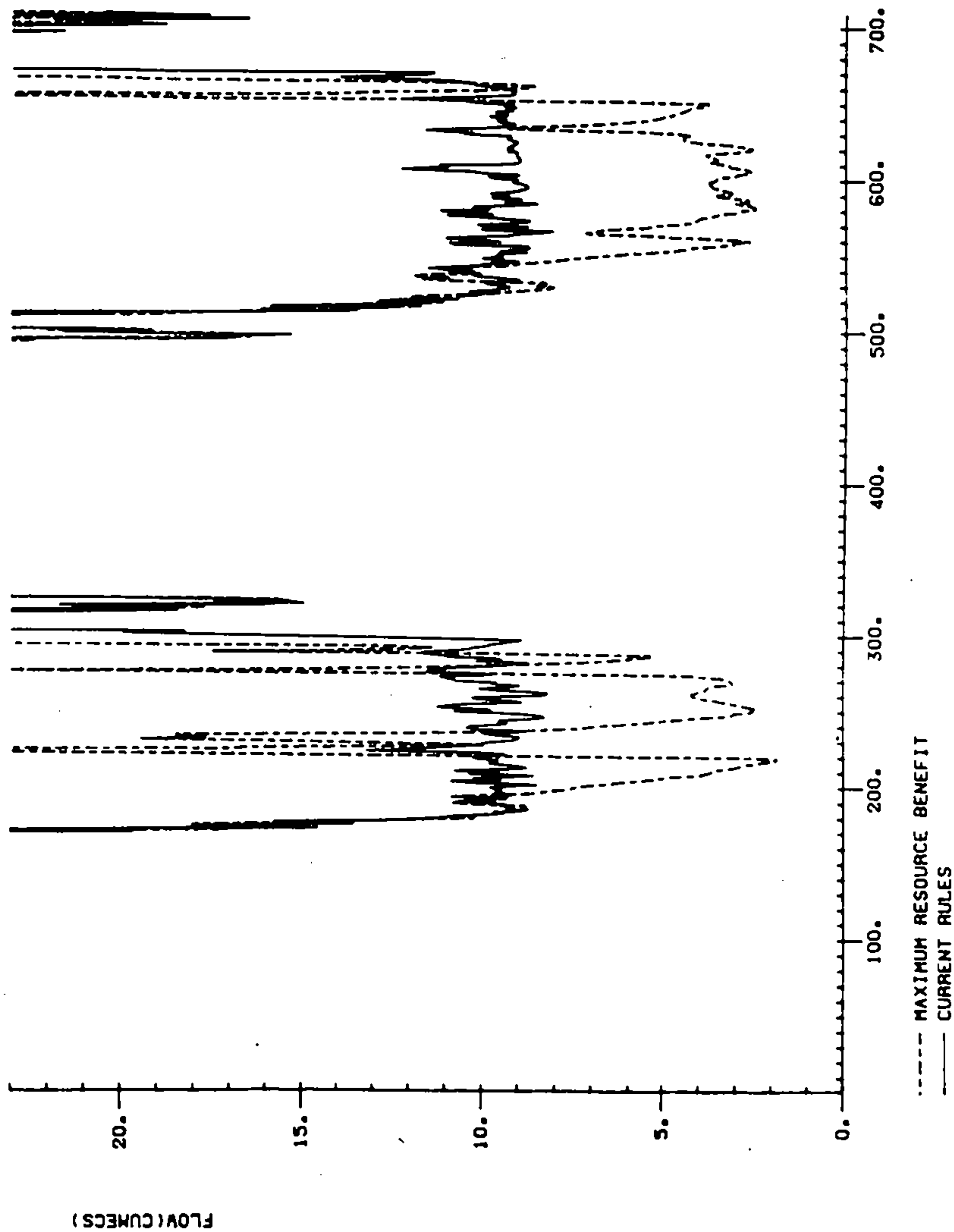


Figure 3.18 THAMES AT MOLESEY
1952 - 1953

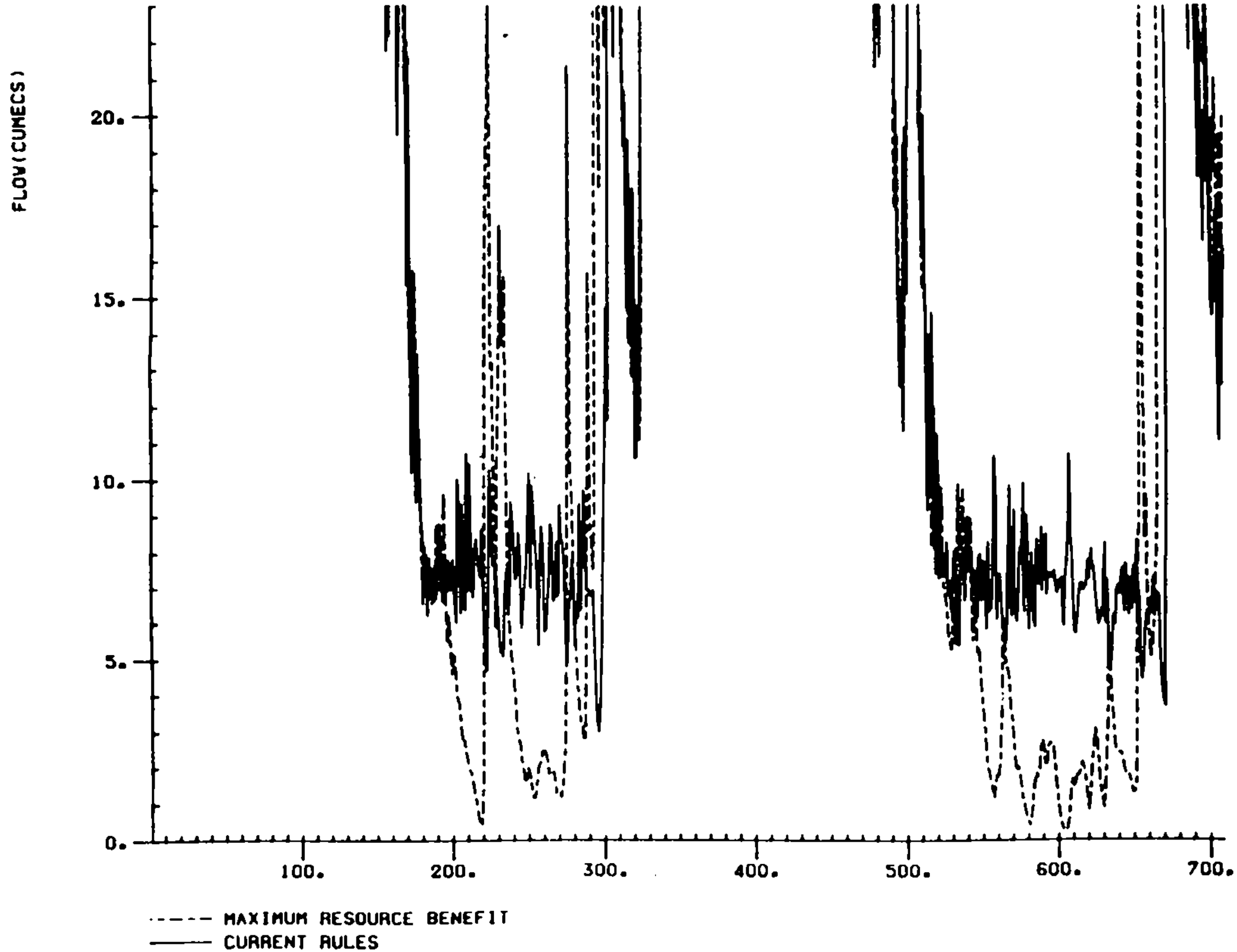


Figure 3.19 THAMES AT TEDDINGTON
1980 - 1981

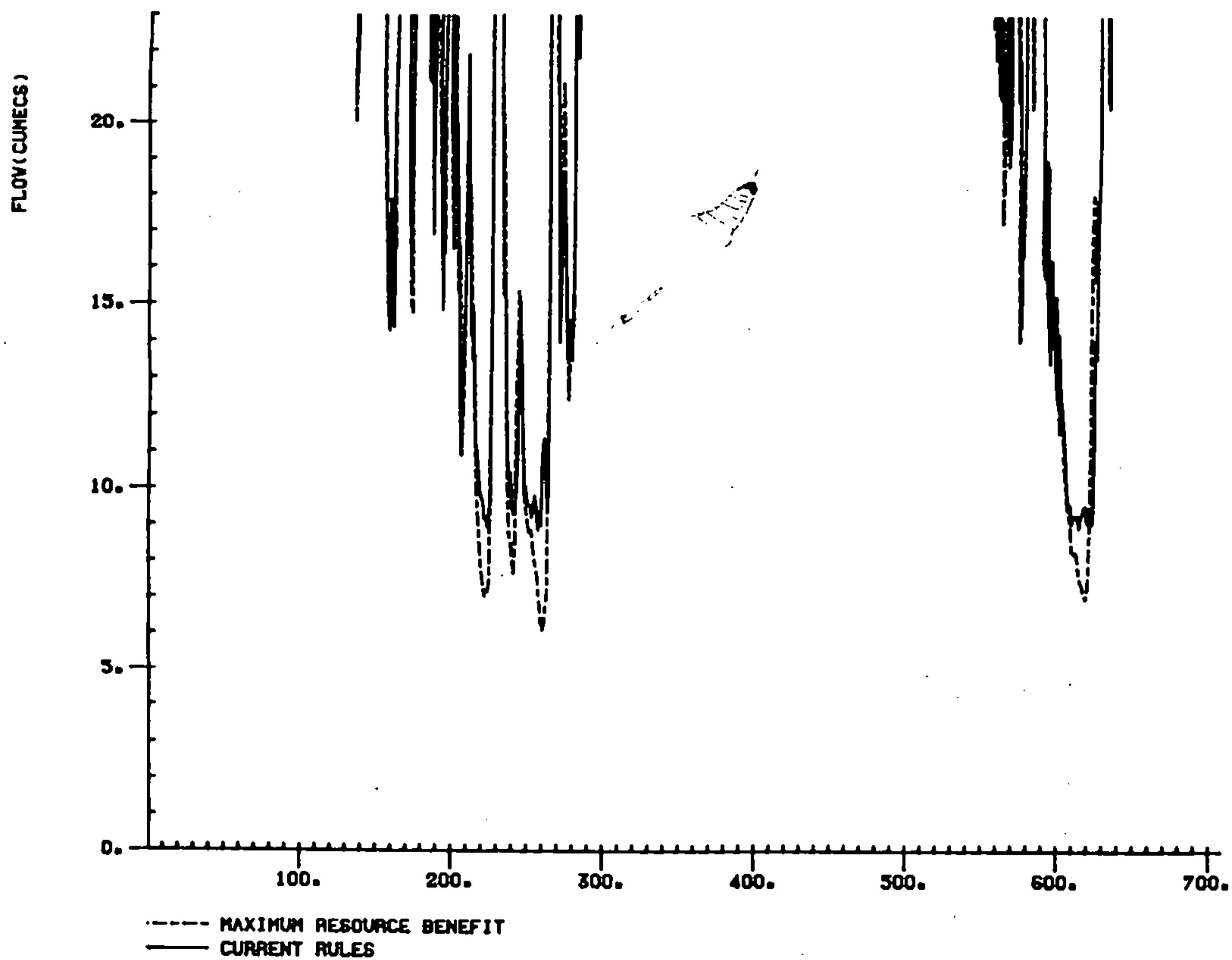
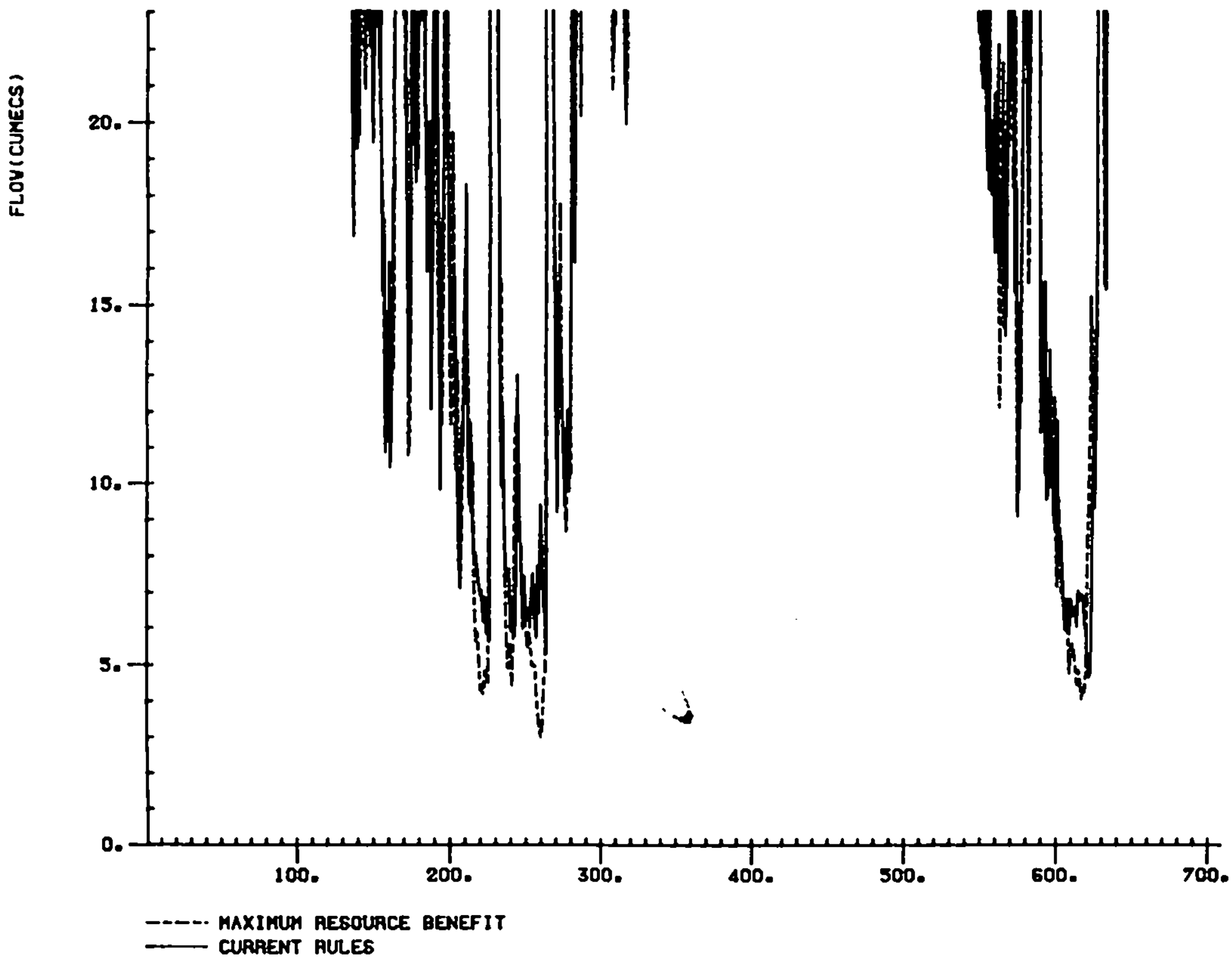


Figure 3.20 THAMES AT MOLESEY
1980 - 1981



proposed strategy and the medium flows such as during the period between days 530 - 610 are considerably reduced. Again the simulated flows at Molesey are generally lower than those at Teddington. The non-drought years show the significant predicted effect of the MRB policy compared with the current rules, summer flow levels being reduced considerably.

3.7.1.4. Finally the effect of increased demands have been investigated. Figures 3.21 and 3.22 show the flows given the chart and MRB strategies assuming the demand levels expected for the year 2006. In general there is little change in the flow patterns. This is because although demands increase by 17% a considerable proportion of this water is returned as effluent. Thus for non-drought years such as 1952-53 at Molesey (Figure 3.21) the low flows are similar to the 1984 demand runs although the low flow period is extended slightly. The effect in a drought year such at 1975-76 is shown in Figure 3.22 The chart strategy would prove to be particularly serious producing very low flows in May through to August (days 530 - 630) and requiring considerable backpumping to maintain water levels. The MRB strategy would avoid these low flow conditions since a considerable quantity of water would be abstracted earlier in the year.

3.7.2. Velocity Changes

3.7.2.1. Velocity changes in the river reflect the pattern of flow behaviour. Under summer flow conditions water velocity falls until very low velocity situations are obtained, as shown in Figure 3.23 at Teddington over the non-drought year of 1952-53. These low velocity conditions are, of course, ideal for the growth of algae since turbidity is reduced allowing greater light penetration and the average water residence time in each reach is long. Under the proposed MRB operating strategy, shown in Figure 3.24, the velocities would be reduced still further in the non-drought summers thereby increasing the potential for algal growth. However, in the spring the proposed MRB policy would have little effect on the river velocity, suggesting that abstracting additional water earlier in the year is preferable to making up losses through the low flow summer months. The effect of the increased demands, as shown in Figure 3.25, would be slightly decreased velocities during high flow conditions but to maintain or even slightly increase velocity in low flow periods because of additional effluent inflows.

3.7.3. DO-BOD Concentrations

3.7.3.1. As previously discussed, the oxygen concentration

Figure 3.21 THAMES AT MOLESEY
1952 - 1953
2006 DEMANDS

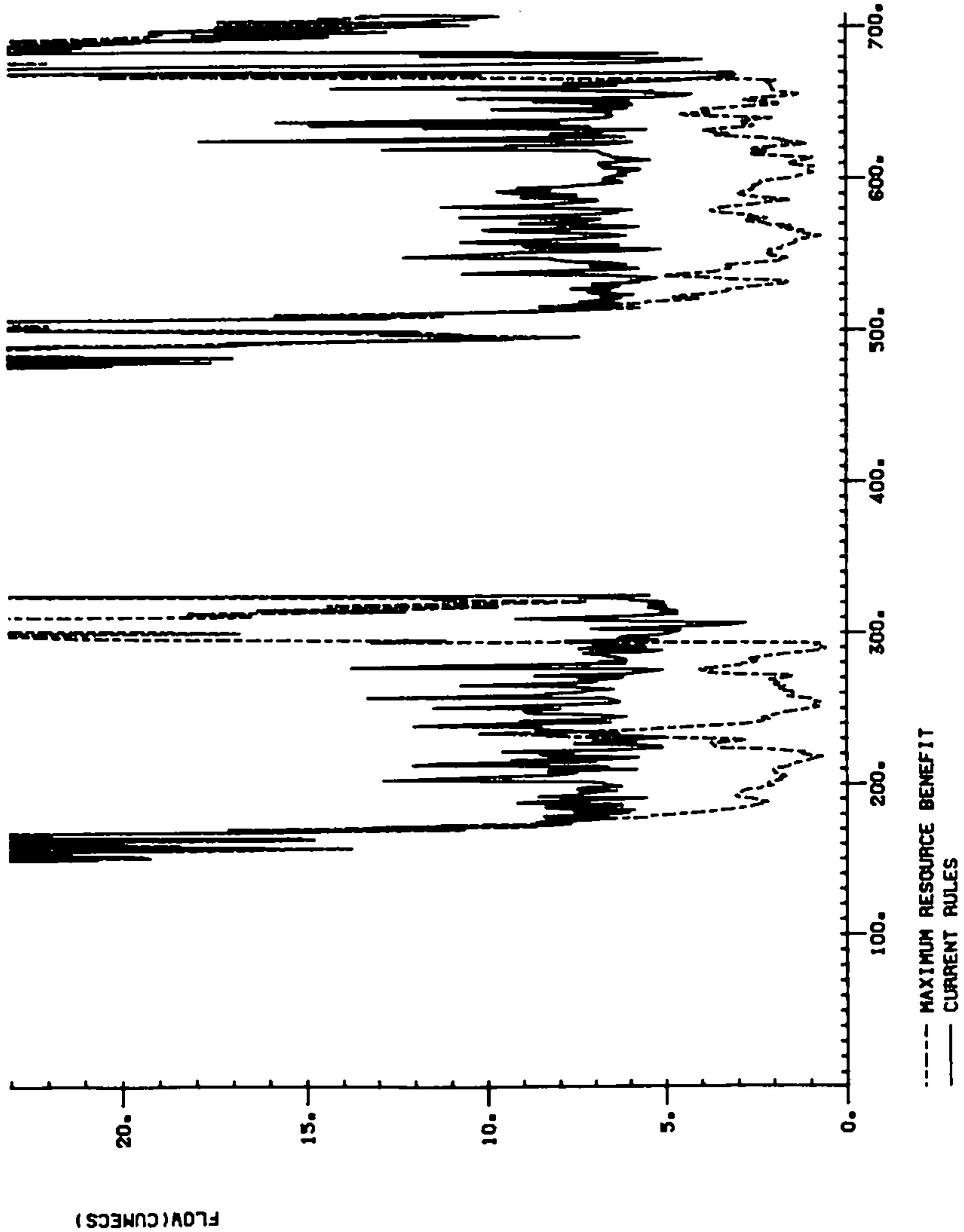
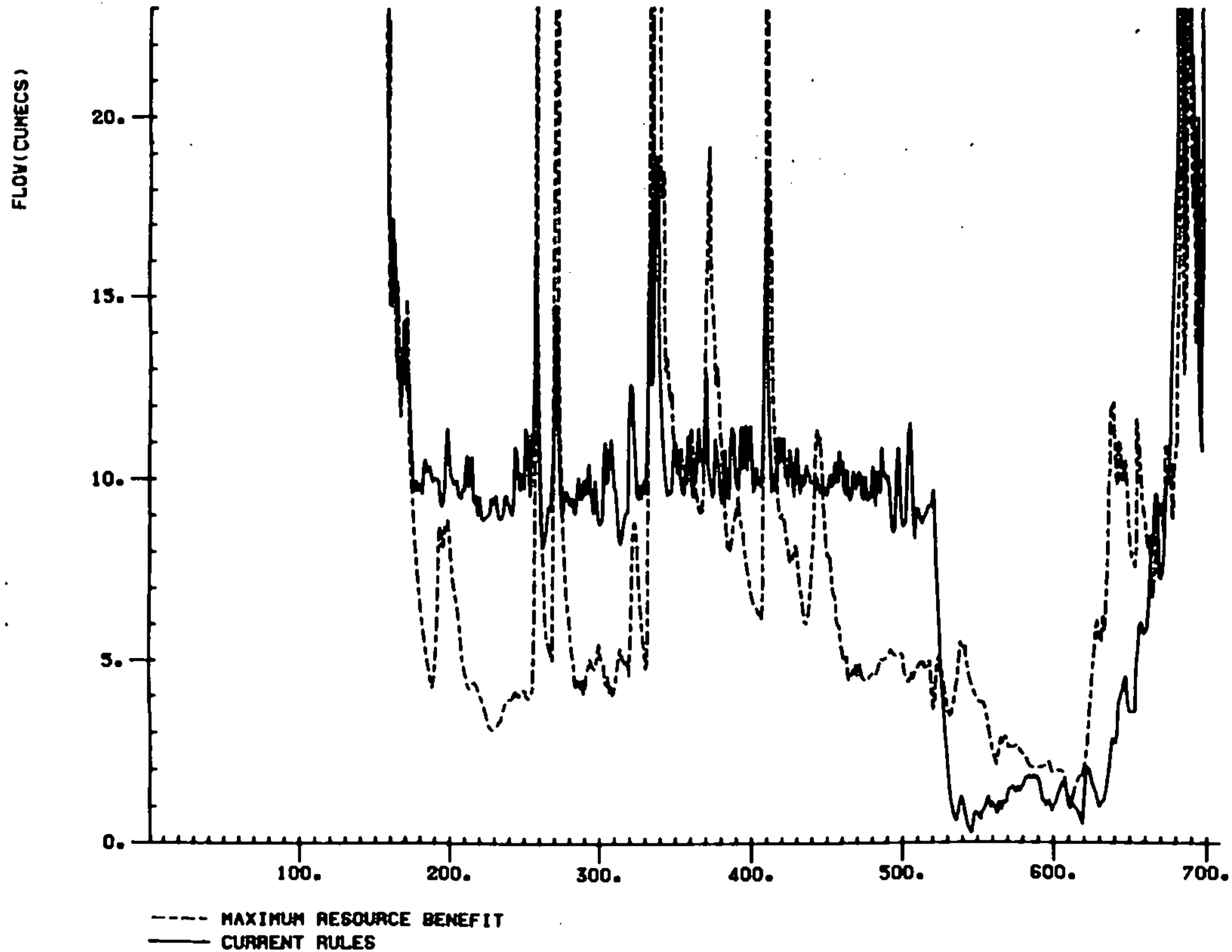


Figure 3.22 THAMES AT TEDDINGTON
1975 - 1976
2006 DEMANDS



REACH 5

Figure 3.23 MEAN DAILY VELOCITY (1952-1953)

CURRENT RULES

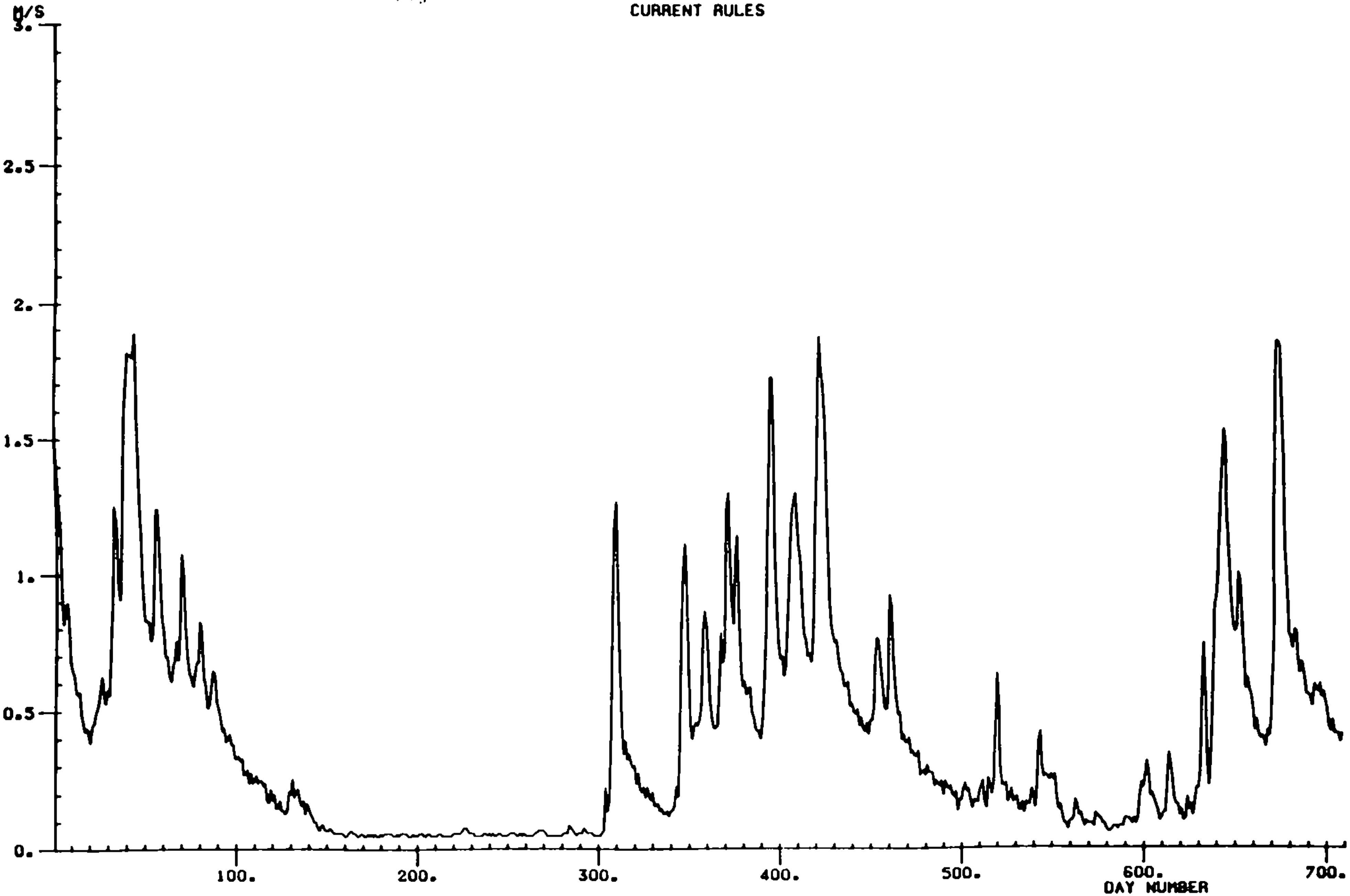
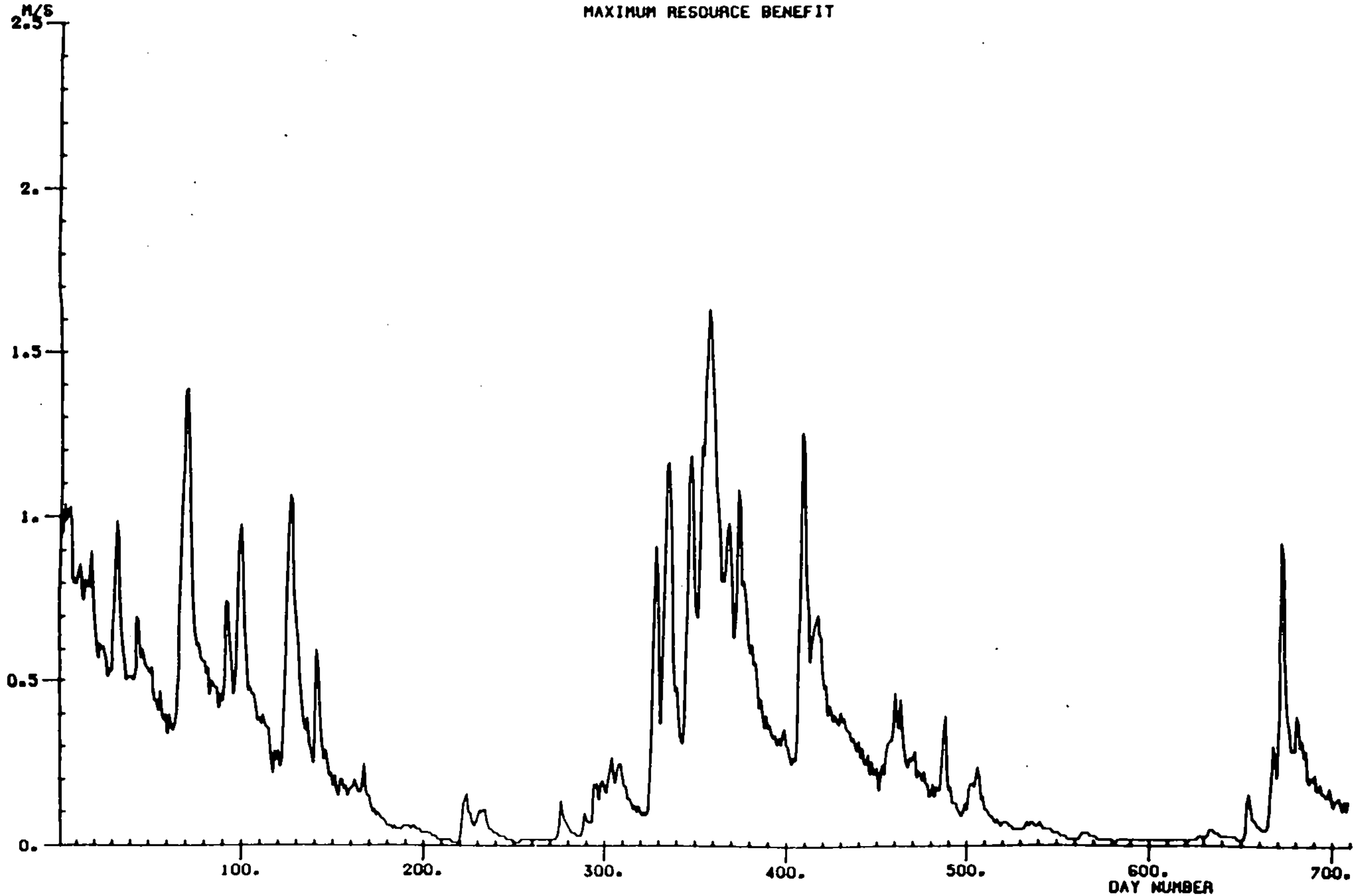
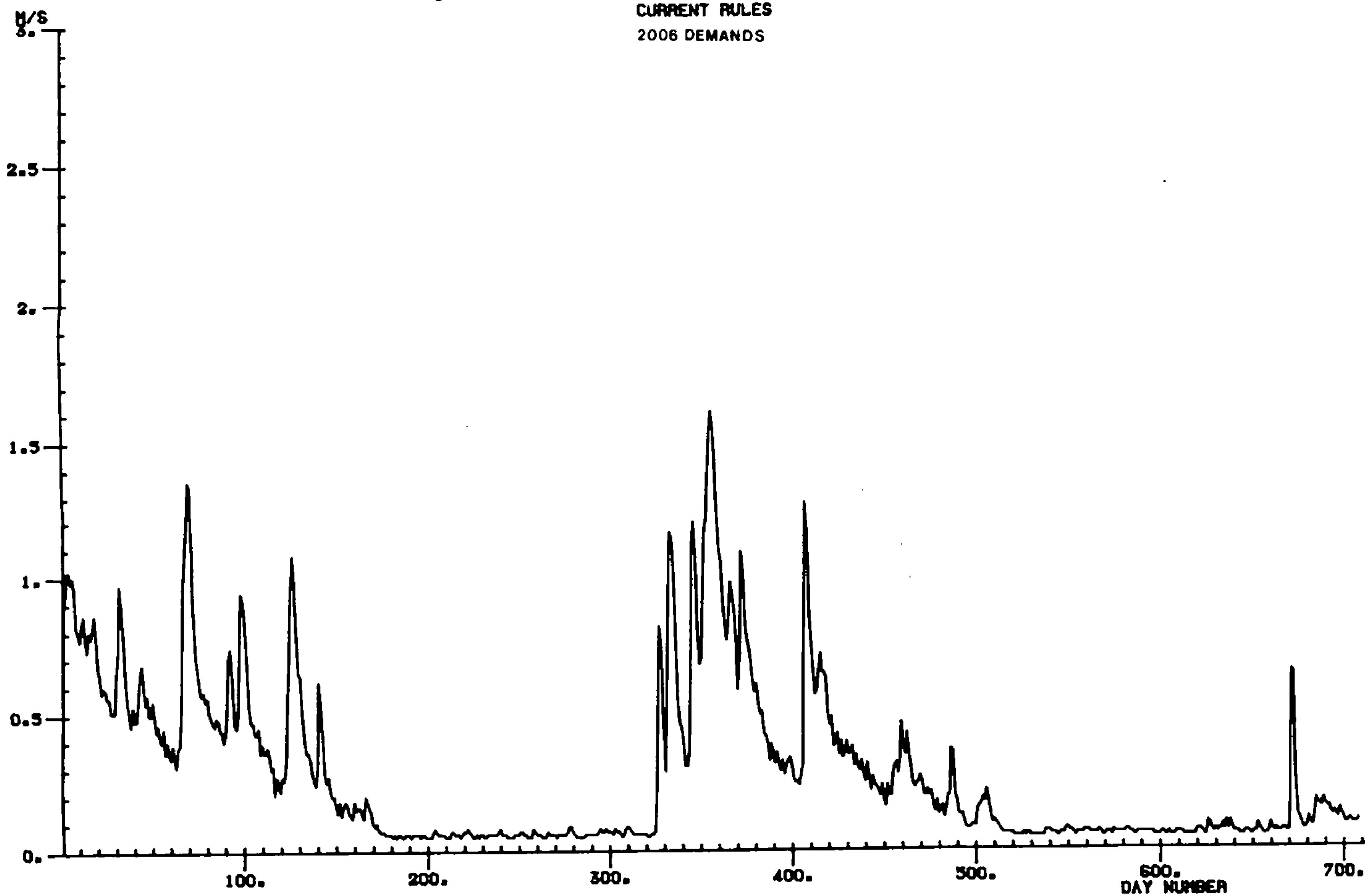


Figure3.24 REACH 5
MEAN DAILY VELOCITY (1952-1953)
MAXIMUM RESOURCE BENEFIT



REACH 5
Figure 3.25 MEAN DAILY VELOCITY (1952-1953)
CURRENT RULES
2006 DEMANDS



in the river is a complex balance of sources and sinks. Reaeration is a significant factor related closely to flow rate so that under high flow or turbulent conditions oxygen exchange with air is high. Even at medium flows the reaeration is important because of the effects of weirs which provide additional turbulence and increase the exchange surface area. Algae are also important producing oxygen by photosynthesis and removing oxygen by respiration and decay following death. The effect of long residence times in the river can be significant providing ample time for algal growth and decay of organic material.

3.7.3.2. The dynamic DO-BOD model encompasses all these processes and is, therefore, a useful tool to evaluate the effects of different flow strategies. As in the case of the flow data, the model has been run initially assuming 1984 demand levels and using the latest water quality data for inputs such as effluents and tributaries. In order to reflect the worst situation with regard to algal growth, the weekly 1975, 1976 chlorophyll-a data has been used. DO and BOD concentrations have been simulated in the lower five reaches of the Thames under drought and non-drought conditions and for the chart and MRB strategies. Results are presented for the Molesey reach since this is most critical in terms of flow.

3.7.3.3. In the case of BOD few differences between chart and MRB strategies have been detected. Figures 3.26 to 3.29 show the BOD concentrations at Molesey Weir for the drought years of 1975-76 and 1944-45 and non-drought years of 1952-53 and 1980-81. Because the MRB flows under drought conditions are very similar to current flows, the dilution of effluents is the same, and, as shown in Figures 3.26 and 3.27, BOD's are similar. It is only in non-drought years, shown in Figures 3.28 and 3.29, that a significant change in the BOD is predicted primarily because of the major differences in flow and therefore dilution between summer MRB flows and current flows.

3.7.3.4. In the case of DO, the concentrations during drought years for the MRB and the current flows are similar, as shown in Figures 3.30 and 3.31 Again this is because the flow patterns in drought years are so similar. However, DO does fall significantly when very low flow conditions prevail, coupled with a major algal bloom. For example, in 1944, shown in Figure 3.31 on day 300 the DO falls to 4 mgl^{-1} under the current rules. The proposed flow strategy which avoids the extremely low flow conditions has the effect of elevating the DO levels slightly.

Figure 3.26 THAMES AT MOLESEY
1975 - 1976

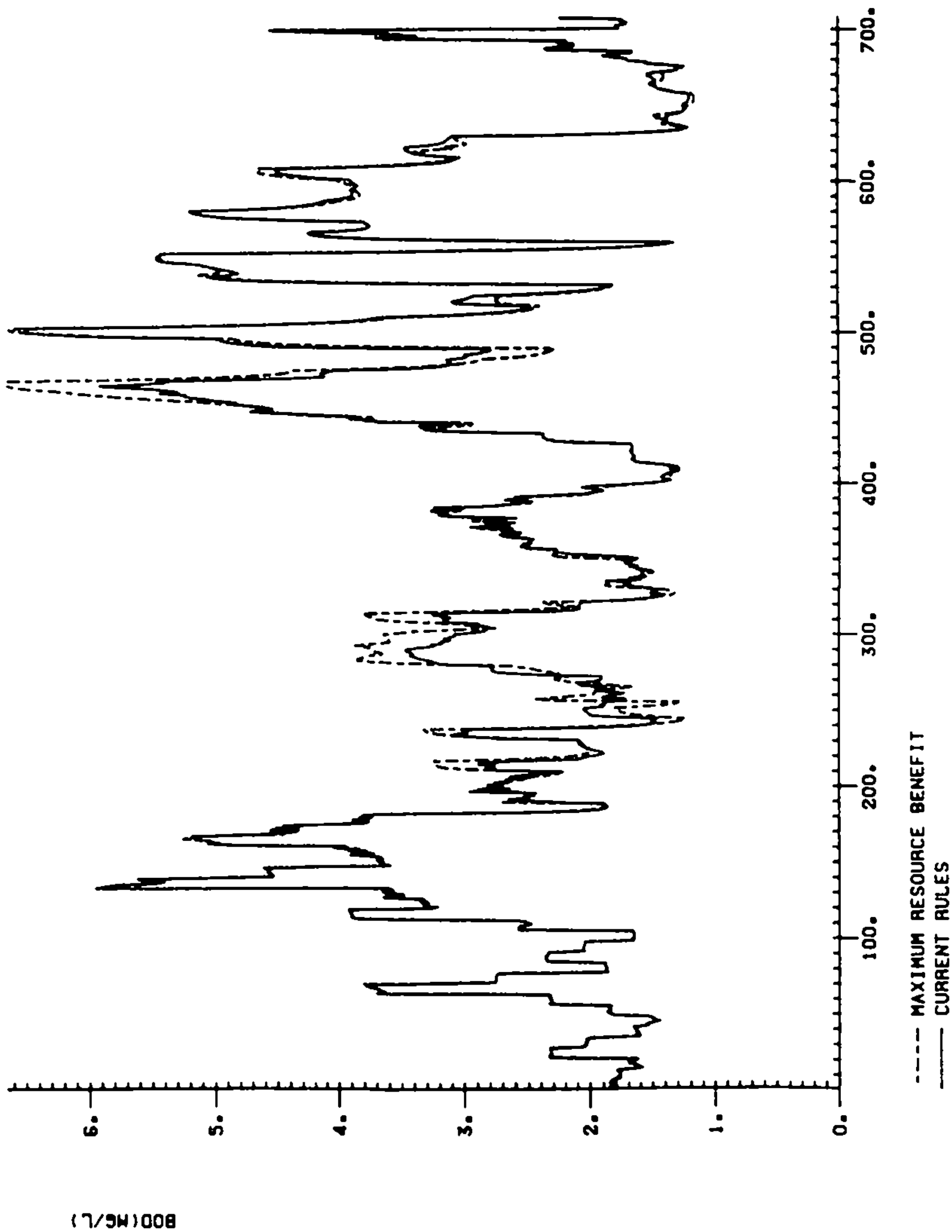


Figure 3.27 THAMES AT MOLESEY
1944 - 1945

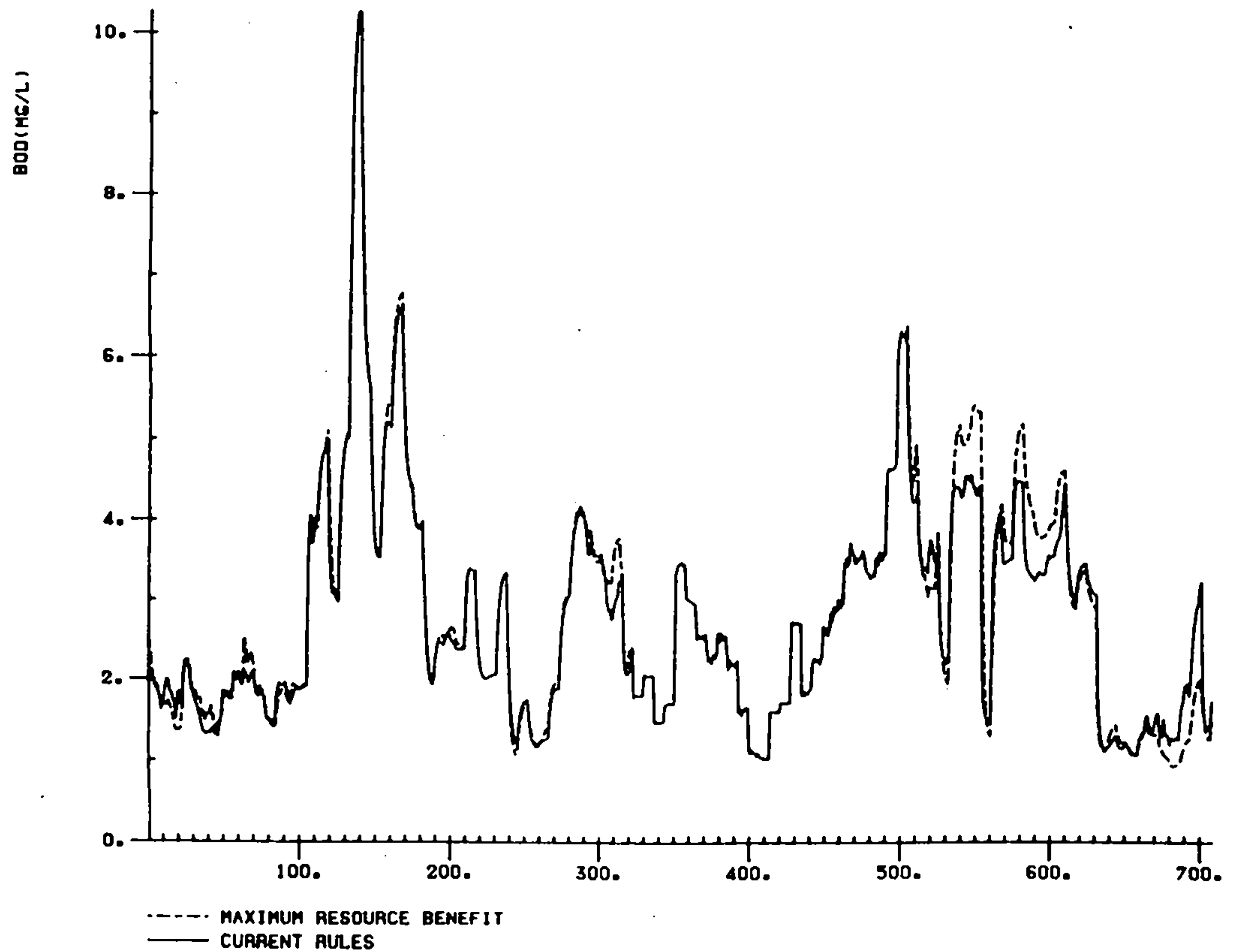


Figure 3.28 THAMES AT MOLESEY 1952 - 1953

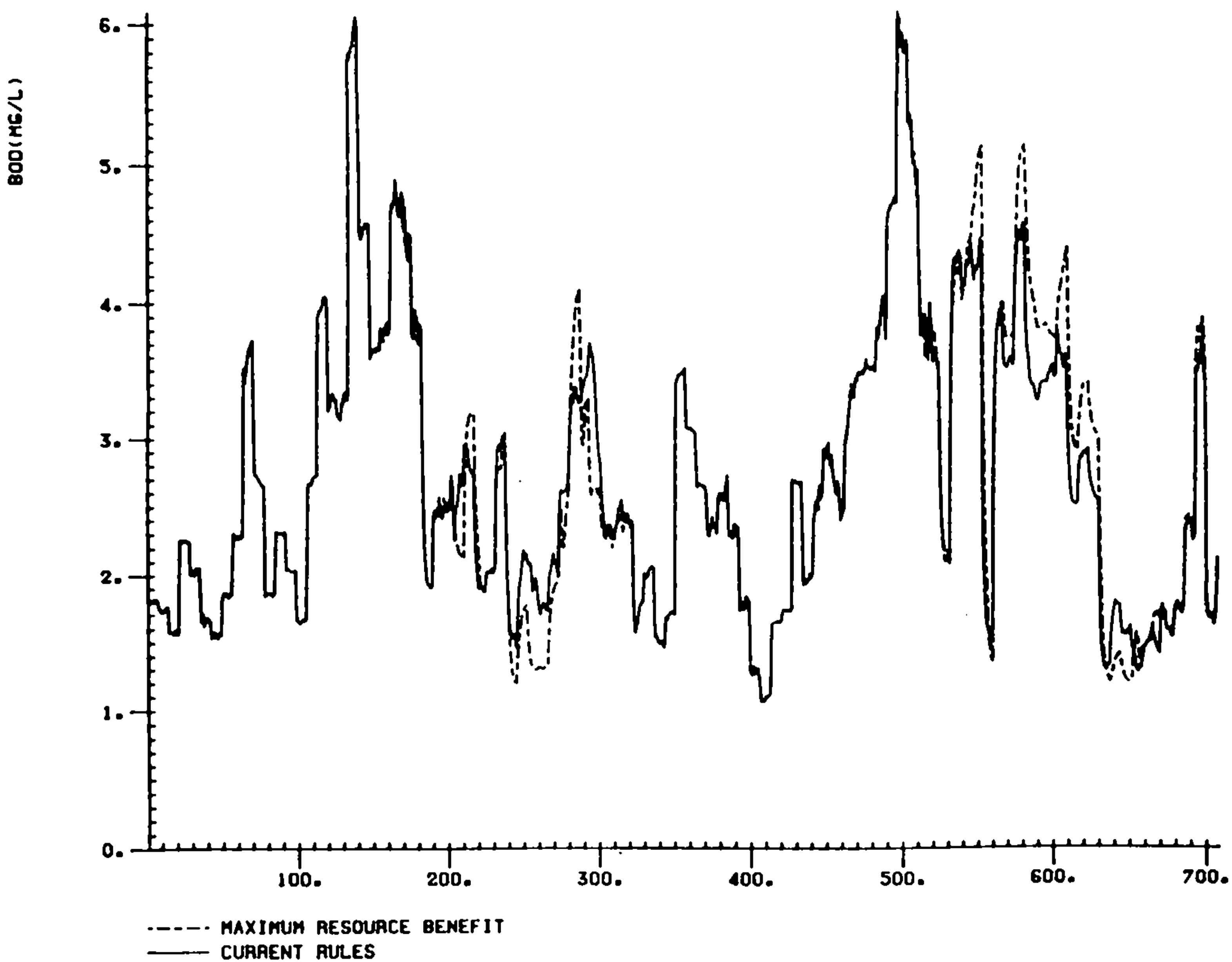


Figure 3.29 THAMES AT MOLESEY
1980 - 1981

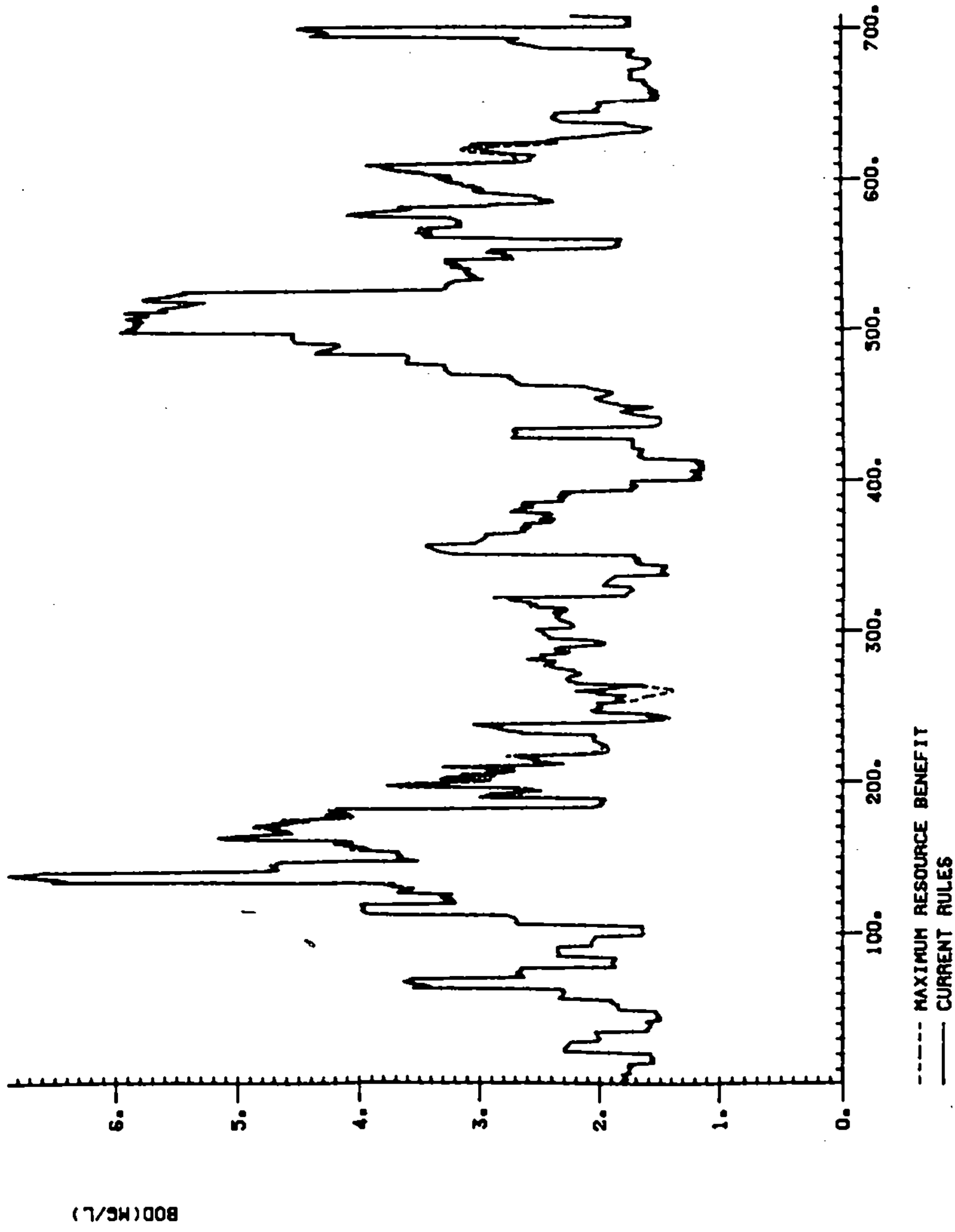


Figure 3.30 THAMES AT MOLESEY
1975 - 1976

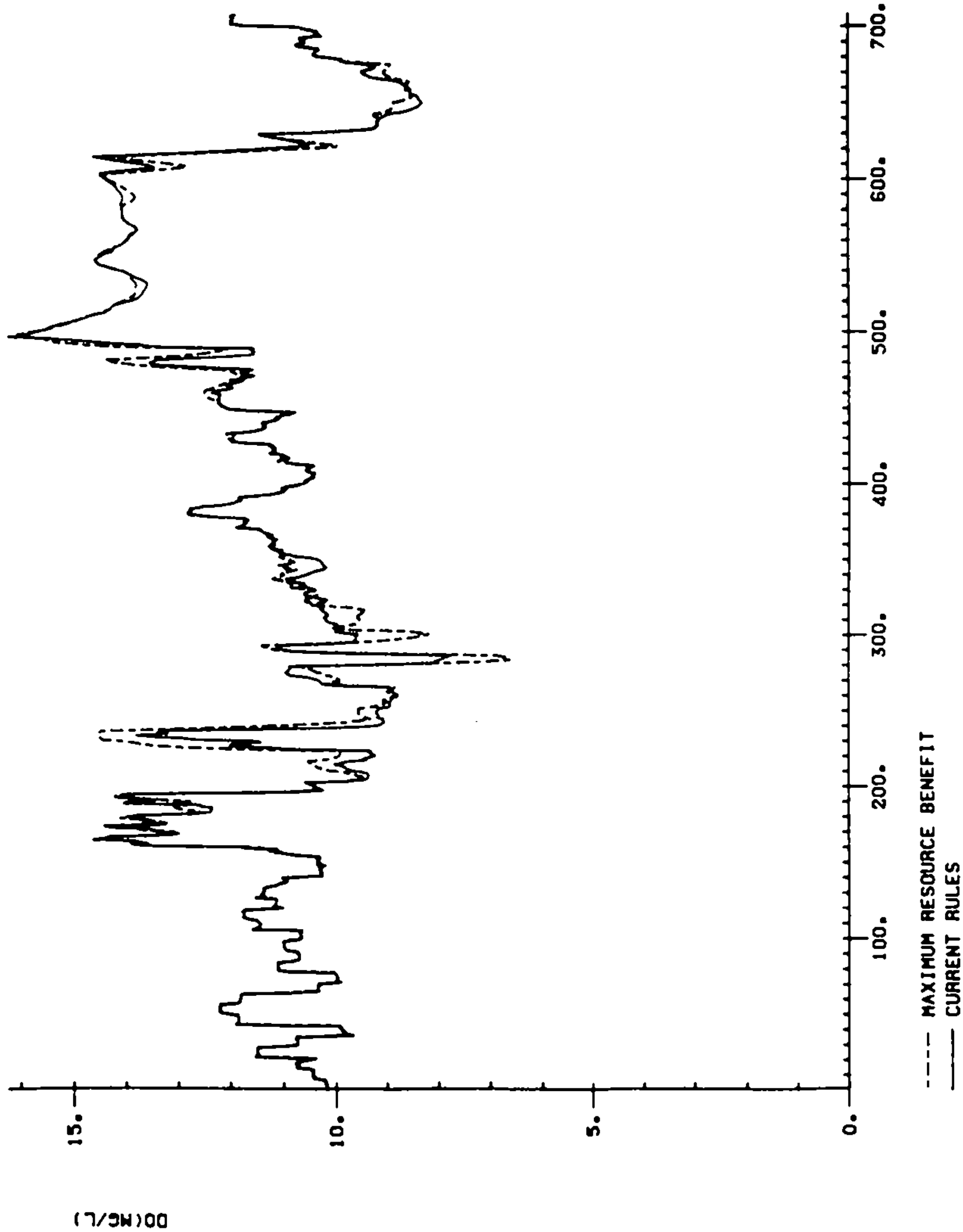
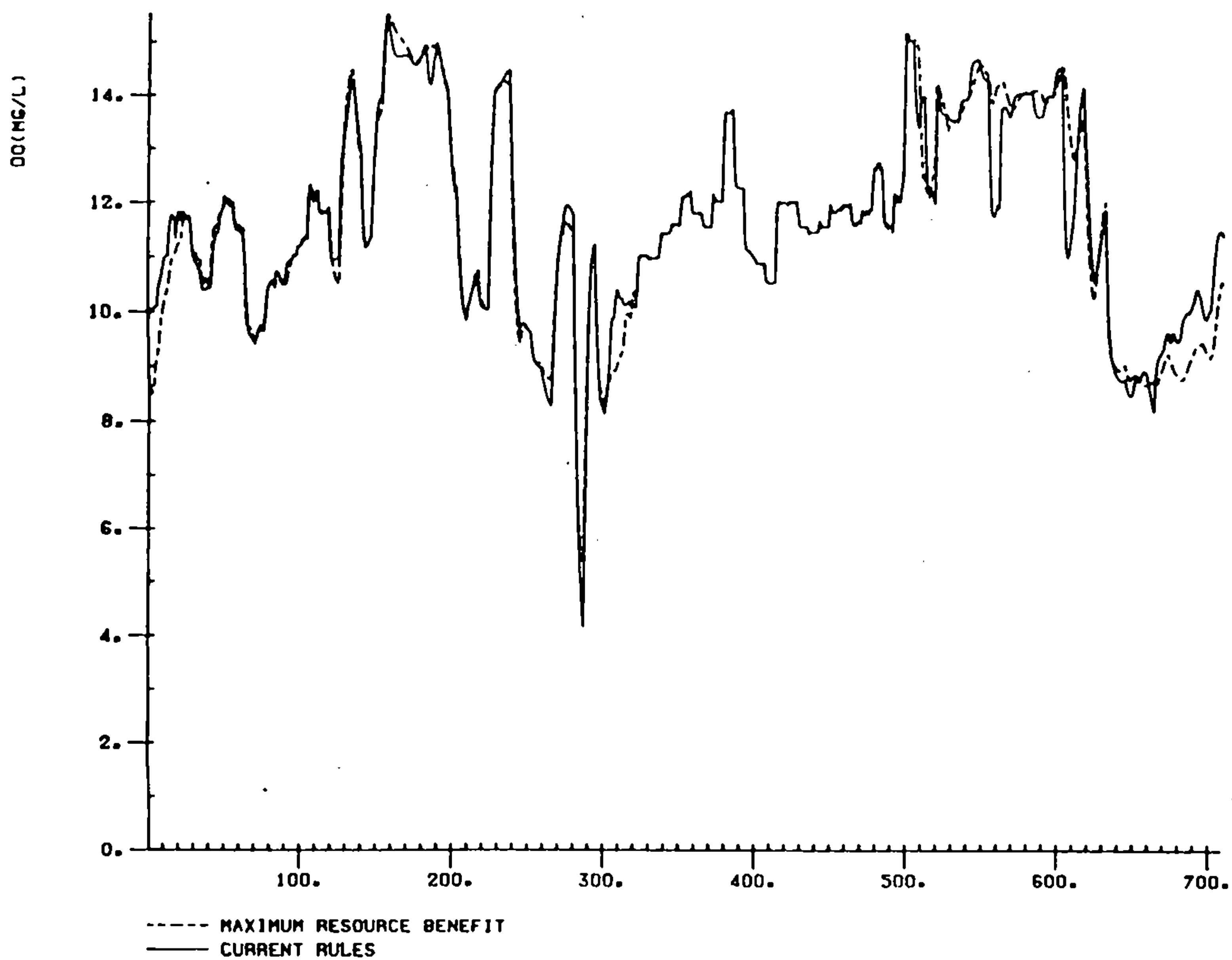


Figure 3.31 THAMES AT MOLESEY
1944 - 1945



3.7.3.5. In non-drought years there is little difference between the MRB policy and the current policy. Despite the decreased flows generated by the MRB policy discussed in the previous section, there is sufficient reaeration to maintain DO levels. However, the model does not account for the increased algal productivity that may be present during these low flow periods in non-drought years and thus the DO situation may be slightly worse than predicted. Given this possibility it may be useful for Thames Water Authority to consider an alternative policy to the MRB approach in non-drought summers.

3.7.3.6. The main effects of increasing the demand are illustrated in Figures 3.32 and 3.33 for BOD and DO levels at Teddington. Because of the increased load of effluent entering the river via the Hogsmill river BOD levels in the Molesey - Teddington reach are slightly higher. This has an effect on the DO levels which appear slightly lower and could reduce the minimum oxygen levels significantly on occasions. However the risks of this happening under the chart strategy is just as high if not higher, than under the MRB strategy because of the lower flows.

3.7.4. Ammonia Concentrations

3.7.4.1. The flow and water quality modelling studies indicate that the lower reaches of the river would be most affected by changes in operating strategy. The reach above Molesey Weir is important because of major abstractions and the very low flow conditions that sometimes prevail. The reach between Molesey and Teddington is also important because of the low flows and the impact of effluent entering the reach via the Mole and Hogsmill Rivers. In order to explore the effect of pulses of ammonia which can occasionally enter the reach the ratio of the flows of the Mole and the Hogsmill to the flow in the reach under the current and MRB strategies have been calculated. Table 3.3 shows these ratios or average dilution factors for the drought months of June, July and August 1976, and these months therefore represent the worst possible situation.

TABLE 3.3. DILUTION FACTORS FOR JUNE,JULY AND AUGUST 1976

	MRB		CURRENT	
	HOGSMILL	MOLE	HOGSMILL	MOLE
JUNE	0.28	0.45	0.27	0.45
JULY	0.26	0.39	0.21	0.31
AUGUST	0.25	0.35	0.25	0.33

Figure 3.32 THAMES AT TEDDINGTON
1975 - 1976
2006 DEMANDS

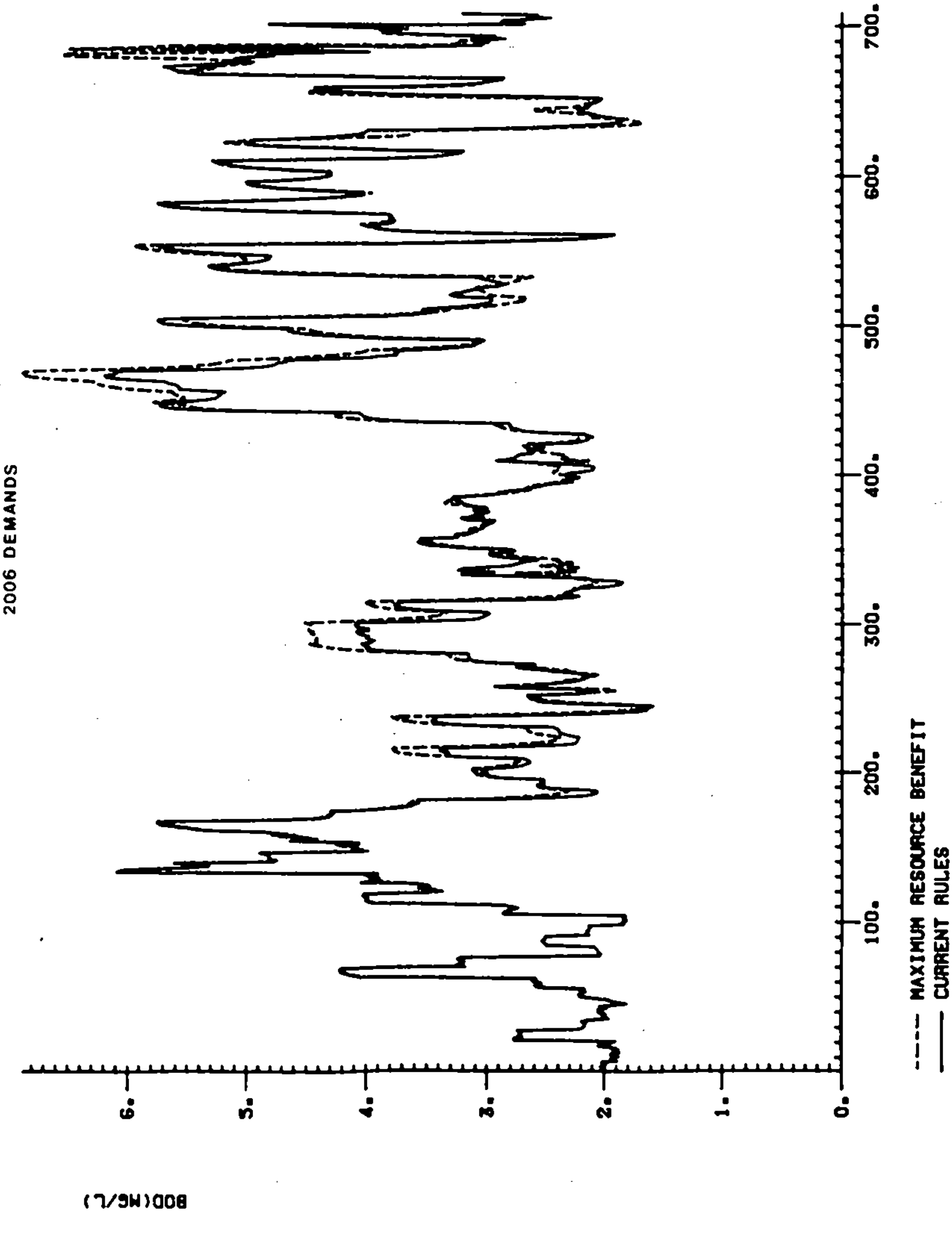
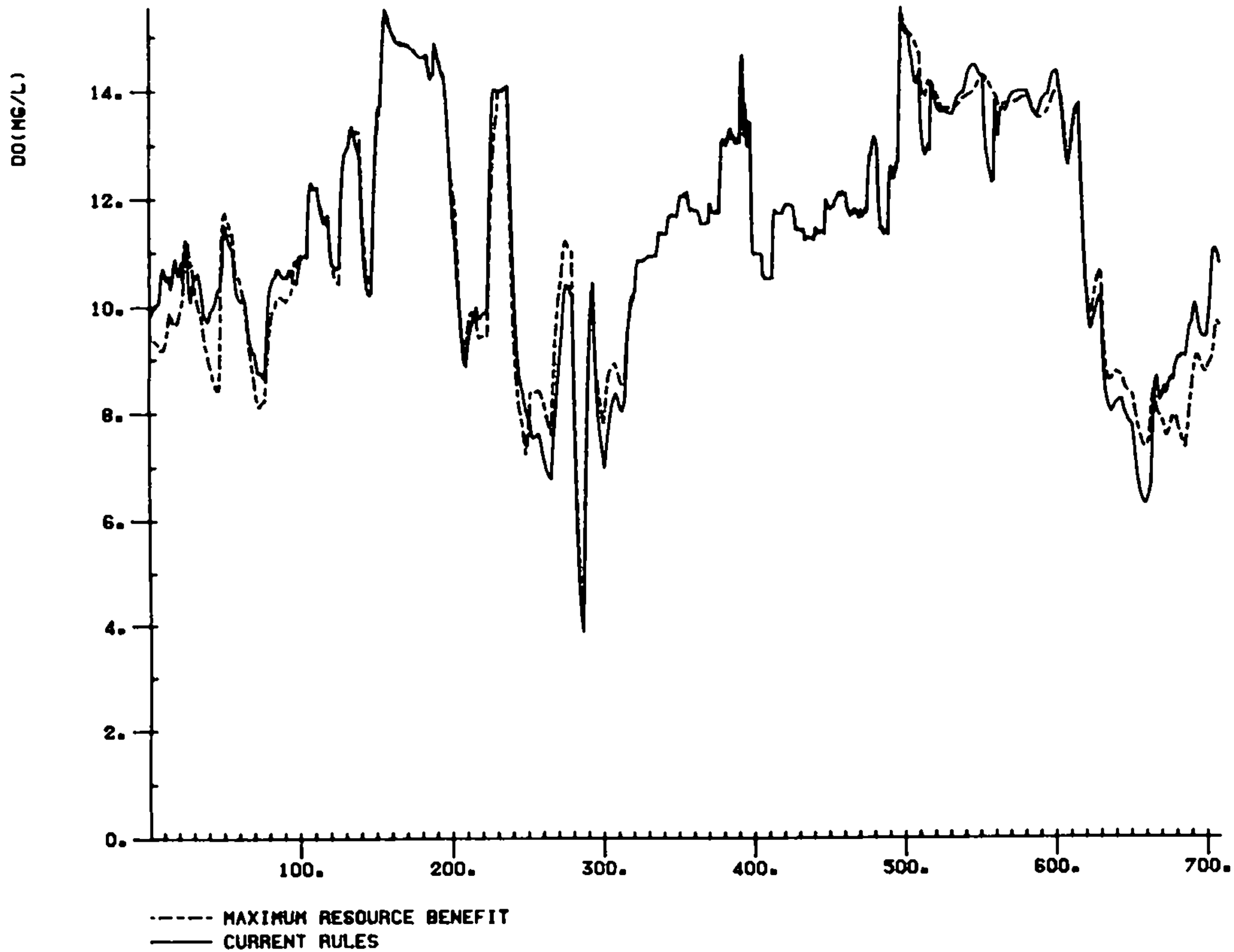


Figure 3.33 THAMES AT TEDDINGTON
1944 - 1945
2006 DEMANDS



3.7.4.2. Recent improvements in effluent treatment facilities have reduced the concentrations of ammonia in the Mole and the Hogsmill. Mean concentrations were 1.27 mgl^{-1} for the Hogsmill and 0.58 mgl^{-1} for the Mole with the peak values being 6.6 mgl^{-1} and 1.2 mgl^{-1} respectively. Multiplying the peak Hogsmill value by the dilution factors under the proposed strategy gives concentrations of 1.8, 1.7 and 1.65 mgl^{-1} in the Teddington Reach for June, July and August 1976. In the case of a pulse of ammonia from the Mole and concentration in the reach would be 0.54, 0.47 and 0.42mgl^{-1} for June, July and August. These levels are considerably higher than elsewhere on the Thames but given the high levels of DO they are unlikely to cause any particular problems.

3.8 CONCLUSIONS

3.8.1. The conclusions from the Modelling study are as follows:-

3.8.1.1. The flows during drought conditions are likely to be very similar under the proposed operating strategy to those resulting from the current approach. The proposed rules will slightly improve the situation since it should be possible to avoid the extreme low flow conditions and back pumping should not be necessary.

3.8.1.2. In general, the flows over Molesey Weir would be lower than the flows at Teddington; this is because of the major abstractions upstream of the weir and the inflow of the Hogsmill and Mole rivers which enter below the weir.

3.8.1.3. Flow in non-drought years would be considerably affected by the MRB policy. By removing as much water as possible in non-drought summers, flows would be significantly reduced.

3.8.1.4. The effect of increasing demand to the expected levels for the year 2006 will not significantly change flow patterns. Increased effluent flows would increase low flows in drought years but additional water would be removed in non-drought years.

3.8.1.5. Velocity changes in the river reflect flow changes. In drought years velocities under both policies would be very similar. However, in the summers of non-drought years the velocities would be reduced under the MRB policy. The effect of decreasing velocity is to increase reach residence time and this may tend to worsen the situation with regard to algal growth. However, non-drought spring conditions velocities would not be significantly affected by the proposed policy, suggesting that abstracting additional water earlier in the year would certainly be preferable to making up losses through low flow summer months. This is particularly true when the increased demand levels are considered.

3.8.1.6. Dissolved oxygen levels in the River Thames are generally high being close to saturation for most of the year. Diurnal variations in summer can be significant and are related to solar radiation levels and the growth of algae within the water column. During summer months extremely high DO concentrations up to 180% of saturation have been recorded but low concentrations of 40-50% of saturation have occasionally been monitored. Flow has an important effect on DO levels because of reaeration processes and the residence time of water in a reach. Reaeration increases with increasing flow rate and effluents are also more diluted. Under low flow conditions residence times are high and allow more time for decay of organic material and create ideal conditions for the growth of algae. Low DO levels often follow the collapse of an algal bloom which may be brought on by a change in weather conditions.

3.8.1.7. In drought years the BOD concentrations for both MRB and chart operating strategies are very similar and it is only in non-drought year summers that any significant increase in BOD levels is predicted. BOD levels are likely to rise given increased demand levels and hence increased effluent loads.

3.8.1.8. In the case of DO levels the proposed rules would improve the DO situation slightly in drought summers since the extreme low flows and back pumping would be avoided. However, in non-drought summers there would be a possibility of increased algal growth and this might have a detrimental effect on DO levels. This might be particularly important in the reach above Molesey Weir because of the generally lower flows in this reach. An alternative to the MRB policy for non-drought summer months should be considered if the FBA consider that algal growth will increase during these periods. This may be particularly relevant given increased demand.

3.8.1.9. Ammonia levels in the River Thames are generally low (less than 0.5 mgl^{-1}) Occasional peaks of 6 mgl^{-1} of ammonia were recorded in the Molesey to Teddington reach in 1976 but recent improvements to treatment facilities have improved the situation in this reach. Assuming drought conditions, peak levels of ammonia would be approximately a third of the peak 1976 concentrations.

3.9. REFERENCES

Beck, M.B. (1978) Modelling of Dissolved Oxygen in a Non-Tidal Stream, In Mathematical Models in Water Pollution Control (Editor A. James), Wiley.

Beck, M.B. & Young, P.C. (1976) Systematic identificaiton of DO-BOD model structure. Am. Soc. civ Engrs envir. Engng. Div. 102, 909-924

Camp, T.R. (1965) Field estimates of oxygen balance parameters. Proc. A.S.C.E., J. Sanit. Eng. Div., Vol. 91, No SA 5-16

Casapieri, P., Fox T.M., Owers P. & Thomson G.D. (1978) A mathematical deterministic river-quality model. Part 2: Use in evaluating the water quality of the Blackwater catchment. Water. Res. 12 1155-1161

Dobbins, W.E. (1964) BOD and oxygen relationships in streams. Proc. A.S.C.E., J. Sanit. Eng. Div., Vol 90, No SA3, 53-78

Edwards, R.W. and Rolley H.L.J. (1965) Oxygen consumption of river muds. J.Ecol 53, 1-19

Knowles, G & Wakeford A.C. (1978) A mathematical deterministic river-quality model. Part 1: Formulation and description. Wat. Res 12, 1149-1153

Kowalezewski, A & Lack T.J. (1971) Primary Production and respiration of the phytoplankton of the Rivers Thames and Kennet at Reading, Fresh Wat. Biology 1, 197-212

O'Connor D.J. (1967) The temporal and spatial distribution of dissolved oxygen in streams. Water Resources Research, Vol. 3 No. 1 pp 65-79

Owens M. Knowles G. and Clark A (1969) The Prediction of the Distribution of Dissolved Oxygen in Rivers, Advances in Water Pollution Research, 4 International Conference Proceedings, Prague.

Rolley H.L.J. and Owens M (1967) Oxygen Consumption Rates and Some Chemical Properties of River Muds, Water Research 1, 759-766

Steel J.A. (1978) Reservoir Algal Productivity, In Mathematical Models in Water Pollution Coltrol (editor A. James) Wiley.

Streeter H.W. and Phelps E.B. (1925) A Study of the Pollution and Natural Purification of the Ohio River III US Public Health Bull. Wash. No 146

Thames Water Authority (1978). The 1976 Drought.

Thomann R.V. (1972) Systems Analysis and Water Quality Management, McGraw-Hill, New York.

Whitehead P.G. (1978) Modelling and operational control of water quality in river systems. Water Res. 12 377-384.

Whitehead P.G. (1979) Applications of recursive estimation techniques to time variable hydrological systems. J.Hydrol 40, 1-16

Whitehead P.G. (1980a) An instrumental variable method of estimating differential equations of dispersion and water quality in non tidal rivers. J. Ecol. Model. 9 1-14

Whitehead P.G. and Hornberger G. (1984) Modelling algal behaviour in the River Thames. Water Research, in Press.

Whitehead P.G. Young P.C. and Hornberger G. (1979) A systems model of flow and water quality in the Bedford Ouse river system -I. Stream flow modelling. Water Res. 13, 1155-1169

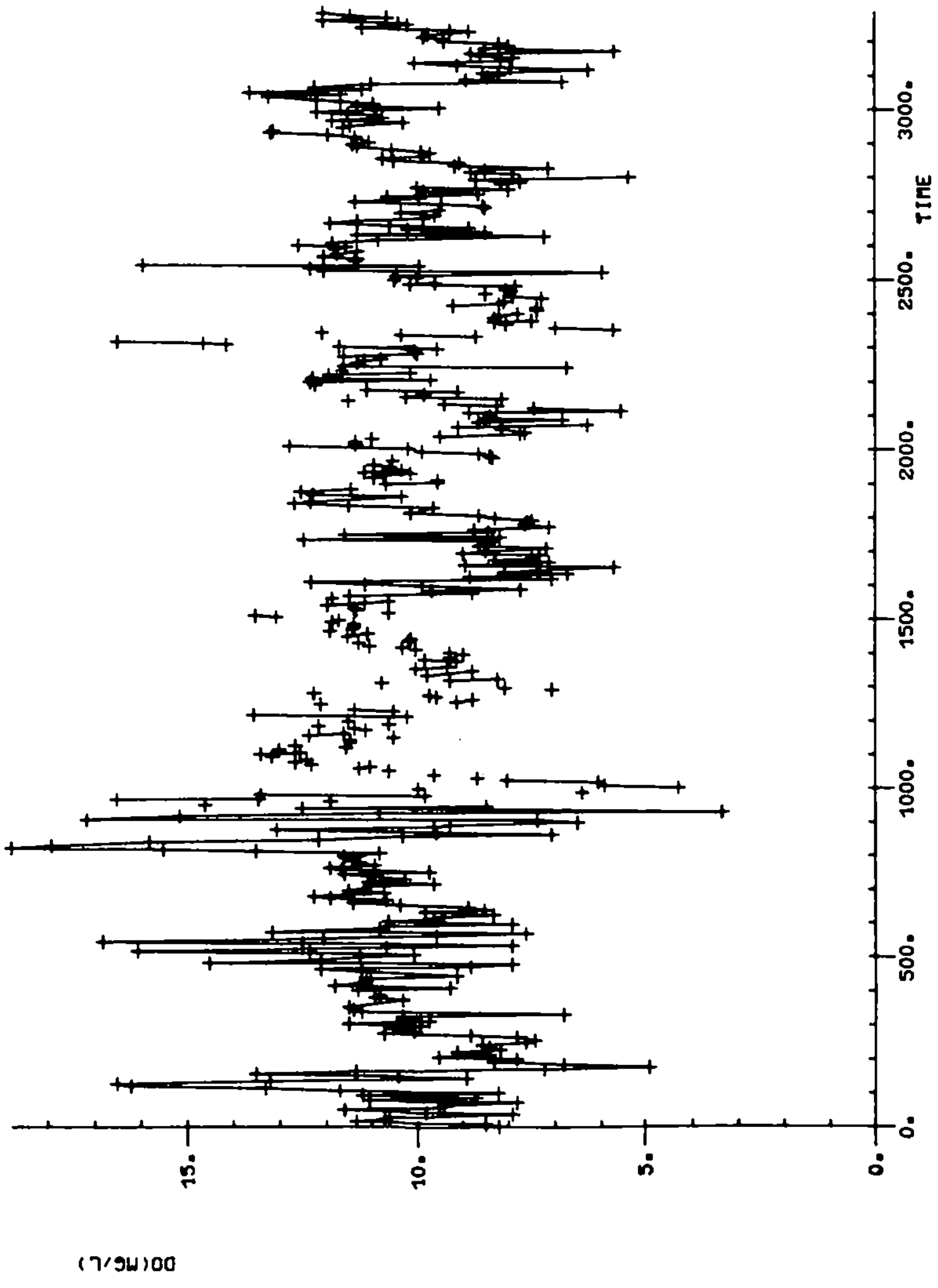
Whitehead P.G. Beck M.B. and O'Connell P.E. (1981) A systems model of flow and water quality in the Bedford Ouse river system - II Water quality modelling. Water Res. 15, 1157-1171

Whitehead P.G. and Williams R.J. (1982) A dynamic nitrate balance model for river basins. IAHS Conference Proceedings, IAHS publication 139.

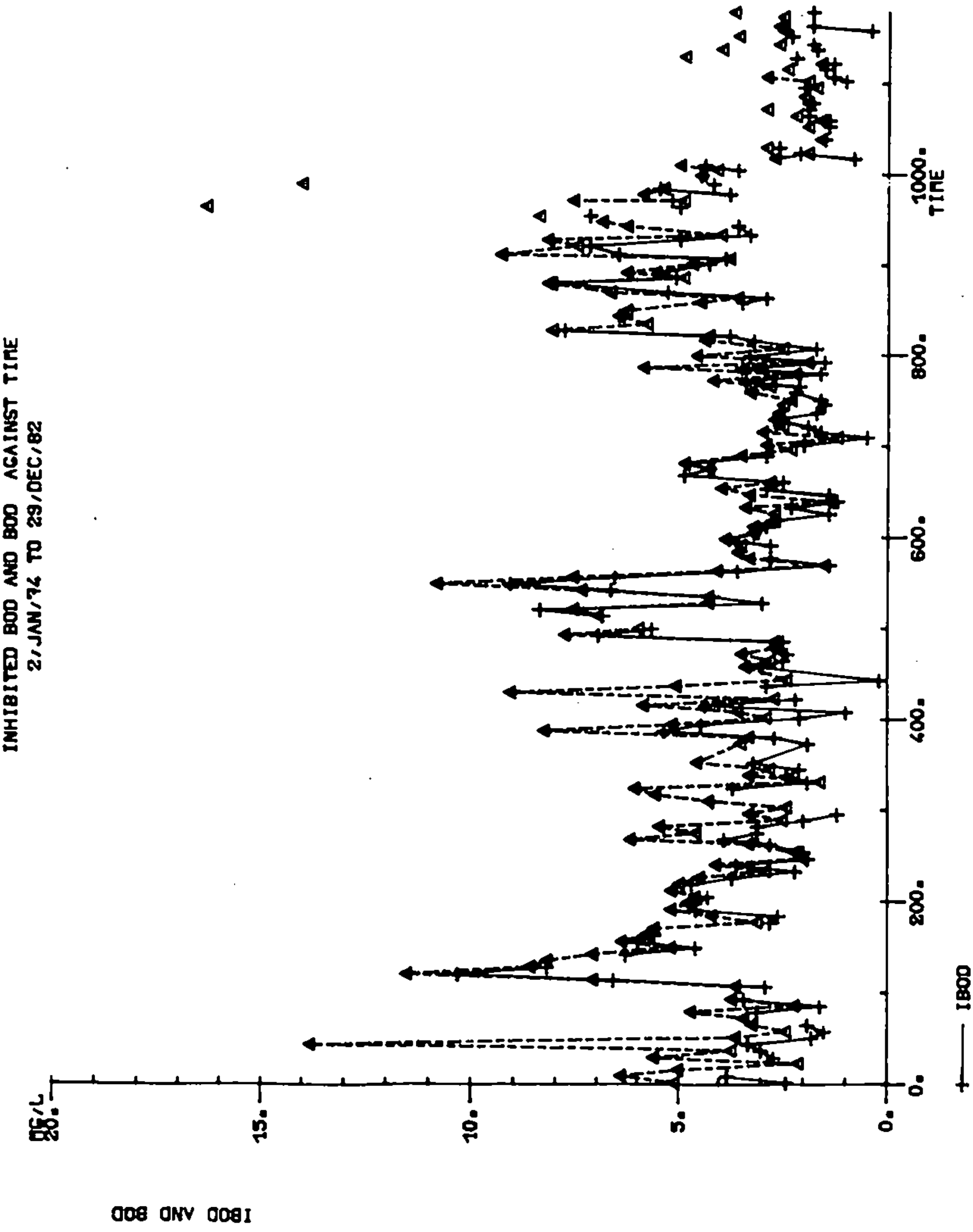
APPENDIX 3.1

Water Quality Data at Teddington Weir (1974-1982)

Appendix 3.1. Figure 1.

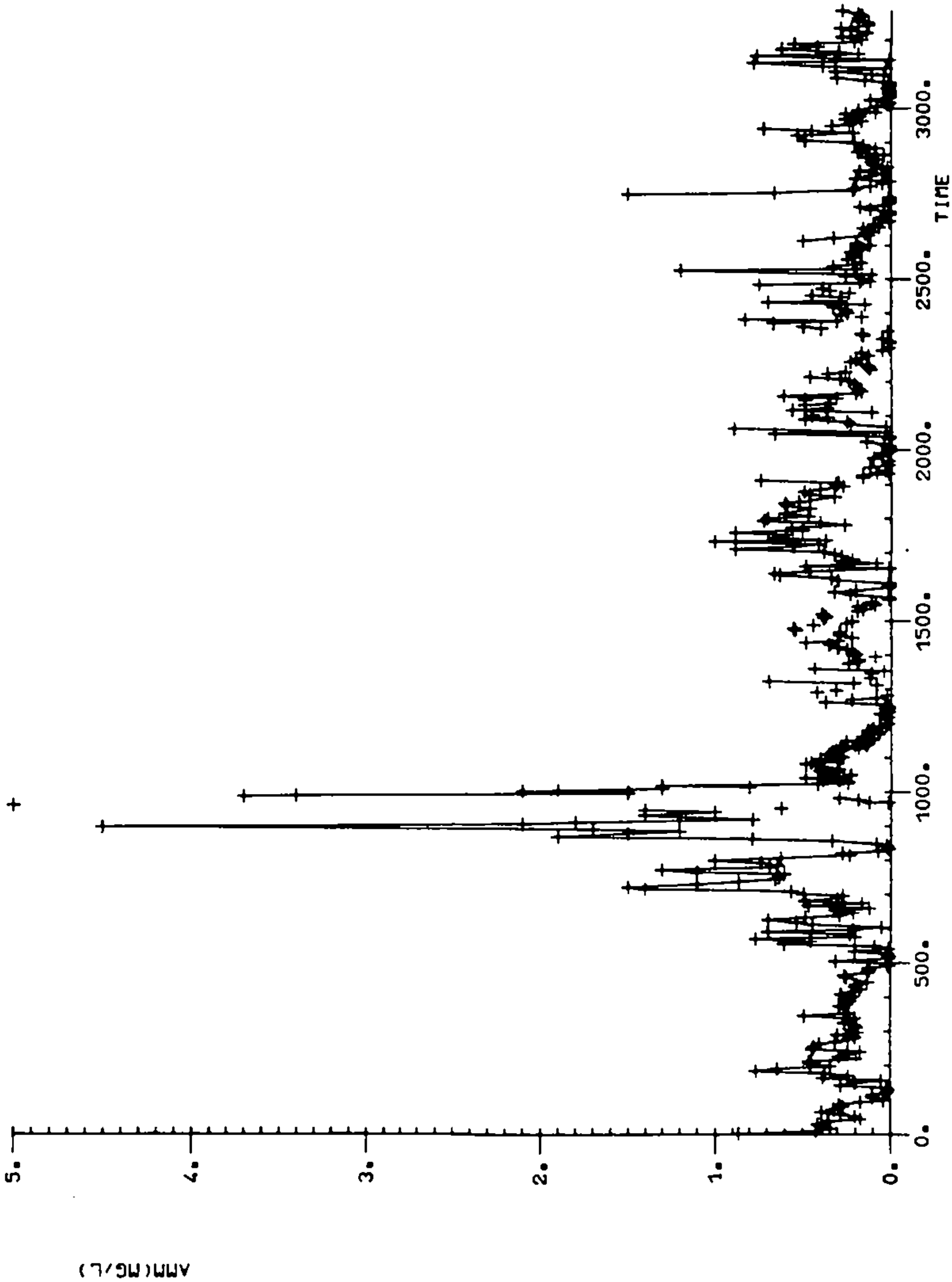


Appendix 3.1. Figure 2.



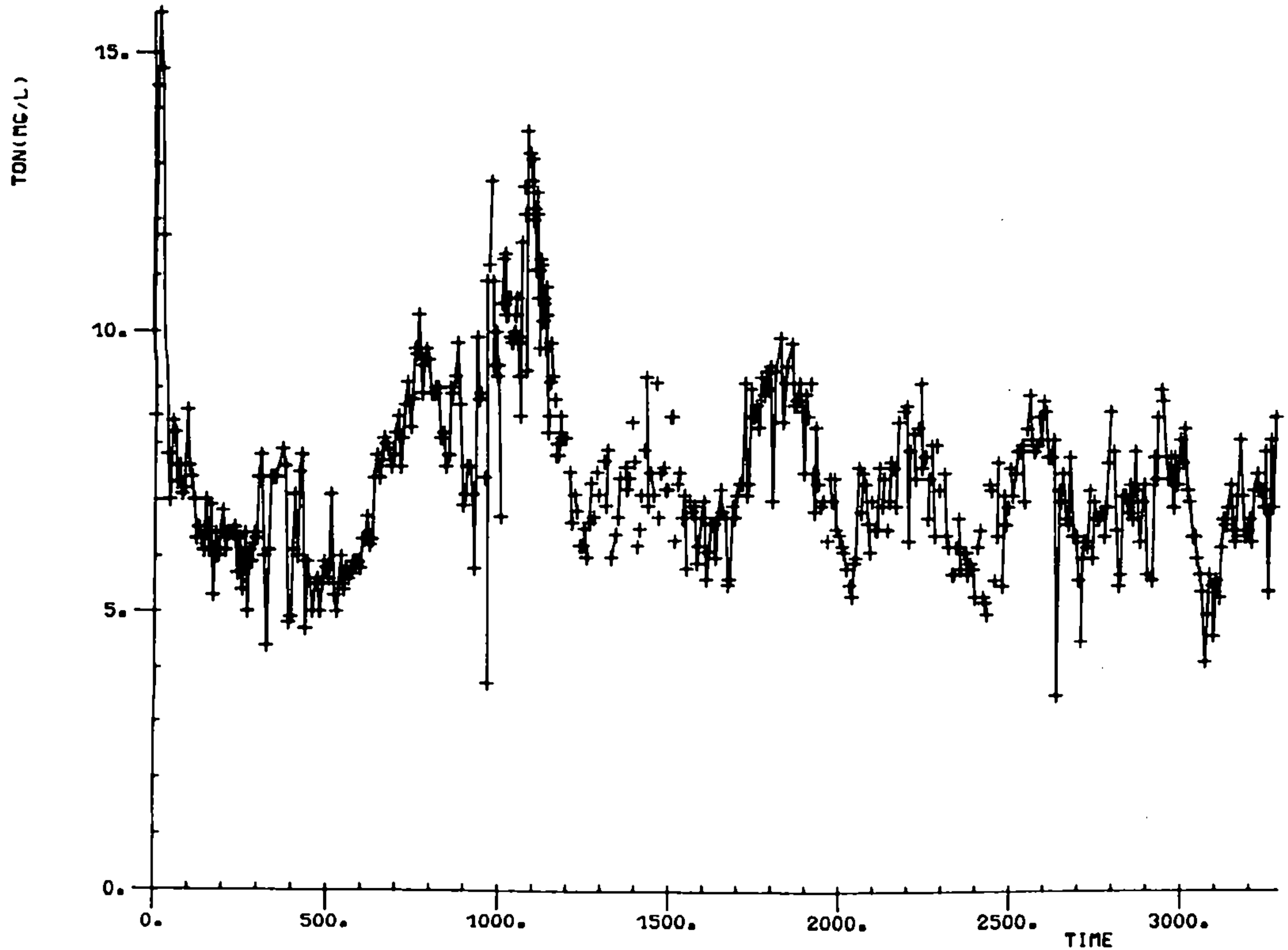
Appendix 3.1.Figure 3.

THAMES AT TEDDINGTON WEIR
2 1 74 TO 29 12 82



Appendix 3.1. Figure 4.

THAMES AT TEDDINGTON WEIR
2 1 74 TO 29 12 82



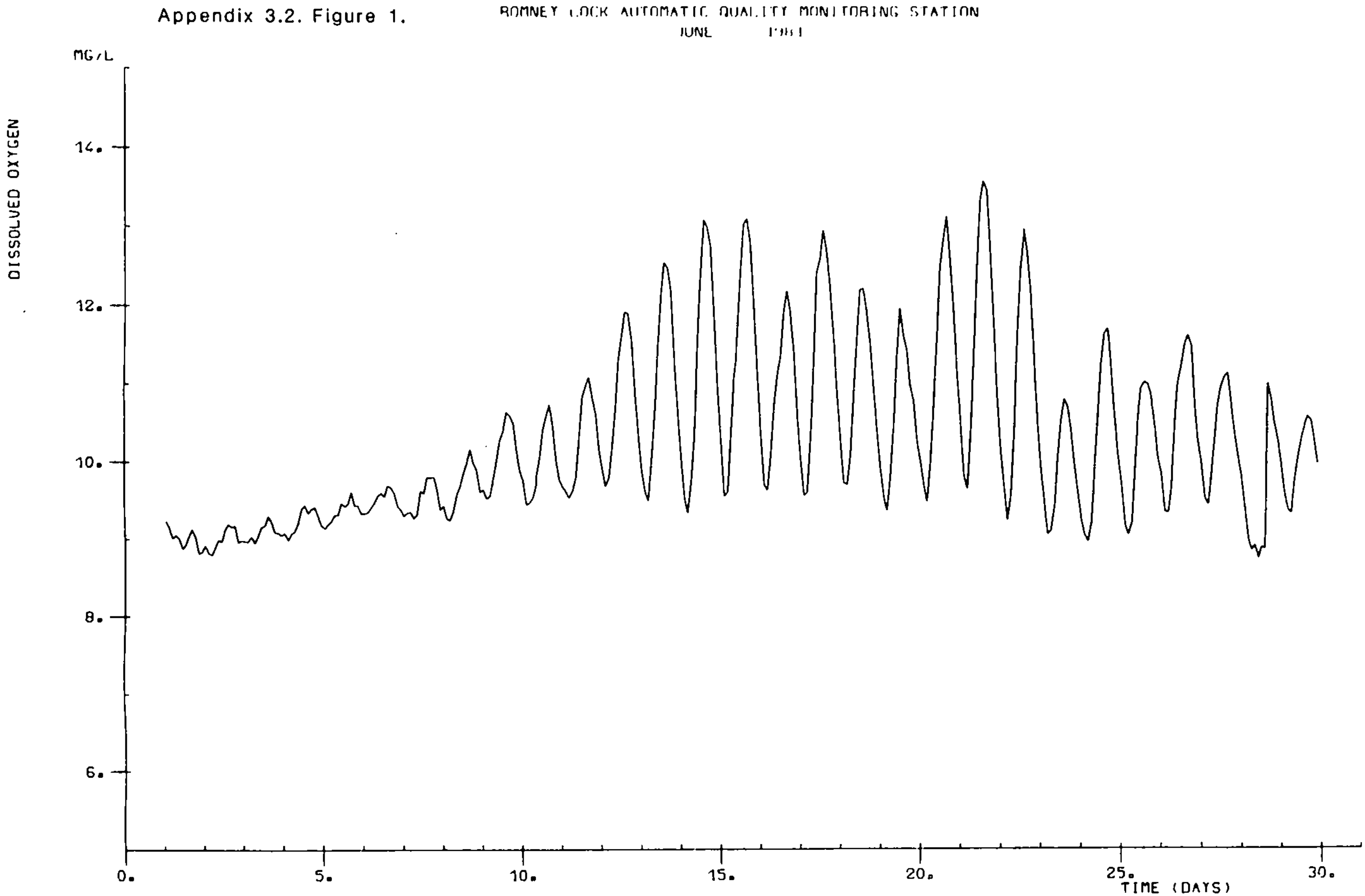
APPENDIX 3.2

DO Data - Romney Lock Automatic
Quality Monitoring Station

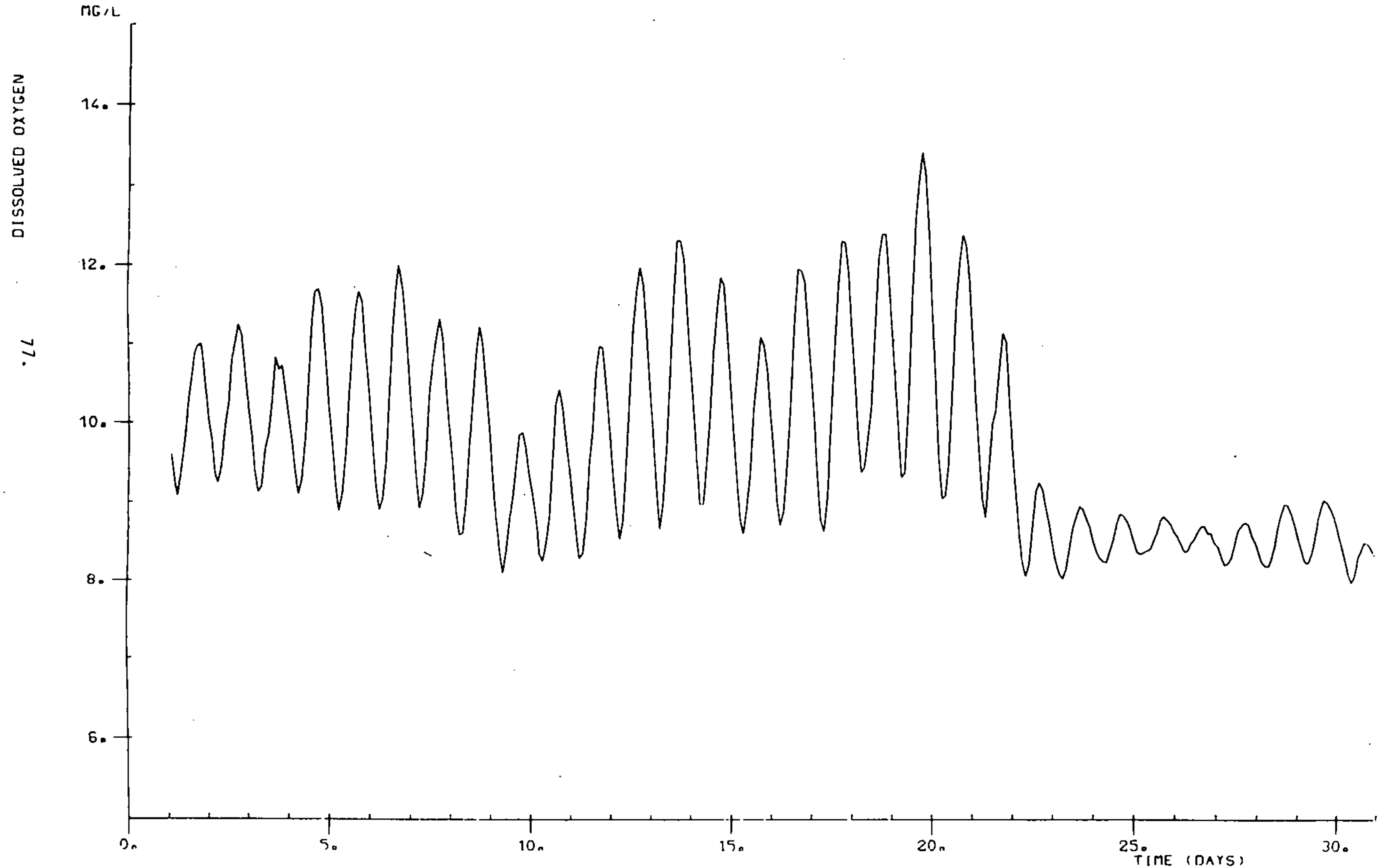
and

Bracknell Solar Radiation Data

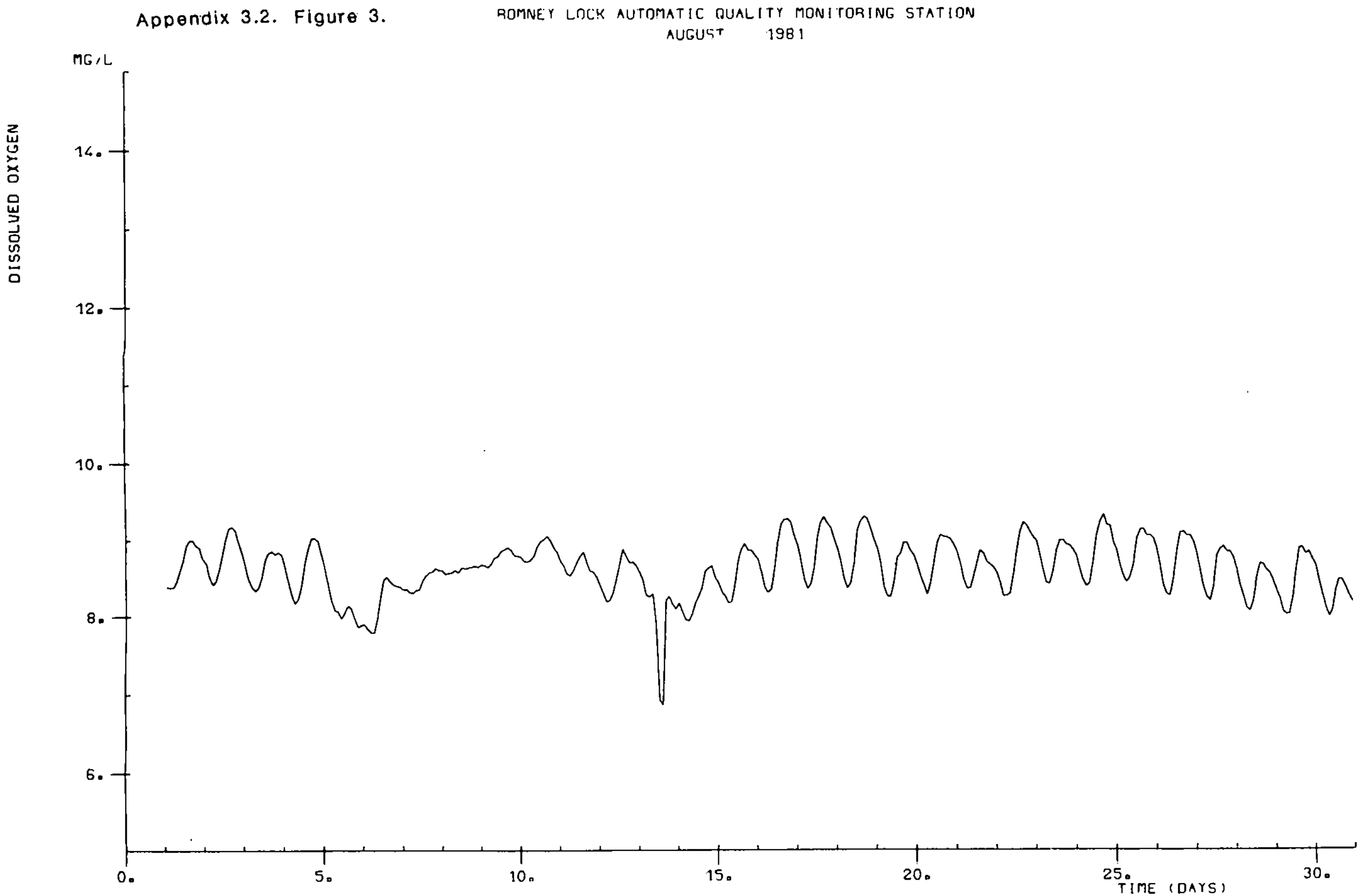
Appendix 3.2. Figure 1.



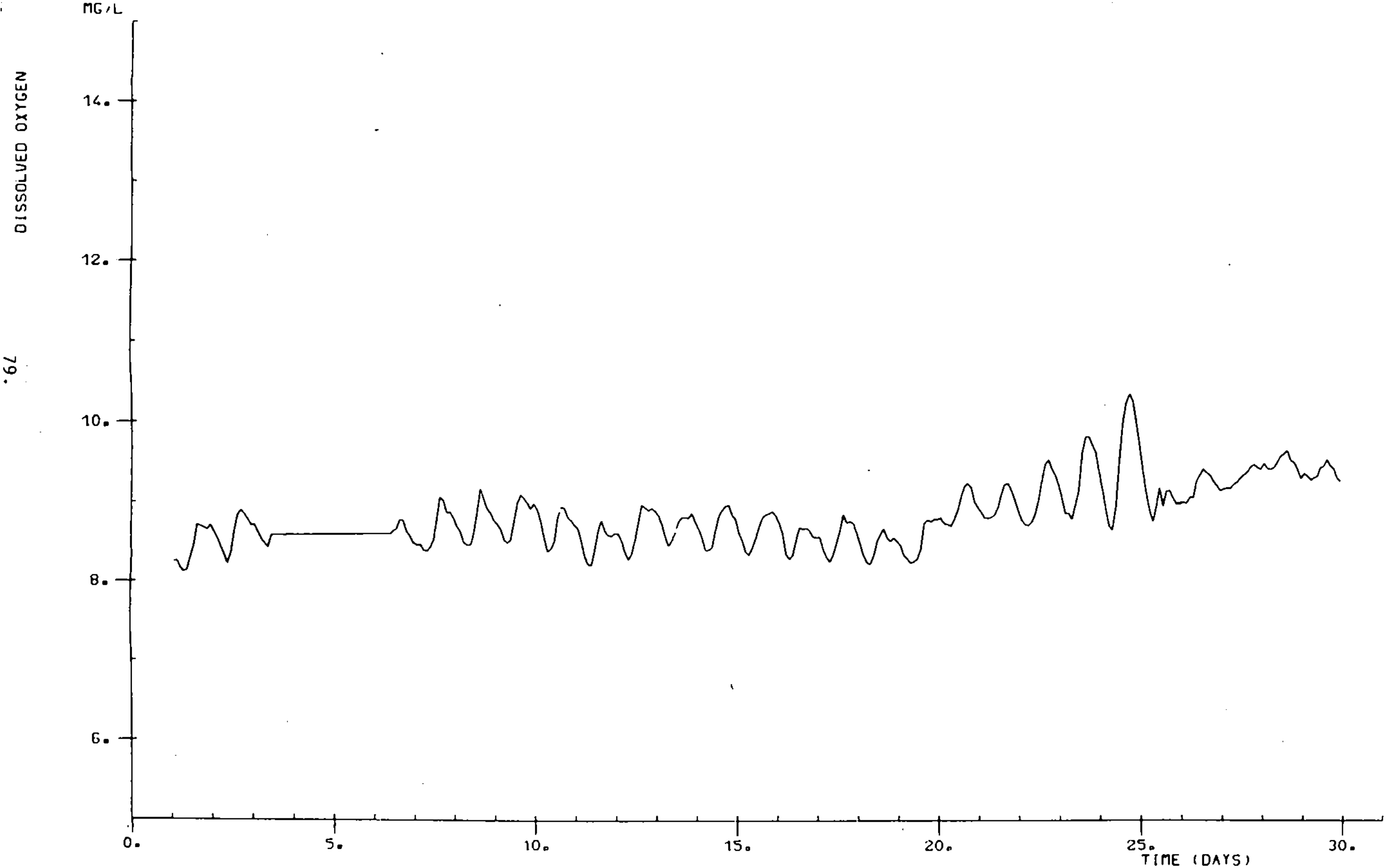
Appendix 3.2. Figure 2. ROMNEY LOCK AUTOMATIC QUALITY MONITORING STATION JULY 1981



Appendix 3.2. Figure 3.

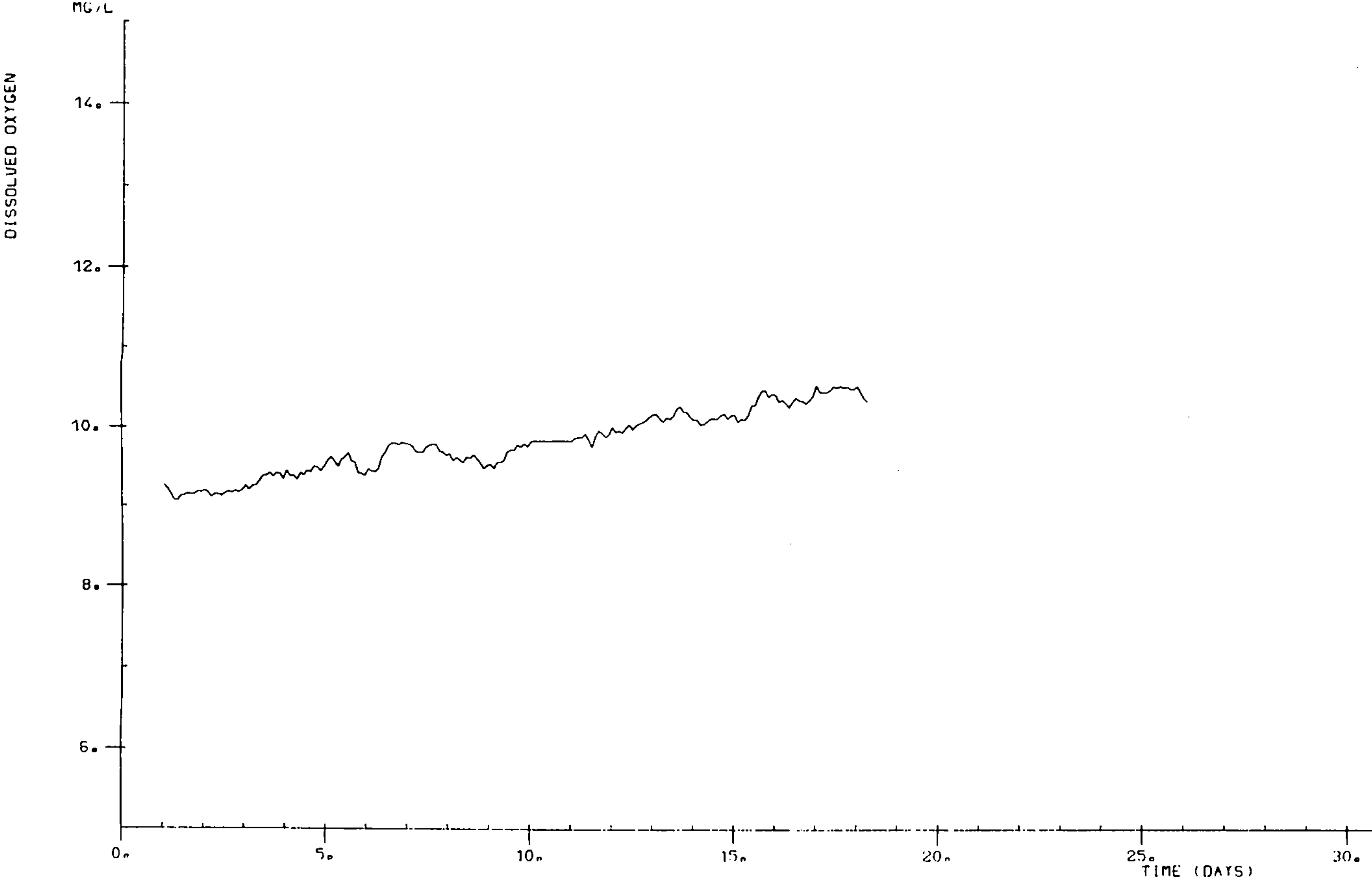


Appendix 3.2. Figure 4.   ROMNEY LOCK AUTOMATIC QUALITY MONITORING STATION
SEPTEMBER 1981

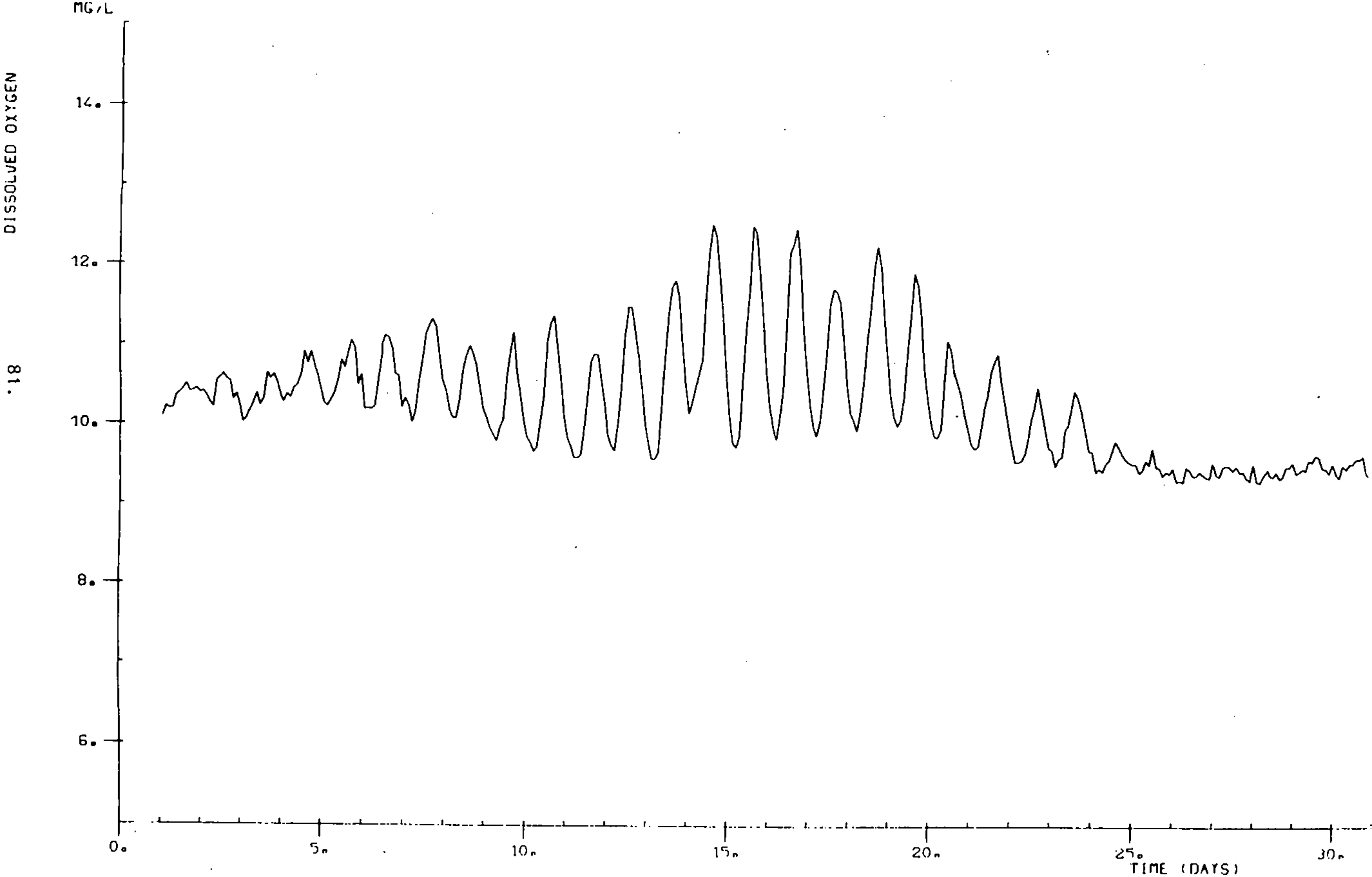


Appendix 3.2. Figure 5.

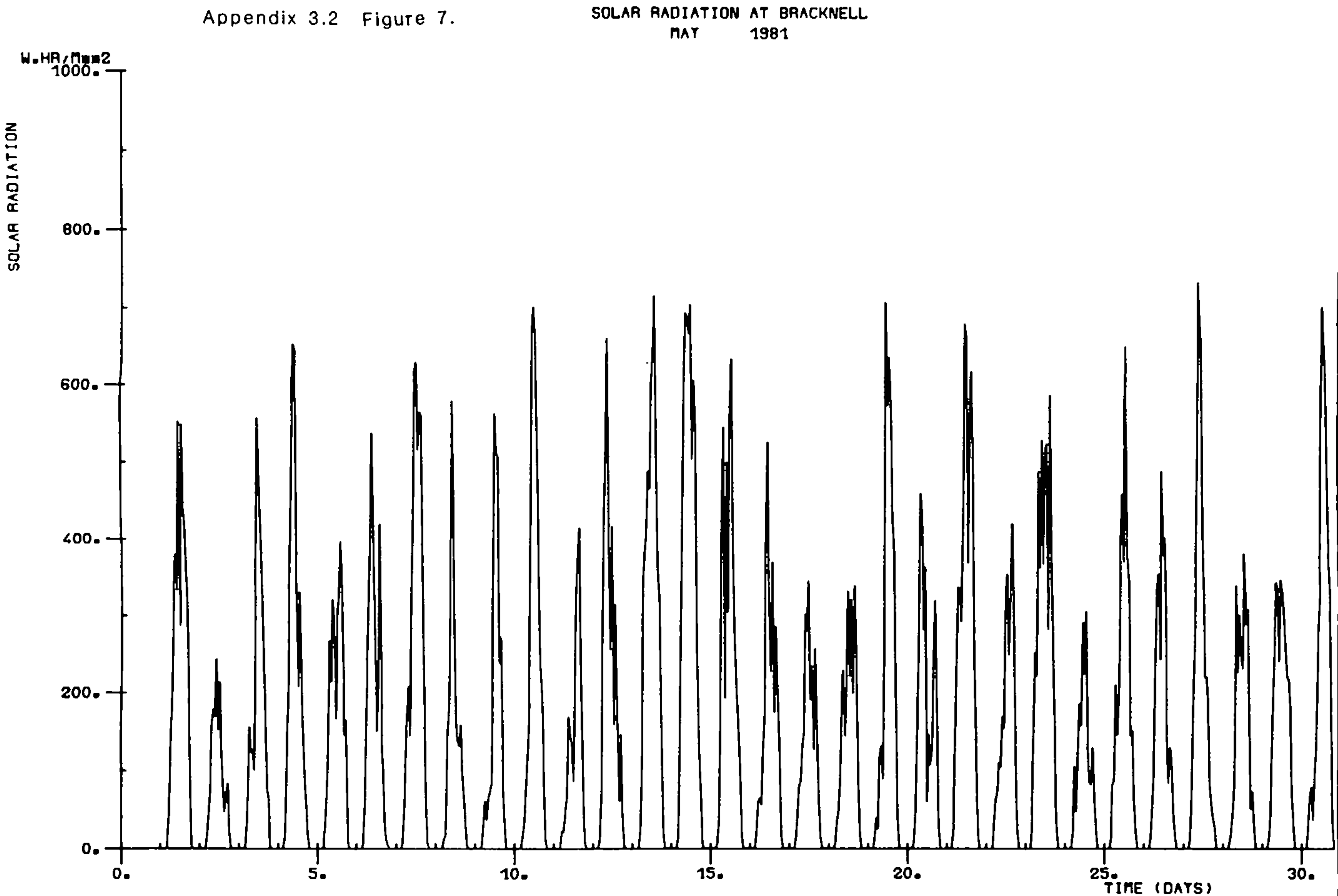
ROMNEY LOCK AUTOMATIC QUALITY MONITORING STATION
OCTOBER 1981



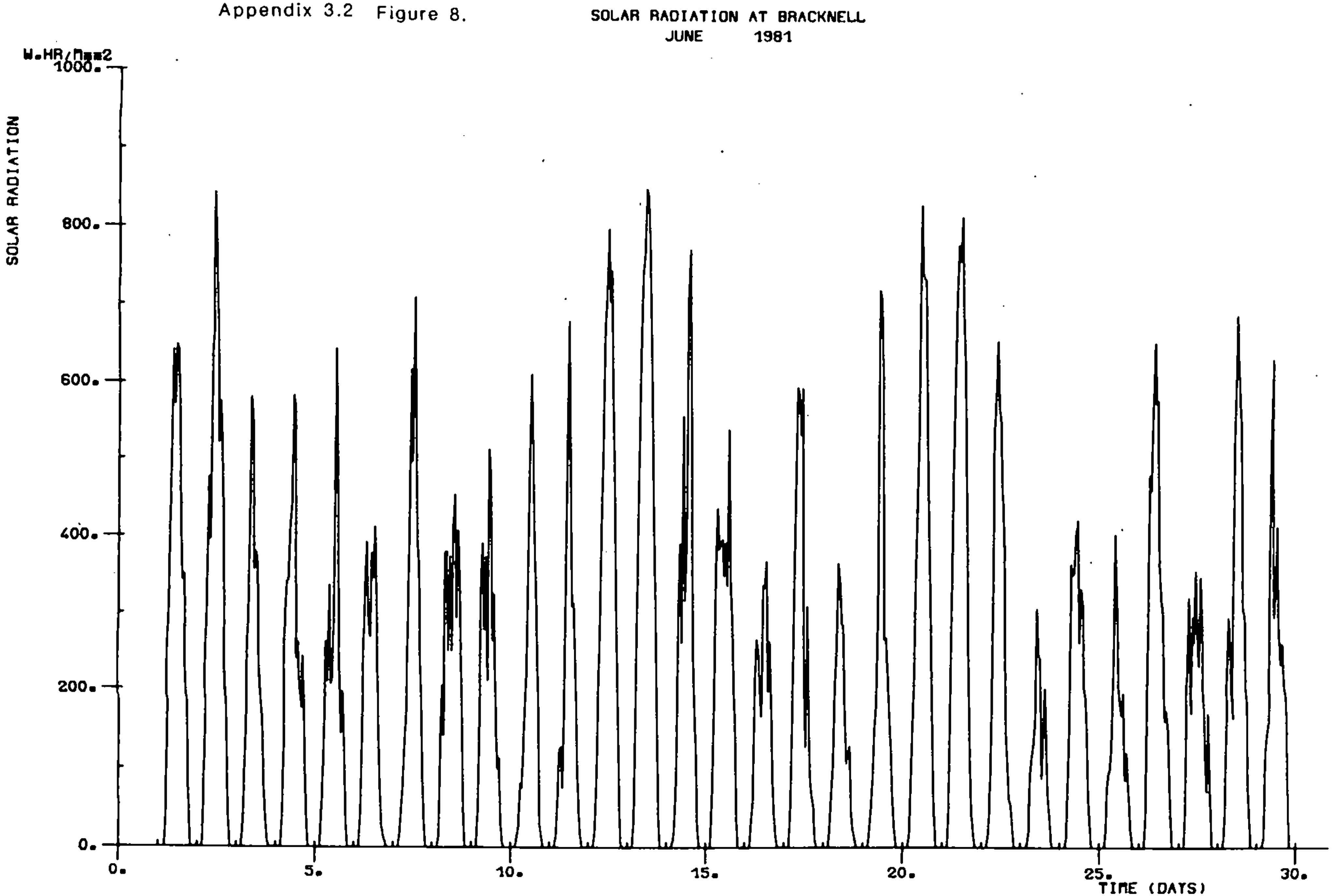
Appendix 3.2. Figure 6. ROMNEY LOCK AUTOMATIC QUALITY MONITORING STATION
MAY 1981



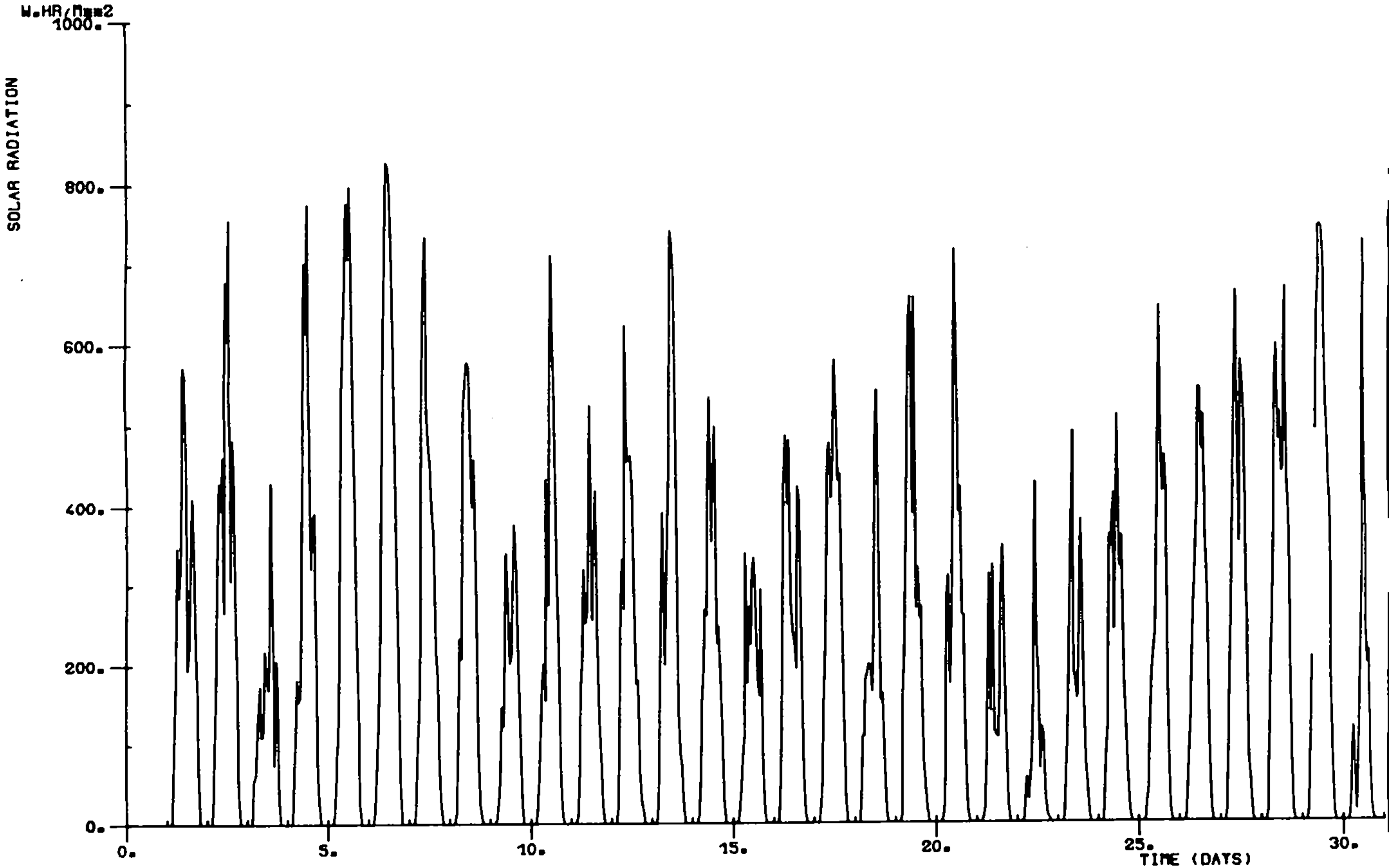
Appendix 3.2 Figure 7.



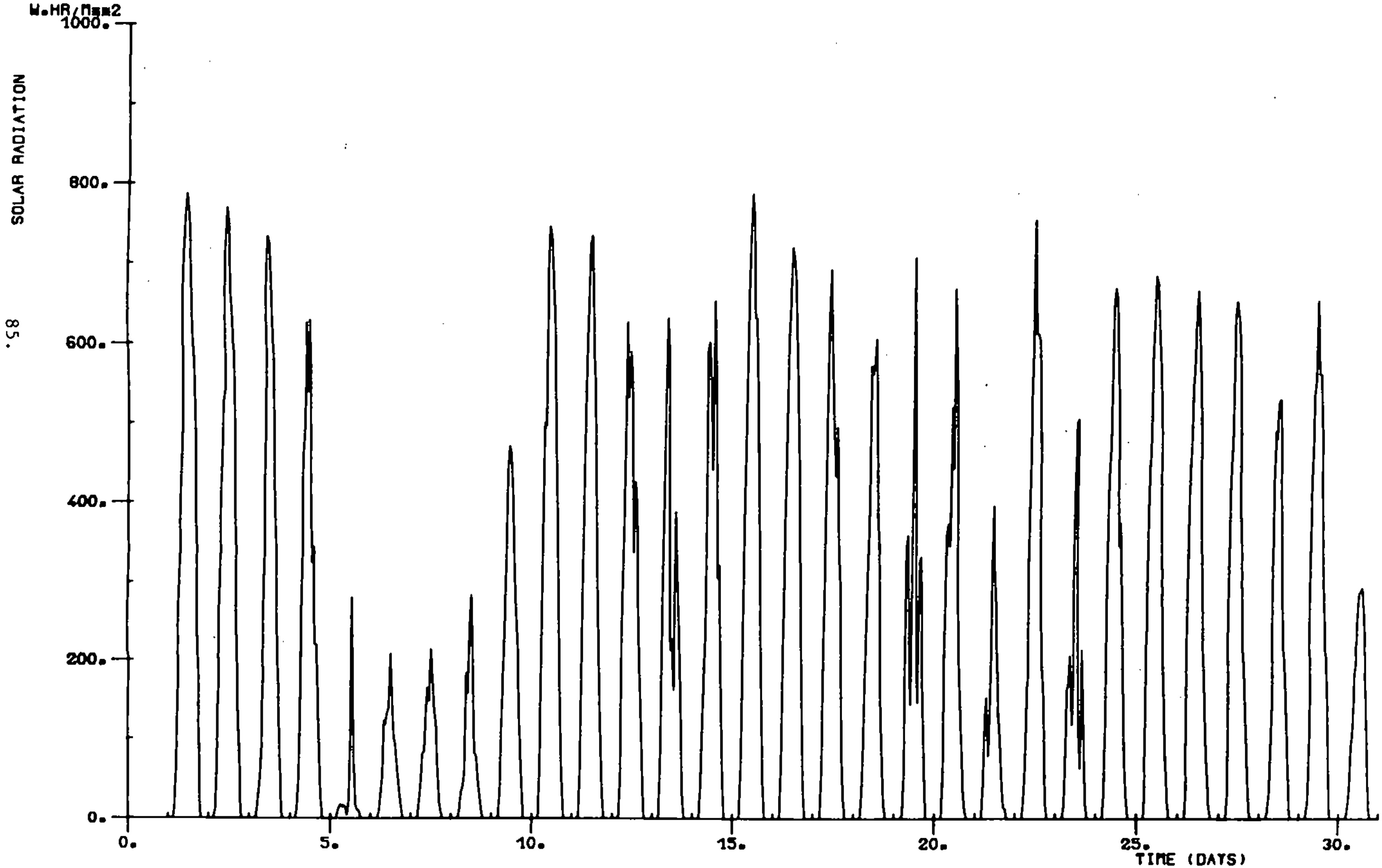
Appendix 3.2 Figure 8.



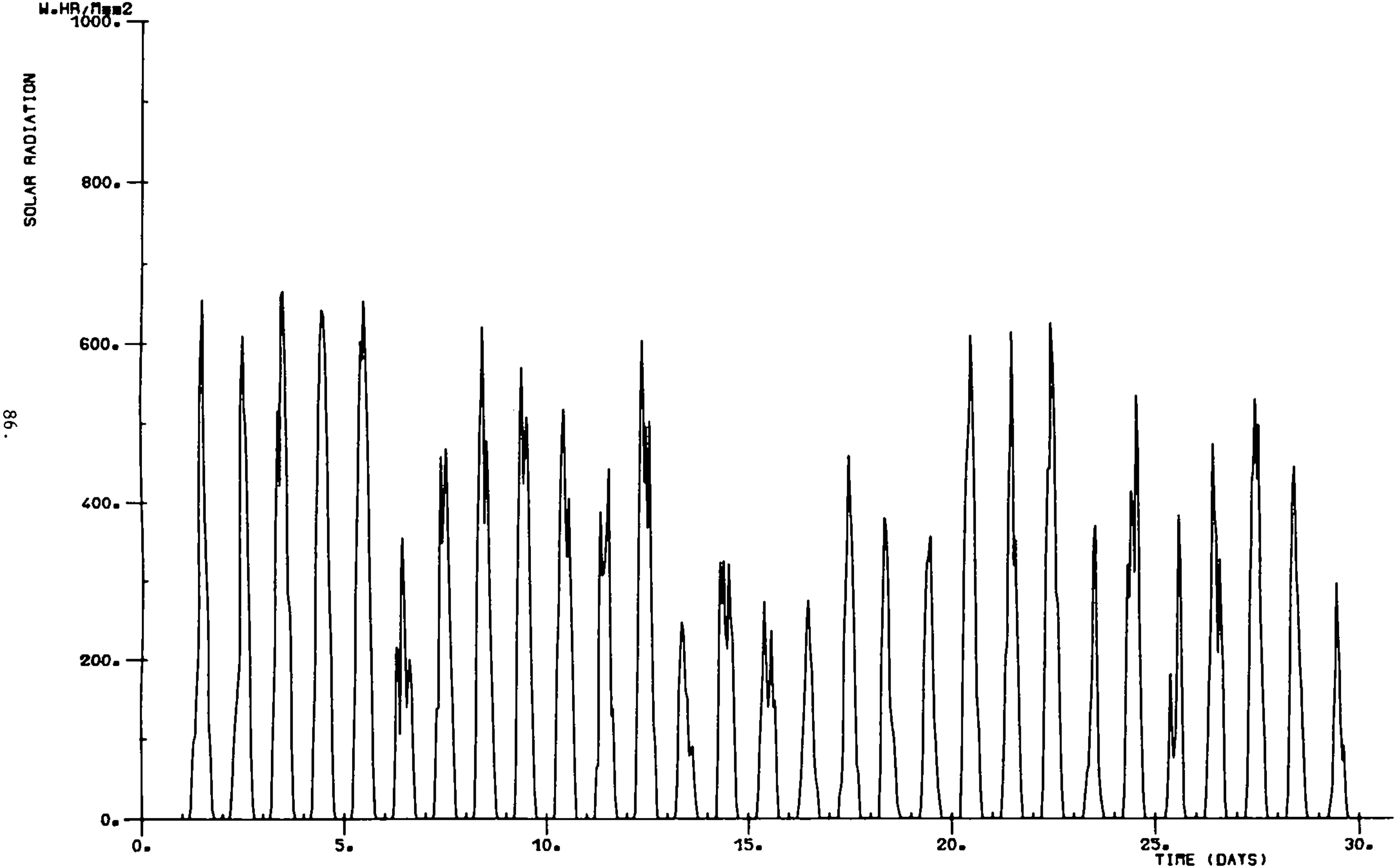
Appendix 3.2. Figure 9. SOLAR RADIATION AT BRACKNELL JULY 1981



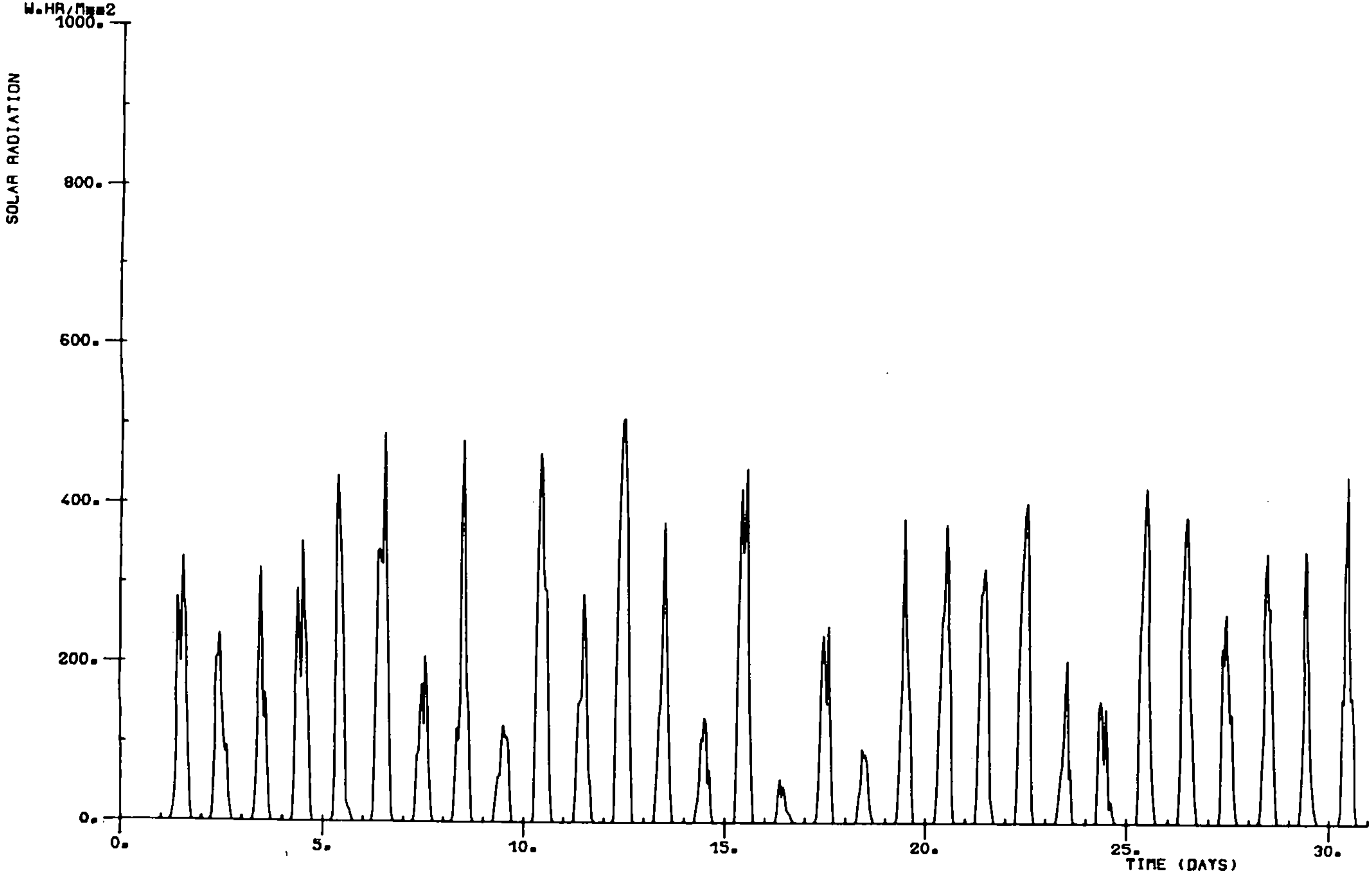
Appendix 3.2. Figure 10. SOLAR RADIATION AT BRACKNELL
AUGUST 1981



Appendix 3.2. Figure 11. SOLAR RADIATION AT BRACKNELL SEPTEMBER 1981

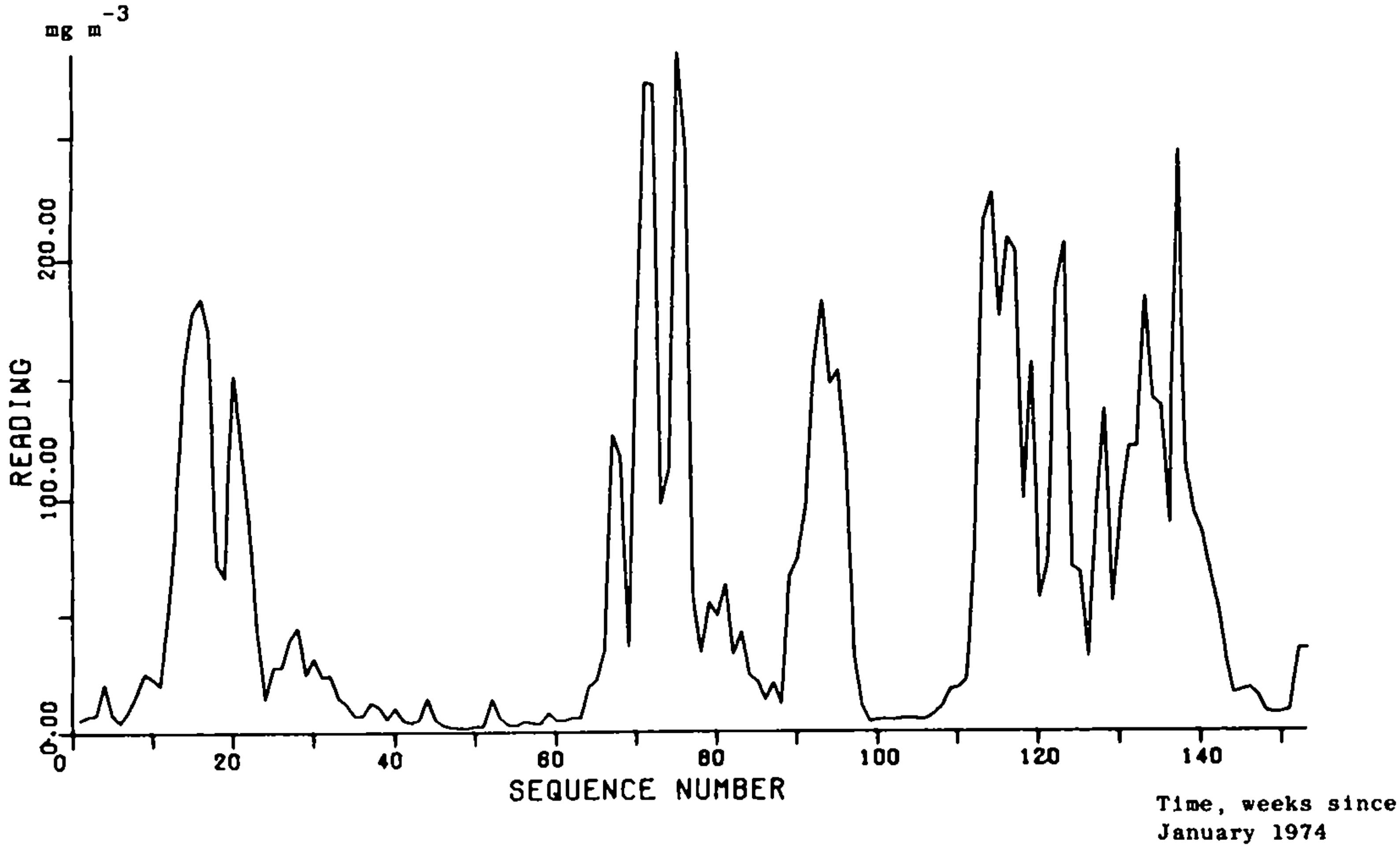


Appendix 3.2. Figure 12. SOLAR RADIATION AT BRACKNELL
OCTOBER 1981

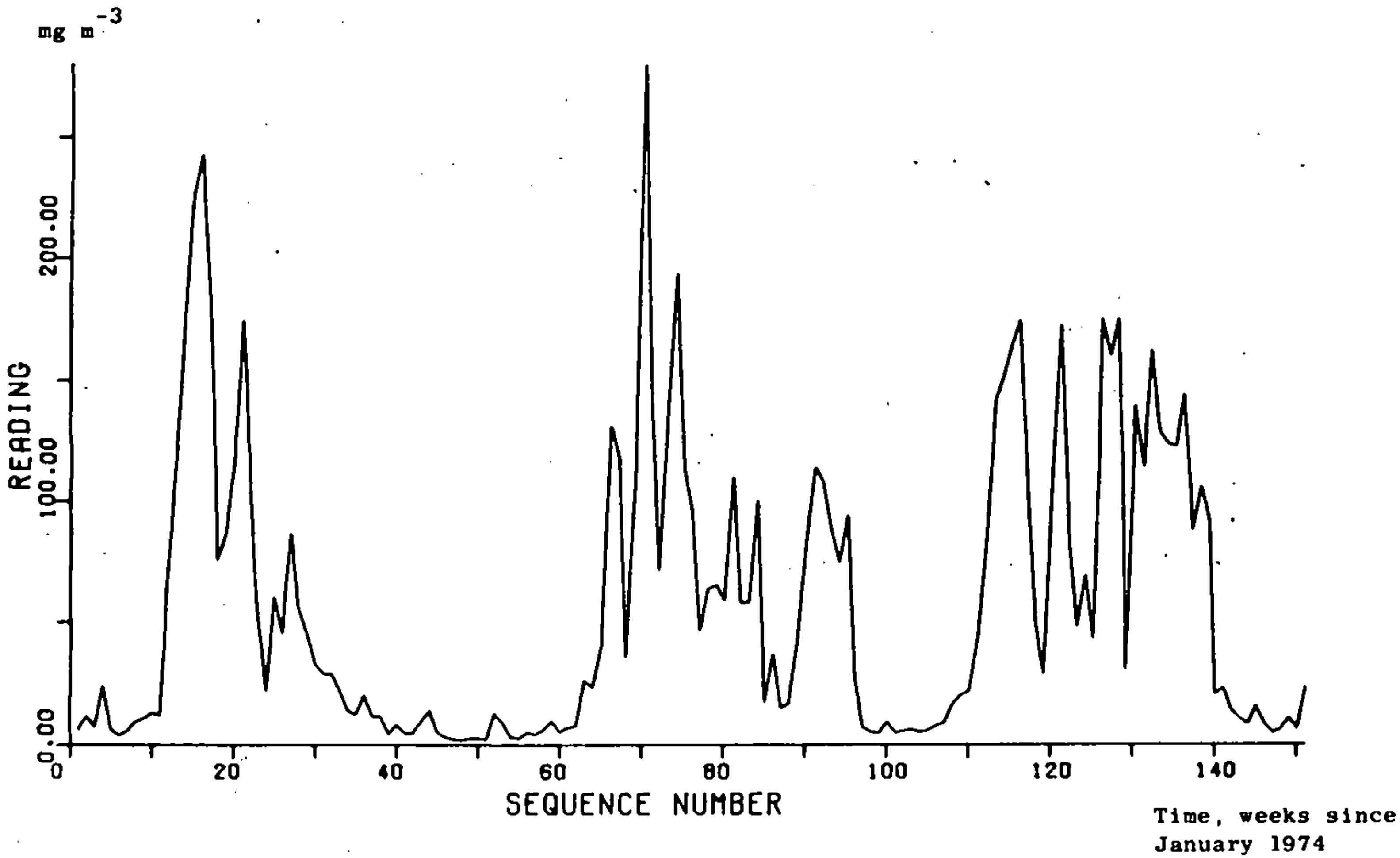


APPENDIX 3.3

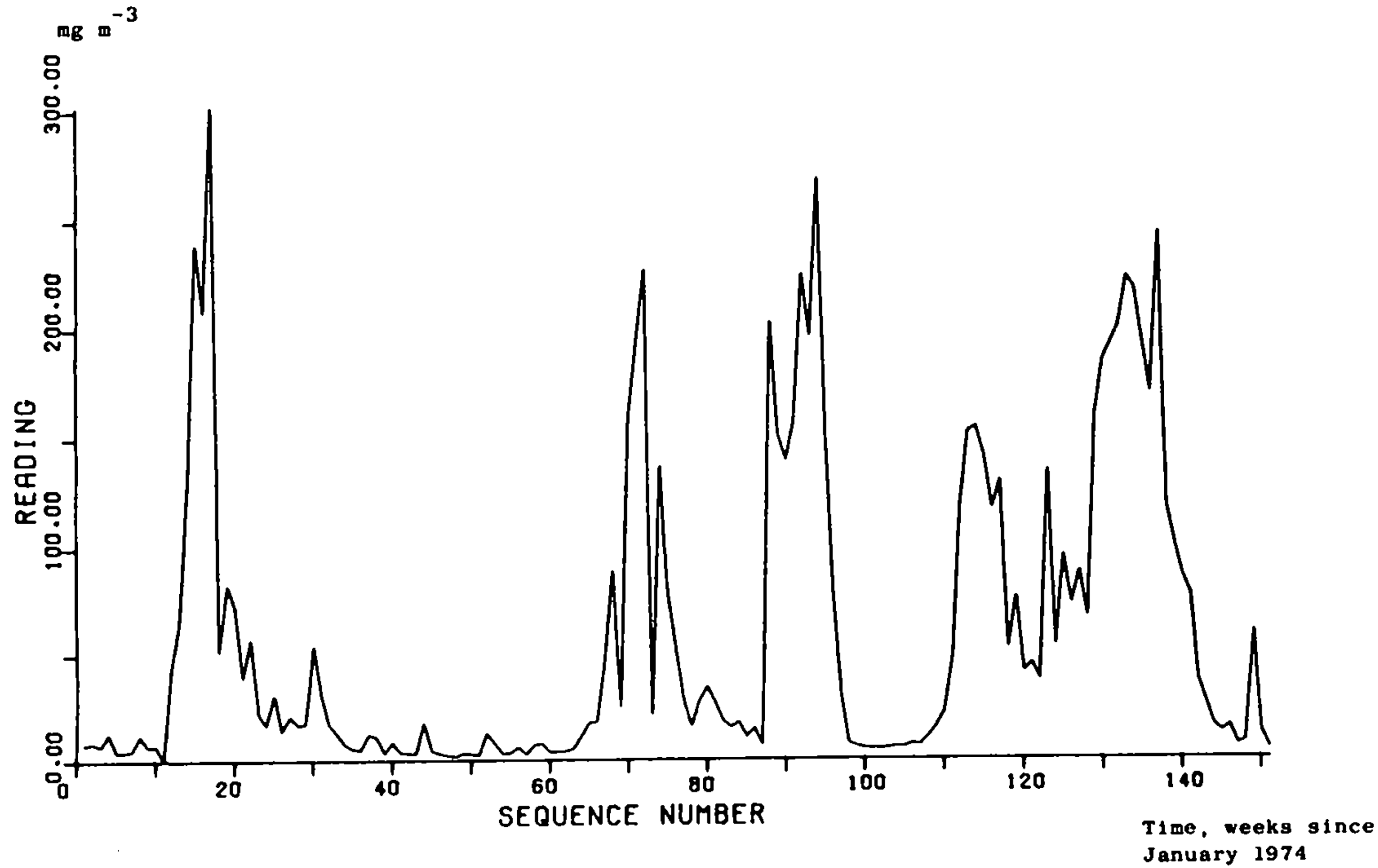
Thames River Chlorophyll a Data (1974-1976)



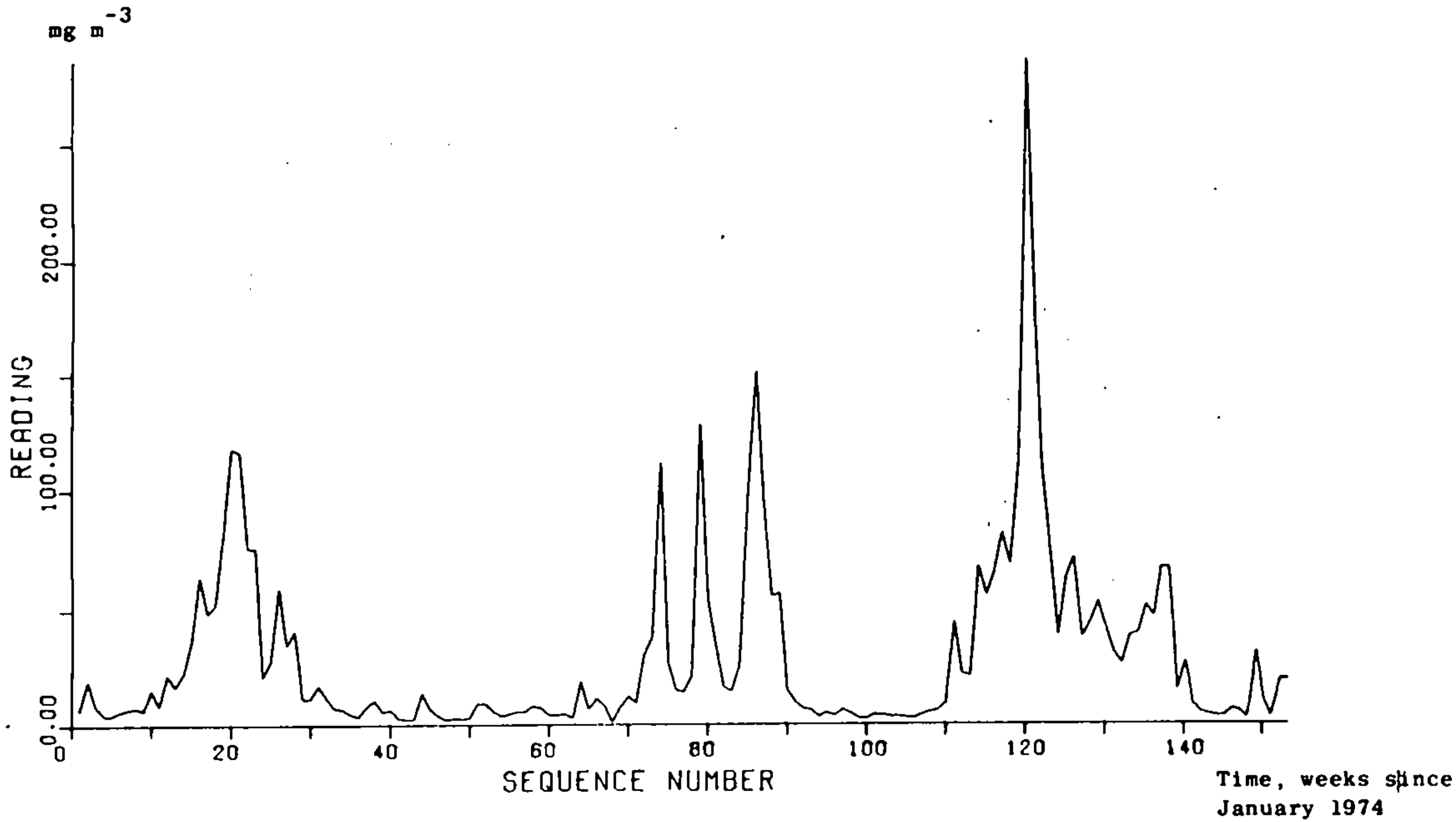
Appendix 3.3 Figure 1. Chlorophyll a at Teddington



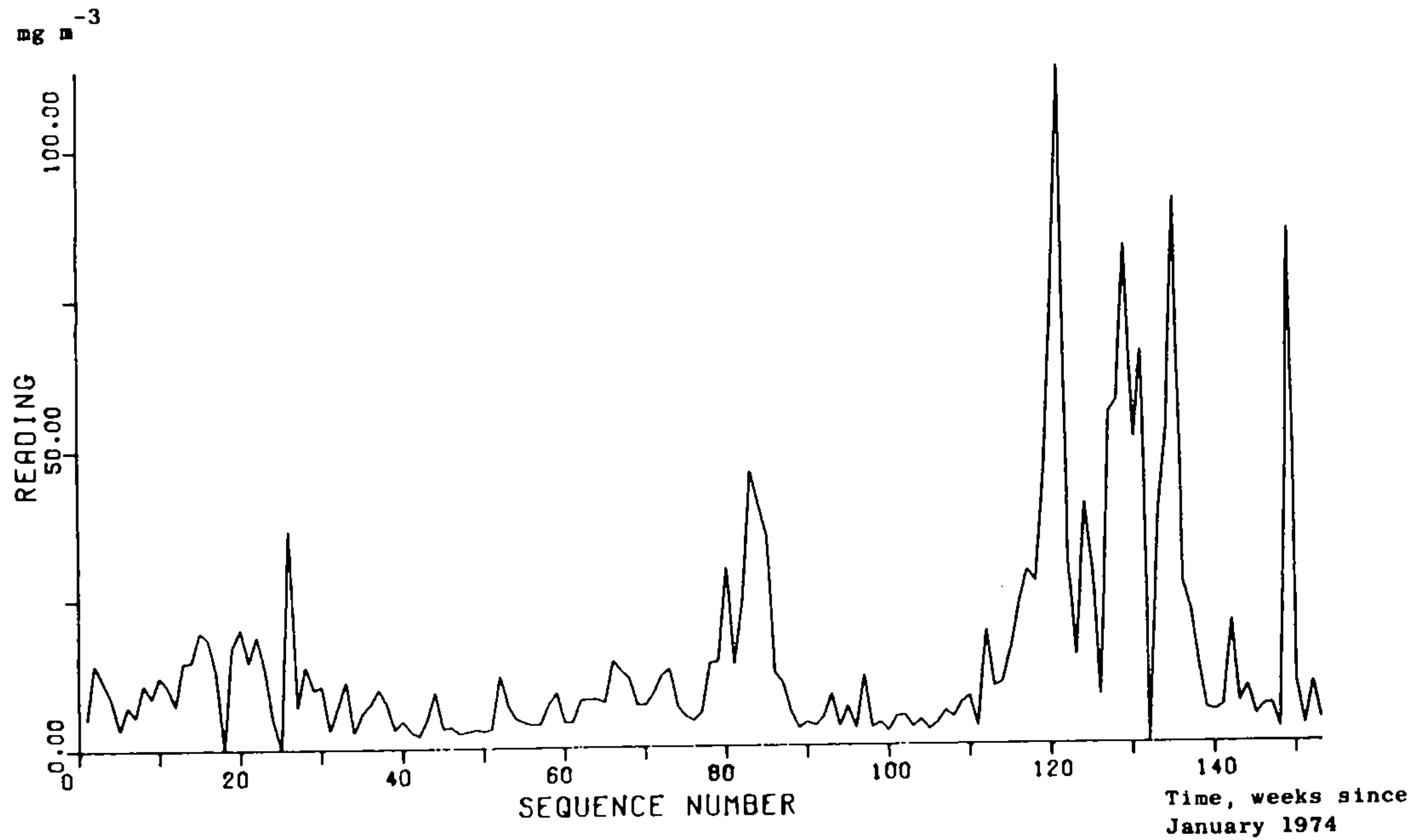
Appendix 3.3. Figure 2. Chlorophyll a at Staines



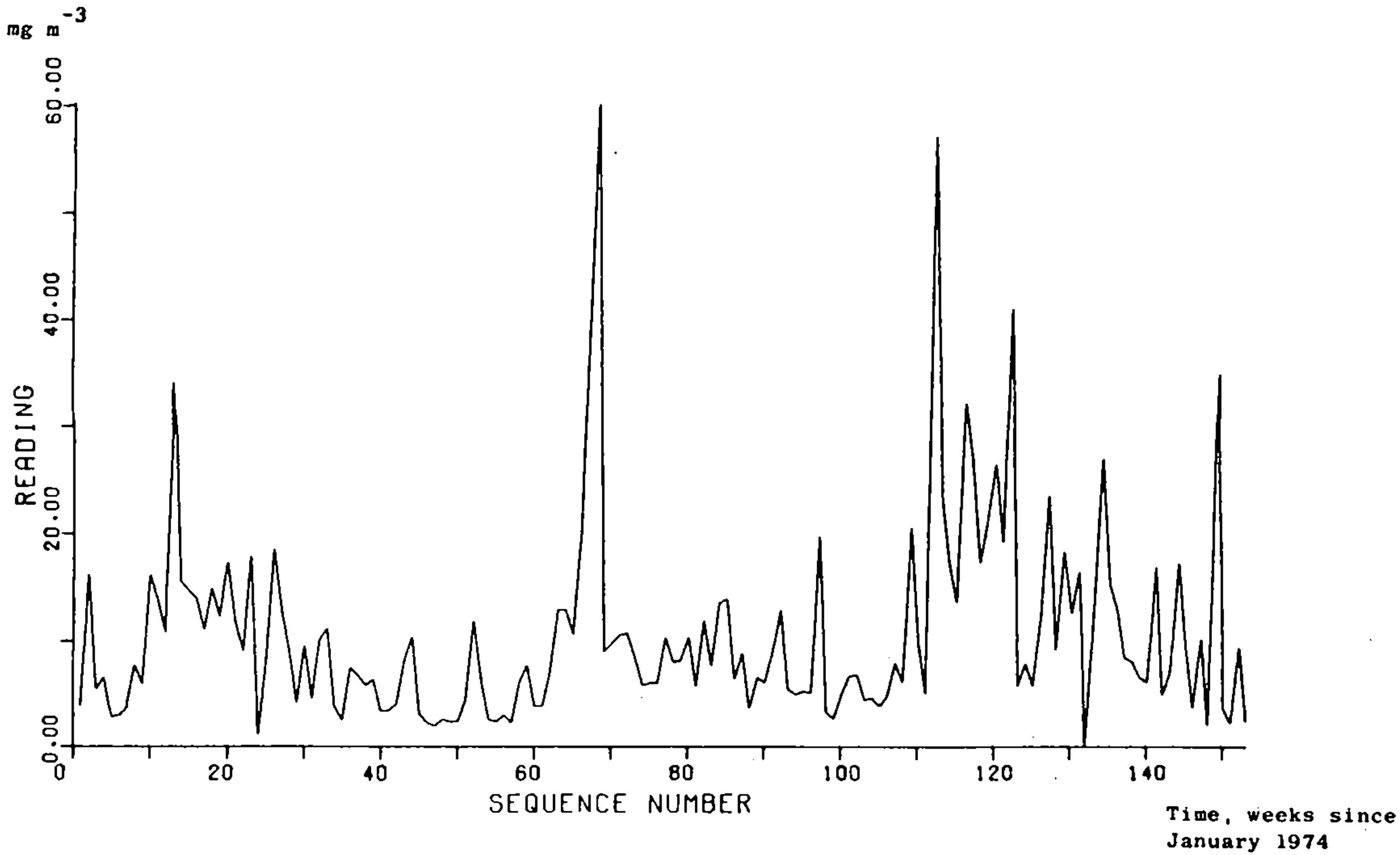
Appendix 3.3. Figure 3. Chlorophyll a at Caversham



Appendix 3.3 Figure 4. Chlorophyll a at Swinford



Appendix 3.3 Figure 5. Chlorophyll a at Buscot



Appendix 3.3. Figure 6. Chlorophyll a at Castle Eaton

4. EFFECTS ON THE WATER QUALITY OF THE THAMES ESTUARY

P.J. Radford

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4. EFFECTS ON THE WATER QUALITY OF THE THAMES ESTUARY

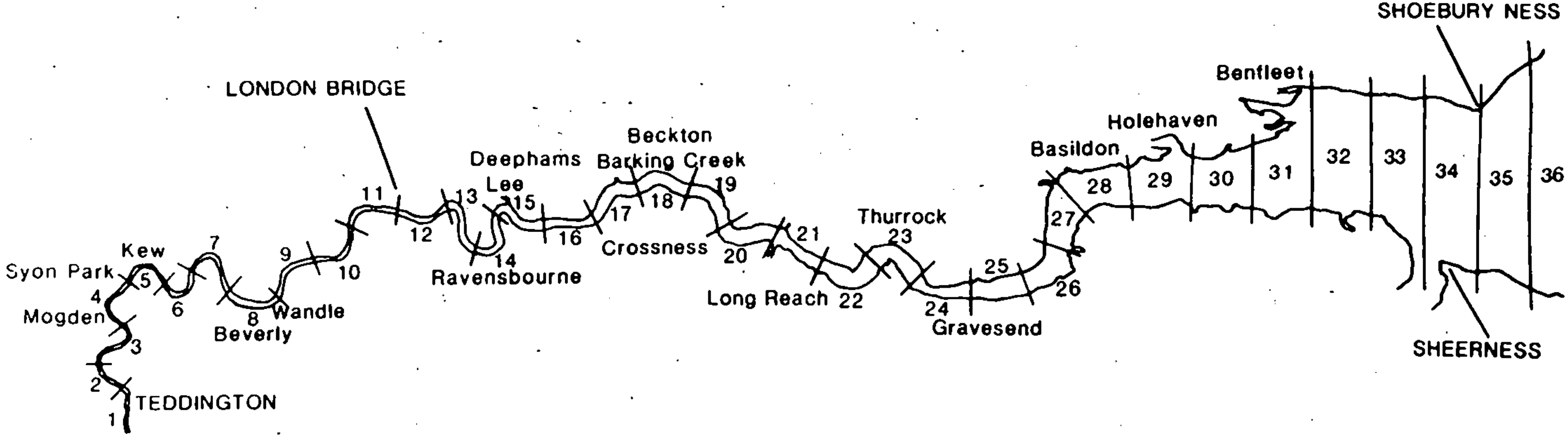
P.J. Radford

4.1 BACKGROUND AND PROGRESS

4.1.1. The water quality of the Thames Estuary has been the subject of intensive study for at least a century and a large body of information exists which enables one to view current trends in their historical perspective. In 1964 the Department of Scientific and Industrial Research published a comprehensive summary of the most relavent and reliable data then available, and this provides a most authoratative basis for contemporary study. That volume includes an explanation of a mathematical model of the water quality of the Thames Estuary which has subsequently been developed and validated by Barrett et al. (1978). This model which has the merit of comparative simplicity, has led to realistic predictions of water quality during a period where there has been a progressive reduction in polluting load (1969-1975). The Thames Model has become the father of water quality models throughout the world and many variations of it have been applied to the major British Estuaries including the Severn (Radford, 1980) and the Humber (Mallowney 1982a) and abroad,(Maskell and Odd, 1977).

4.1.2. In this study the steady state, moving segment version of the Thames Water Authority Model (Barrett et al. 1978) has been used as the basis for the construction of a simulation model to predict the effects of changes in river flow on water quality. The new model retains the 36, two-mile divisions of the estuary (slices) from Teddington to Shoebury Ness shown in figure 4.1. and uses the same average volume data. Slice 1 and slice 36 represent the boundary conditions which contribute to water quality from the Thames at Teddington, and the open sea respectively. It has been adapted to compute daily rather than quarterly distributions of fast carbon, slow carbon, fast nitrogen, slow nitrogen, ammonia, nitrate, dissolved oxygen and salinity. To achieve this it has been necessary to compute dispersion coefficients for each slice for a wide range of river flow conditions and compute the relationship between these two variables. Integration through time has been achieved using Eulers point slope technique with approximately 25 iterations per day. The partitioning of latteral inputs between adjacent slices has been done according to the moving segment technique used in the original Thames Model. The simulations are driven by daily river flow data, generated by the IH River Thames Water Quality Model which yields predictions for both present and hypothetical abstraction patterns under historic or future demands (see section 2 for general relationships between models and section 3 for details of the River Thames Water

Thames Estuary from Teddington to Shoebury Ness



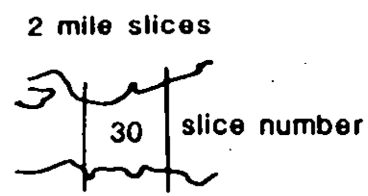


Figure 4.1 The 36 x two mile slices used in the Esturine Water Quality Model

Quality Model). Inputs of contaminants are based on measured discharges from all of the major sewage works and lateral rivers and are expressed on a quarterly basis for the validation of historic flow sequences or annually for hypothetical conditions.

4.1.2.1. The progression of this study has been as follows. The daily simulation model was calibrated using 1977/78salinity data and validated against the observed weekly salinity distributions for 1975/76. The validated model was then used to measure the predicted effect on water quality of the current (1984) water demand and improved water treatment. These changes provide a yardstick against which the importance of possible proposed changes might be judged. A comparison was then made between the current system of abstraction control based upon "the Chart" method and an alternative strategy known as the maximum resource benefit (MRB) method. This comparison was made for a number of historic river flow sequences, chose to be representative of the range of flow conditions experienced over the years. The period 1975/76 was chosen as a classical drought sequence and 1944/45 as the most severe drought in recent history; 1952/53 could be considered as a typical non-drought sequence and 1980/81 as representing recent high flow conditions.

4.1.2.2. Another important consideration has been the effect of possible increased water demand on both the Chart and the MRB regimes. In considering what future demands to assume, Thames Water have estimated that if no additional resources such as new reservoirs were introduced, the reliability of the water supply to London would have reached an intolerably low level by the year 2006; further resources would therefore need to be introduced by that date. The forecast demand for 2006 were thus chosen for assessment (without additional reservoirs etc) as this represents the maximum possible demand on the river and therefore the "worst case" situation. The Thames Water demand forecast for the London area for 2006 is equivalent to 117% of the current (1984) demand.

4.1.2.3. Changes in flow conditions are known to influence the location of the areas of mud deposition and resuspension. A small statistical study has therefore been made to gauge the importance of this factor for the ecosystem.

4.2 DESCRIPTION OF THE THAMES ESTUARY

4.2.1. The Thames Estuary (Figure 4.1.) begins at the limit of tidal influence at Teddington Lock. At the Weir the majority of the residual flow of the River Thames cascades into the esturine water at a rate which normally ensures a reach of completely fresh (non-saline) water down to Richmond Half Tide Lock.

The water level in this 6 km reach is maintained at an artifically high average level by closing the Lock for two hours each side of high water. Just below Richmond the estuary receives a considerable input of water which is discharged from the Mogden Sewage Works. This water is clearer and less salty than the main body of esturine water and the combined Teddington, Mogdon and a number of other major sewage works ensures that the salinity of the estuary down to London Bridge is below one part per thousand except in drought years. The largest single sewage input is from the Becton discharge which enters the estuary near Barking Creek, some 50 km from Teddington Weir. At this point the salinity of the estuary is about 15 parts per thousand. The estuary opens out from this point allowing increasing mixing with the sea water until, at Shoebury Ness its salinity reaches 35 parts per thousand which is close to normal North Sea salinities.

4.3 A REVIEW OF EXISTING WATER QUALITY DATA

4.3.1. Introduction

A large body of data exists which relates to the water quality of the Thames Estuary. The vast majority of this data is in manuscript form and refer to measurements made between 1922 and 1976. The more recent data (1977 to 1984) are stored in the Thames Water Data Archive in a computer retrievable form. The following determinands have been measured at various frequences over this time period and data summaries can be obtained using the Data Archive computer programs.

i) Ph value
ii) Conductivity
iii) Total Solids
iv) Suspended Solids (105°c)
v) Suspended Solids (500°c)
vi) BOD
vii) Chemical Oxygen Demand
viii) Total Organic Carbon
ix) Temperature °C
x) DO mg l^{-1}
xi) DO (% Saturation)
xii) Chloride as Cl^-
xiii) Ammoniacal Nitrogen
xiv) Un-ionised ammonia as N
xv) Nitrate as N
xvi) Nitrite at N
xvii) Total Oxidised Nitrogen as N
xviii) Organic Nitrogen

Most of these determinands are recorded on a weekly basis being measured at some 27 locations along the axis of the estuary from Teddington to Shoebury Ness (Table 4.1). Other less common determinands are measured

TABLE 4.1. THE RELATIONSHIP BETWEEN SLICE NUMBER, DISTANCE FROM LONDON BRIDGE (MILES) DISTANCE FROM TEDDINGTON (Km) AND THE SAMPLING ZONES USED IN THE WEEKLY ESTURINE SURVEYS BY THAMES WATER ARE SHOWN IN THIS TABLE.

Slice No.	Miles from London Br. of mid point	Kn. from Teddington of mid point	Thames Water Sampling Zone No	Km. from Teddington of mid point	Identification
2	-19	0.5	1	0	Teddington
3	-17	3.7	2	4	
4	-15	6.9	3	9	Syon Park
5	-13	10.1	4	12	Key
6	-11	13.4	5	15	
7	-9	16.6	6	18	R. Beverly
8	-7	19.8	7	22	
9	-5	23.0	8	26	
10	-3	26.2	9	28	
11	-1	29.5	10	31	London Bridge
12	+1	32.7	11	34	
13	3	35.9	12	36	
14	5	39.1	13	40	R. Lee
15	7	42.3	14	44	
16	9	45.5	15	47	
17	11	48.8	16	50	Barking
18	13	52.0	17	53	
19	15	55.2	18	58	
20	17	58.4	19	62	
21	19	61.6			
22	21	64.9	20	67	Thurrock
23	23	68.1			
24	25	71.3	21	72	
25	27	74.5	22	77	
26	29	77.7			
27	31	81.0	23	82	Basildon
28	33	84.2			
29	35	87.4	24	87	
30	37	90.6	25	92	Benfleet
31	39	93.8			
32	41	97.0			
33	43	100.3	26	100	
34	45	103.5			
35	47	106.7	27	111	Shoebury Ness

from time to time. In addition a number of coninuous monitoring stations have been set up which measure dissolved oxygen, temperature and salinity at Chelsea, Tower Pier, Crossness and Perfleet. The computerised Data Archive provides more than enough data for the calibration and validation of models of water quality such as developed by WRC and IMER.

4.3.2. Data for modelling

Mathematical models require three types of data for their sucessful implementation.

a) Time series data for driving the model e.g. Historic river flows

b) Time series data for validating a model e.g. Salinity and DO.

c) Parameter values for specifying process equations e.g. Dispersion coefficients.

Daily river flow data at Teddington Weir were available on the Data Archive to drive a model for all years between 1920 and 1981. The Thames Water Quality model also requires data of salinity, ammonia, BOD, TON and DO, and the 27 axial samples taken weekly provide sufficient discrimination in space and time. Salinity data are also essential for the estimation of estuarine dispersion coefficients, and because these are known to be correllated with river run-off, a number of salinity distributions are required to represent the whole range of run-off conditions. Subsequently, a model, calibrated with one set of salinity data, needs to be verified against the same data set and then validated against an independent data set. In each case the Data Archive provided sufficient data for modelling requirements.

4.4 A REVIEW OF DO-BOD MODELS FOR ESTUARIES

4.4.1. The Thames Model (Barrett et al. 1978) was developed in the 1950's and used for predicting the effects of pollution in the Thames Estuary over a period when the estuary was undergoing a considerable change through increasing treatment of sewage effluent. A quarterly steady-state model was therefore appropriate as a coarse management tool for indicating the effectiveness of anti-pollution measures. Barrett et al. clearly state the purpose of this model and describe the equations which specify the biological and physical processess involved in detail. They conclude by presenting a validation of the model based upon quarterly data for the years 1969, 1971, 1973 and 1975 respectively. The agreement between predicted and observed distribution of DO, ammoniacal nitrogen and oxidized nitrogen is excellent and both clearly demonstrate the improvement in the water quality of the estuary over that time period.

4.4.2. The same basic equations have been adapted by Maskill and Odd (1977) of Hydraulics Research, to allow for simulations over individual tidal cycles as well as for the computations of dynamic equilibrium conditions attained over consistant spring/neap cycles. More recently the Water Research Centre have delveloped a model of the Humber based upon a Hydraulics Research Model for the prediction of esturine hydrodynamics (Mallowney 1982a). The program predicts water levels throughout the system under varying conditions of tide and freshwater flow. These heights are used to determine the variation with time of surface width and cross-sectional area and this information is used as the input to a computer model of water quality (Mallowney 1982b). These ideas were further extended by a combined project involving IMER and Hydraulics Research (Hydraulics Research 1983) under the terms of a contract to model Tolo Harbour Hong Kong. The resultant model was a three-dimensional tidally averaged water quality and ecosystem model. It included all the water quality equations of the Thames Model, an extra term to allow for the oxygen demand of deposited mud and an ecosystem sub-system model based on a general ecosystem model of tghe Bristol Channel and Severn Estuary (Radford and Joint 1980).

4.5 THE THAMES WATER QUALITY MODEL

4.5.1. The Steady State Model.

The paper by Barrett et al. which describes the processes included in the Thames Model in detail has been reproduced as an appendix to this report by kind permission of the authors, Pergamon Press and the International Association on Water Pollution Research. A flow diagram which gives the chief state variables and their interrelationships is given as Figure 4.2. The equations which represent this system formed the basis of a steady-state model, written in Fortran by Mallowney, which was given to Thames Water as a tool to aid estuarine management. This computer program allows for considerable flexability as to the length, number and volume of the notional slices into which the estuary may be divided and as to the magnitude and sitings of the inputs of discharging loads of BOD, nitrate and ammonia. In practice it is a major undertaking to obtain accurate estimates of esturine volumes so the model has always used the original data which applies to a series of two mile long slices measured in both directions from London Bridge (see figure 4.1.) The confusion caused by the resultant mixture of imperial and metric units is partially remedied by

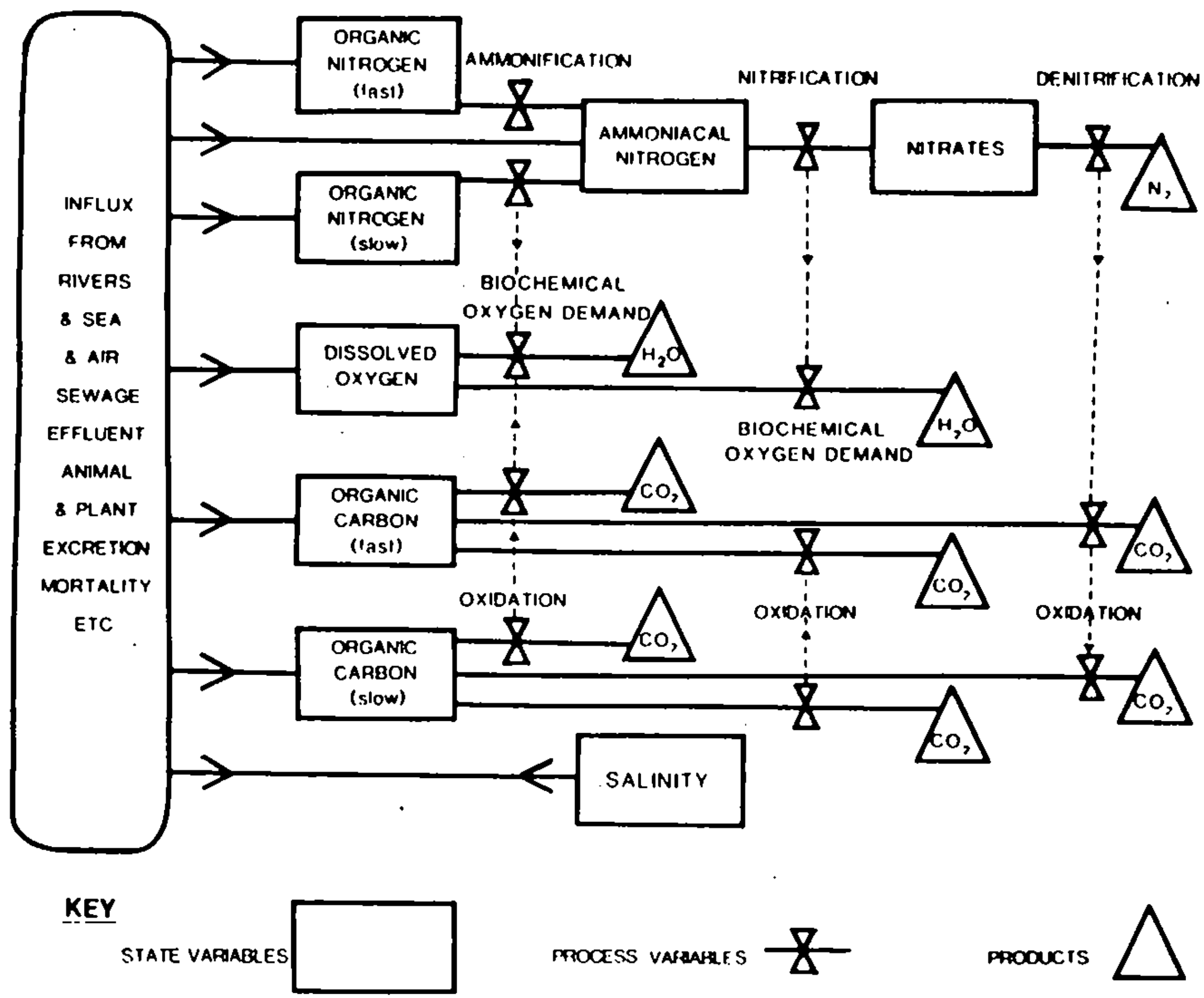


Figure 4.2 A flow diagram of the pricipal proceses included in the Estuarine Water Quality Model .

reference to Table 4.1. This implies that there are a maximum of 34 slices from Teddington to Shoebury Ness, or 36 if we include the boundary slices of the river just above Teddington and the sea just below Shoebury Ness. Most of these volumes were available from the original Thames Water Quality Model but it was necessary to calculate the volumes of slices Nos. 2 and 3 from charts provided by the Port of London Authority (1972). The steady state model had already been validated by Barrett et al. (1978) and the measure of agreement between observed and estimated distribution of DO, Oxidised nitrogen and ammoniacal nitrogen may be judged by reference to Figure 2 of Appendix 4.1. It was judged that these quarterly steady state solutions had insufficient discrimination in time to detect the small effects which might result from changing from The Chart to the MRB method of water obstraction. It was decided to convert the model to a tidally averaged, daily simulation model, for the purposes of this study.

4.5.2. The Dynamic Simulation Model

4.5.2.1. The initialisation sub-routine of the Thames simulation model is, in effect, the original Thames study state model. This program reads the basic data which defines the Thames system and calculates the steady state solution assuming the system is in equilibrium. This data includes the volume and surface area of the 34 slices, initial conditions or arbitrary estimates of salinity and temperature distributions and parameter values which relate to each slice, such as dispersion coefficients, and reaeration coefficients. Also given are the daily inputs and locations of discharges of fast carbon, slow carbon, fast nitrogen, slow nitrogen, ammonia, nitrate, dissolved oxygen and water flow additions. The most important differences between the models at initialisation are (i) It is assumed that the river flow on the first day of simulation has persisted for sufficient time to achieve dynamic equilibriam. (ii) The dispersion coefficients are assumed to be correlated with river flow and so are calculated accordingly within the sub-routine (iii) The water temperature of each slice is not given explicitly but is computed using empirical coefficients according to the time of year of the first day. The first of these assumptions is not unrealistic if simulations start of January 1st which was our practice. The dispersion coefficients were calculated using the 1977/78 weekly surveys of salinity according to well established techniques (Uncles and Radford 1980) The greater range of river run-offs for 1975/76 would have provided a better calibration data set but as these were not included in the Thames Data Archive they could not be made available in time for this calibration exercise. However, the highest salinities observed during the drought in September 1976, were used to confirm the lowest expected dispersion coefficients. It was assumed that dispersion varied linearly with river run-off and with position along the estuary according

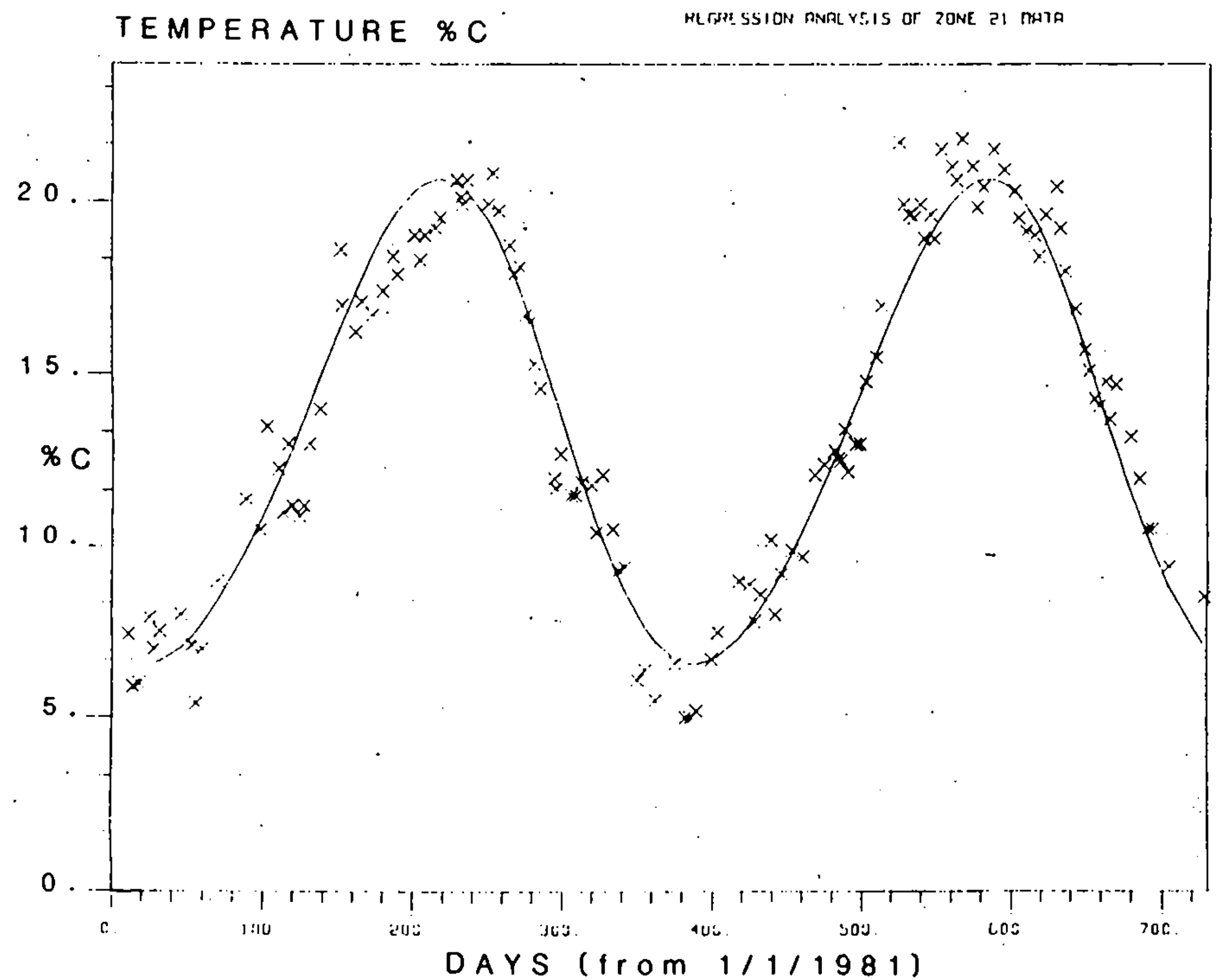


Figure 4.3. Measured water temperature (x) over a two year period (1981/82) and the fitted cosine curve for Zone 21.

to computed empirical relationships.
Equations of the form:-

$$T = P(1) + P(2).\cos\,[\pi\,(t-P(3))\,/180] + P(4).\cos\,[\pi\,(t-P(5))\,/90]$$

were fitted to measured water temperature for each slice where T = water temperature t = time in days from January 1st; P(I) = empirical parameters.

Points from these curves were used to simulate water temperature for the initial conditions and for subsequent simulation. Examples of the raw data (1981/82) and the fitted curves are given in Figure 4.3 for zone 21, i.e. slice 24. When this whole initialisation subroutine is executed it computes the steady state solution for the estuary which applies to the first day of the simulation run.

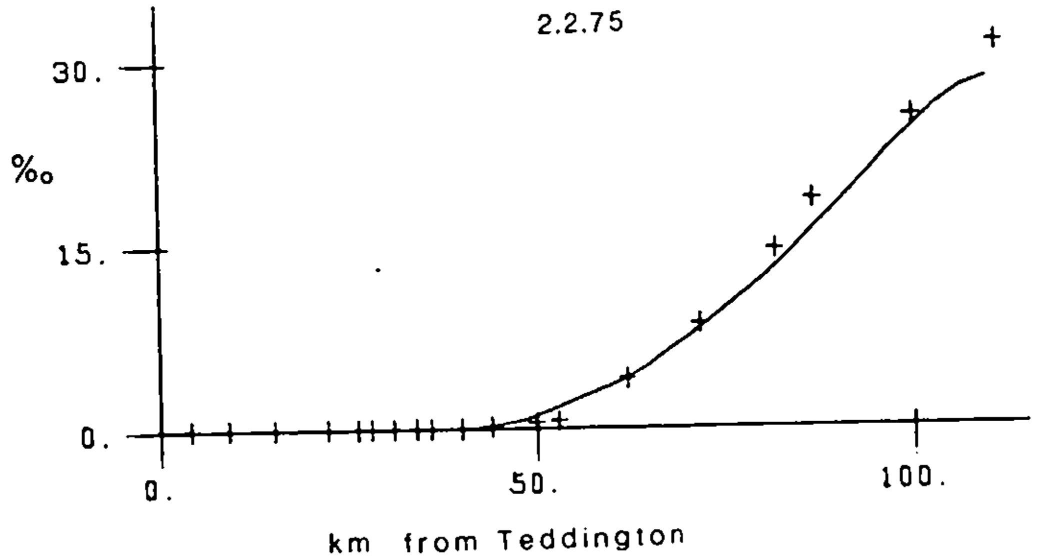
4.5.2.2. The Dynamic Water Quality Model takes the steady state solution for the first day as its initial condition for subsequent simulation. It uses all the process equations in exactly the same manner as the original steady state model but it replaces the interactive equilibriating equations with a finite difference equivilent so progressing the state variables through time in small incremental steps DT until the predetermined finishing time is reached (Radford et al 1981). The model is driven by records of daily river flow data at Teddington taken from historic sequences or alternatively from simulated scenarios of interest to the investigation (generated by the River Thames Water Quality Model, see section 3). It is assumed that water temperature is determanistic and follows the fitted curves given in 4.5.2.1. Polluting loads may be entered as average daily rates relating to quarterly or annual discharge from historic records (or hypothetical). The model generates time series data of the state variables DO, BOD (fast and slow organic carbon), nitrate, ammonia and salinity on a daily basis. Simulated distributions of these variables may be compared with measured values on any day of the year that these were available.

4.5.3. Model Validation

4.5.3.1. The Thames Steady State model has been studied extensively over the 20 years of its existance and its most recent assessment is presented by Barrett et al. (Appendix 4.1.) In this paper the improvement in esturine water quality over the period 1969 - 1975 is clearly demonstrated in the data presented, and duplicated by the predictive model. It is necessary however to demonstrate for this study that the dynamic model would adequately reproduce observations on a day to day basis. Having calibrated the model using mostly 1977/78 data it seemed useful to validate it against 1975/76 data, a period of extremely high and low river flows. One would not always expect such a close degree of agreement between daily distributions as one would for quarterly data because of odd day to day events not programmed into the model which are averaged out in period data. Such phenomina would include the effect of spring/neap cycles on the accuracy of the tidal correction, flash floods, and anomolous polluting loads. However the validation run of the simulation model for salinity produced sufficient agreement on a large enough proportion of the sample days to convince one of it's validity. Appendix 4.2. gives plots of the salinity distributions measured on 90 separate occasions in 1975/76 together with simulations for the same days. The measure of agreement is in general exceptionally close, especially for the periods of low flow when management strategy would have its greatest effect, but the model also faithfully reproduces the whole range of conditions as is demonstrated in Figure 4.4. Here the highest and lowest salinity distributions for the period 1975/76 are given and it can be seen that the simulations fit both extremes very closely. Again the closeness of the fit is best judged in relation to the possible range of solutions as demonstrated in Figure 4.5. In this figure each daily simulation for salinity has been plotted for the 365 days of 1977 giving an envelope for the predicted salinities for that year.

4.5.3.2. The most important variable from the point of view of maintaining a viable ecosystem is the concentration of oxygen in the water column. Results generated by the Water Quality Simulation Model for 90 sampling occasions in 1975/76 are given in Appendix 4.3. The model successfully generates the full range of oxygen distributions from winter and summer although, as must be expected odd events prevent a good validation on some occasions. However the very different oxygen sag curves for 2/3/75, 27/7/75, 12/11/75, 25/1/76 and 12/9/76 are all reproduced with some realism. The most consistant divergencies between observed and simulated distributions occur in the summers when phytoplankton blooms greatly increase the oxygen saturation (e.g. 3/8/75, 10/8/75, 28/6/76, 18/7/76 and 29/8/76) during the daytime. It is quite probable that this oxygen is rapidly lost from the water column by dispersion to the atmosphere and by night time respiration of the phytoplankton. The simulation therefore tends to give a slightly pessimistic prediction of the oxygen status of the water column. In the drought of 1976 it was necessary to back pump over Teddington Weir between 6-12 millions of gallons per day to maintain an adequate flow through the locks. This policy had a serious effect on water quality in the reach of the Thames above London Bridge. The model was not programmed

SALINITY ‰



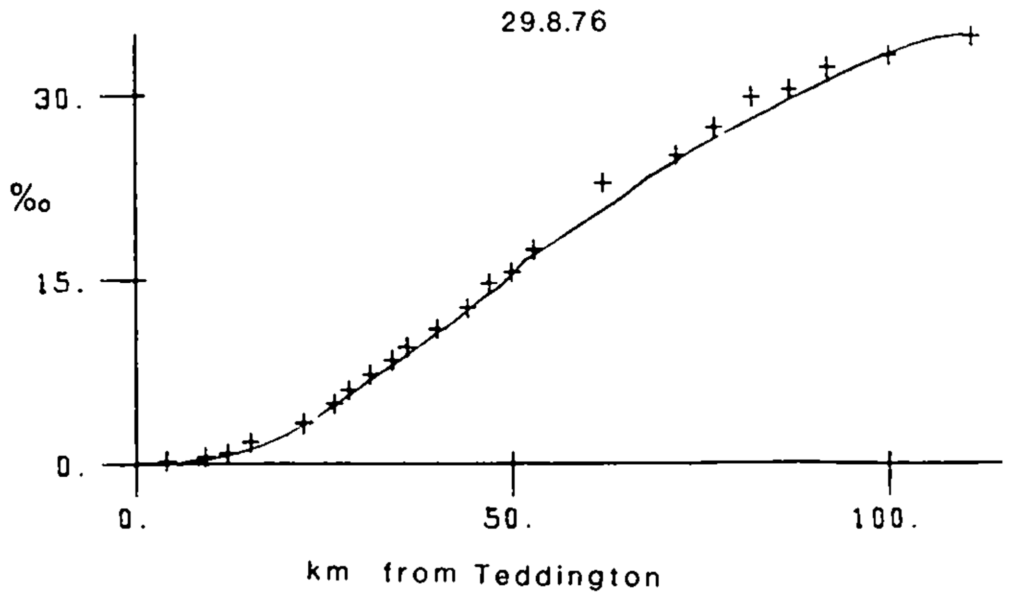


Figure 4.4. Observed (+) vs. Simulated (continuous lines) 'mid-tide' axial salinity distributions (‰) from Teddington (0 km) to Shoebury Ness (111 km)

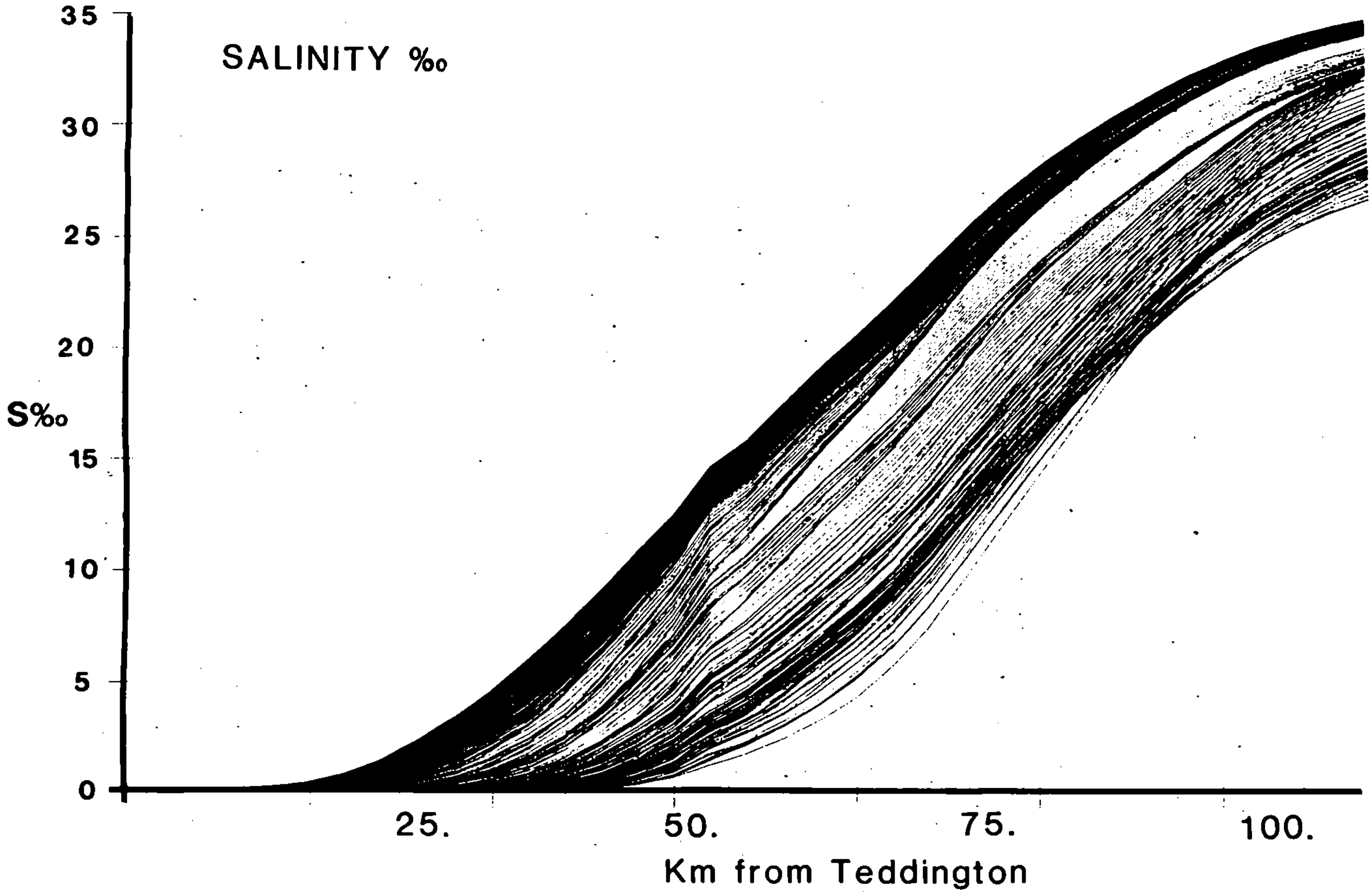


Figure 4.5 Simulated daily distributions of salinity for the 365 days of 1975 .

to simulate this management strategy and so it tends to over-estimate the oxygen concentrations for the period from 12/9/76 till the end of the year. As a result of the experience gained by this strategy it has been decided by Thames Water that back-pumping is not a useful technique and would not be built into any future plan of the estuarine management.

4.5.4. Selection of test simulation runs

Having validated the model it was necessary to select a series of river flow sequences which could be used as the basis for a critical evaluation of the effect of alternative management strategies on the system. The period of the 1975/76 drought was an obvious candidate since it provided the most severe low flow conditions in recent history and its impact is still fresh in the minds of most people, in a general sense. A slightly more severe drought was recorded in 1944/45; severe in the sense that the period of low flows extended over eighteen months rather than the 12 months duration of the 1975/76 event. It was decided to select, also, a non-drought sequence where a six month drought looked as if it might develope into a full eighteen month drought but in the event there was sufficient rainfall in the two winter periods to alleviate the effect of two consecutive dry summers. Finally the non-drought sequence 1980/81 was chosen as a fairly average period of river run-off. The 1975/76 sequence was run for the relavent historic flows and discharge loadings in the validation simulation. All four sequences were then executed for current abstractions and current demands and in all four cases equivilent simulations were performed for the Chart and the MRB management strategies. The whole exercise was repeated for a possible future demand.

In considering what future demands to assume for the purposes of this study, Thames Water have estimated that if no additional resources such as new reservoirs were introduced, the reliability of the water supply to London would have reached an intolerably low level by the year 2006; further resources would therefore certainly be introduced by that date. The forecast demand for 2006 was thus chosen for assessment (without additional reservoirs), as this represents the maximum possible demand on the river at low flow and therefore the "worst case" situation. The Thames Water demand forecast for the London area for 2006 is equivalent to 117% of the "current" (1984) demand.

4.6 EFFECTS OF ABSTRACTION POLICY ON WATER QUALITY

4.6.1. Introduction

The model has been run for a number of classical drought sequences, and non drought sequences, in each case for the period from January 1st of the first year, to December 7th of the second year. The mean axial distribution for the 706 days has been computed, together with a measure of the seasonal variability of the data. The measure of variance chosen is the standard deviation, expressed as a percentage of the mean value; this is known as the "Coefficient of Variance" (C of V). For each classical drought sequence the results have been presented in the following ways:-

a) Graphical data giving the mean and coefficient of variance (C of V) for each of the five state variables, salinity (S‰), Ammonia (NH_3) BOD nitrate (NO_3) and DO%. In each case a direct comparison is made between "The Chart" vs "MRB" policies.

b) Times series data giving daily values of salinity and oxygen for a number of specific locations, giving direct comparisons of "The Chart" vs "MRB" policies.

The figures (Figure 4.6. - Figure 4.36) all appear at the end of this section (Section 4.6.) in which they are referenced.

4.6.2. Historic Demands (1975/76) vs
Current Demands (1984)

It was considered to be valuable to undertstand by how much the estuarine water quality has changed since 1975/76 due to the increased abstraction demands on the system, the improved standard of effluent treatment and any small change in the design of The Chart up until 1984.

4.6.2.1. Classical Drought Sequence 1975/76
Historic vs. Current Management

The axial distributions of the five variables are given in Figure 4.6. where they may be compared with the equivalent distributions for the same drought sequence but with 1984 demands, polluting loads, and the current version of "The Chart". This comparison provides a yardstick against which the effects of proposed changes may be measured. The most notable features of this figure are:-

i) The mean salinity distribution has moved slightly up-estuary and its variance has decreased a little. This effect is probably due to a combination of a slightly increased water demand coupled with greater control of variations in discharge from sewage works.

ii) Ammonia levels have increased but up-estuary of London Bridge the variance has decreased considerably. It is on record that ammonia levels at Teddington in 1975/76 were very variable due to lack of adequate control at some sewage works.

iii) Total BOD has increased up-estuary of London Bridge but decreased down-estuary. In general the variance has not changed by a significant amount for most of the estuary.

iv) Nitrate levels have slightly increased since 1975/76 but in general they are less variable under current conditions above London Bridge.

v) The large mid-estuary dissolved oxygen sag that was observed in 1975/76 has been considerably reduced. This is due to the increased levels of treatment provided by recently introduced sewage works. The variance of the oxygen concentrations have also been considerably reduced.

Figures 4.7., 4.8.,4.9. and 4.10 give time series plots of daily salinity and oxygen concentrations for four specific locations in the estuary (Syon Park, London Bridge, Barking Creek and Hole Haven Creek respectively). These locations have been chosen as easily identifiable places where the largest changes have occurred between 1975/76 and the present.

i) At Syon Park the salinity would only be changed by a very small amount under present conditions and the water would in general be very slightly less salty. The averaged oxygen concentrations would not be significantly changed but values would be marginally less variable.

ii) At London Bridge salinity would be slightly greater at the start, but not significantly different at the height of the drought. The oxygen concentrations would be less variable and would show no tendency to sag whereas historically levels fell to about 50% of saturation in 1975.

iii) At Barking Creek salinity would be very slightly increased but the effect of new treatment works is to substantially reduce the observed oxygen sag which had reached as low as 10 parts per thousand in 1975/76.

iv) At Hole Haven Creek there would be no significant difference in the salinity regime. The oxygen levels would be about 10% higher under the present water treatment regime.

4.6.3. Current Demands (1984)

For the following comparisons of management control (The Chart v MRB) the model was run using the 1984 demands and polluting loads in each case.

4.6.3.1. Classical Drought Sequence - 1975/76
The Chart vs. MRB (1984)

The effect of an "MRB" policy on the mean axial distributions of the five variables is very small and only detectable for the tidal reach up-estuary of London Bridge. (Figure 4.11.) This is also true of the seasonal variability of the concentrations (C of V).

The time series data (Figures 4.12., 4.13. and 4.14) demonstrate the minute effect of the "MRB" vs. the current "Chart" system of management. At Teddington and Syon Park salinities are not increased by more than 0.4 parts per thousand and on occasions the MRB policy reduces salinity by as much as 1.5 parts per thousand. At London Bridge there is no consistent effect of the "MRB" policy but the day by day variation is predicted to be a little smaller under this regime. Oxygen levels are not significantly affected by the proposed change in management system.

4.6.3.2. Classical Drought Sequence - 1944/45 -
The Chart vs. MRB (1984)

In the 1944/45 classical drought sequences no important difference has been observed between the results of the current "Chart" system and the proposed "MRB" system of management (Figure 4.15) In general the system responds in a manner analogous to that demonstrated for the 1975/76 drought sequence. The slight increase in ammonia, BOD and nitrate and the resultant small decrease in dissolved oxygen concentrations are due to the small increase in water abstracted early in the year which contributes to the greater resource reliability of the MRB policy.

The daily time series results for Teddington (Figure 4.16), Syon Park (Figure 4.17) and London Bridge (Figure 4.18) do, in general show how that in each case the salinity increases slightly earlier in the year under the MRB policy which avoids the formation of extreme peaks that characterise the current policy. On only one occassion does the proposed policy result in a higher peak salinity and that is at London Bridge in 1945 and the difference is less than one part per thousand. The oxygen concentrations are very similar under the two different management rules, the greatest difference occuring at those locations where the oxygen concentrations are highest (Teddington and Syon Park). At London Bridge oxygen concentrations would be identical.

4.6.3.3. Non-Drought Sequence 1952/53 The Chart vs. MRB (1984)

The average concentrations of salinity, NH BOD, and NO_3 are all much lower in non drought years (1952/53) (Figure 4.19) compared to a classical drought sequence (1944/45) (Figure 4.15) For example the peak nitrate concentration is reduced from 17 to 12 ppm, BOD from 10 to 7ppm, ammonia from 0.7 to 0.6ppm and saline intrusion is displaced about 5 km seaward. It is under these conditions that the MRB policy allows for the abstration of more water so reducing the moderate flows over Teddington Weir. The relative effect of this policy on average distributions is very similar to its effect in the drought year 1944/45. There would be a slight increase in ammonia, BOD, and nitrate and an associated small drop in DO upstream of London Bridge but absolute levels would not approach those experienced under drought conditions i.e. the effect of the different policies is small compared with the effects caused by natural climate conditions.

Under either regime the water above Richmond half-tide lock rarely becomes saline and there is no measurable effect on the oxygen regime. For this reason no time-series have been presented for the stretch of estuary immediately below Teddington. At Syon Park (Figure 4.20) the MRB policy causes salinities greater than the chart system would allow but the maximum values of 1% are still much less than equivalent drought values (3%) under either system.

The same is true at London Bridge (Figure 4.21) where the equivalent salinity figures are 10% for non-drought years compared with 15% for drought years. Also the relative effect is only an increase of a maximum of 5% for MRB compared with the current chart system. The oxygen concentrations are very similar under the two different regimes, the greatest differences occuring at Syon Park (Figure 4.20) where the oxygen concentrations are highest, but the oxygen depletion is never any worse than currently experienced during drought conditions. At London Bridge the predicted oxygen concentrations are identical.

4.6.3.4. Non-Drought Sequence 1980/81 - The Chart vs. MRB (1984)

The average axial concentrations of all five variables are identical under the two strategies (Figure 4.22) The very slight increases in the coefficient of variation (C of V) for salinity and nitrate cannot be considered as being at all significant. The absolute levels of the variables are very low, having been diluted by the large river run off. The limit of saline intrusion has retreated some 15km seaward compared with its position during drought years (1944/45) (Figure 4.15)

As there is no evidence in the daily time series results of significant differences between the two management strategies at Teddington or Syon Park these figures have not been included. At London Bridge (Figure 4.23) the policies produce almost identical results and the absolute level of salinity peaks at 5% compared with 15% under either policy in a drought year.

4.6.4. Future Demands (2006)

It is estimated that 117% of the 1984 water demand will be required by the year 2006. A complete set of simulations has therefore been performed to compare The Chart vs. the MRB methods of management for that demand. Implicit in these simulations are the following assumptions:-

i) Under drought conditions the effluent from the rivers Mole and Hogsmill will be larger in 2006 than at present, and since some of this water will have been imported from the Southern Water Authority Area this will effectively cause a net increase in the Thames flow over Teddington Weir.

ii) All of the major lateral inflows from sewage works down estuary of Teddington will be increased by 17%.

iii) All of the polluting loads of ammonia, nitrate, and biochemical oxygen demand will be increased by 17% (i.e. the effluent quality would remain the same). These assumptions give the "worst possible" scenario.

4.6.4.1. Classical Drought Sequence 1975/76 - The Chart vs. MRB (2006)

The combined effect of the 17% increase in demand and the concomitant increase in sewage discharges on the average axial salinity distribution is small (Figure 4.11 vs. Figure 4.24) As might have been anticipated the peak mean levels of ammonia, nitrate, and BOD have all been increased by about 15-20% and there is an associated small depression in the oxygen levels by 10% This change is small compared to the large increase in oxygen concentration achieved between 1975 and 1984 by improved effluent quality discharged from major sewerage works (Figure 4.6.)

A comparison of the axial concentrations for the chart vs. the MRB (Figure 4.24) system of management indicates that the two regimes produce very similar results on average. The MRB gives slightly higher concentrations of ammonia, nitrate, and BOD but the effect is very small indeed. The variances (C of V) of all variables follow almost identical patterns although they are smaller for the MRB policy.

The time series results at Teddington (Figure 4.25) show a slight reduction in salinity under the MRB policy relative to the present chart system but in both cases peak salinities are reduced compared with those predicted under current (1984) demands; oxygen results are not significantly different. The same comments are broadly true at Syon Park (Figure 4.26) and London Bridge (Figure 4.27) also.

4.6.4.2. Classical Drought Sequence 1944/45
The Chart vs. MRB (2006)

All the comments made regarding the 1975/76 drought sequence apply equally well to the 1944/45 data (Figures 4.11 vs. Figure 4.28) Absolute concentrations of salinity, ammonia, nitrate and BOD and higher than 1975/76 because the earlier drought was more severe than the later one.

The time series plots (Figures 4.29., 4.30. and 4.31) show no significant differences between salinity under the two regimes except for a general tendency for the MRB policy to produce higher salinities earlier in the year but lower salinities later in the year. Oxygen concentrations are not significantly different under the two regimes

4.6.4.3. Non Drought Sequence 1952/53
The Chart vs. MRB (2006)

In general (Figure 4.32 vs Figure 4.28) the mean distributions of salinity ammonia, nitrate and BOD and lower than in drought years but the increased demand and effluent return has resulted in slightly higher levels than those predicted for the 1984 demands (Figure 4.32 vs. Figure 4.12) The MRB policy adds to these increased concentrations but the effect is small and limited to the reach above London Bridge.

A comparison of the two regimes under the 2006 (Figure 4.33, 4.34) yields similar conclusions to those deduced for the 1984 demands (Figures 4.20 4.21.) except that the absolute values of salinity are lower under the higher demands in spite of increased abstraction. The oxygen time series are almost identical, indicating no serious deteriation due to the combined effects of increased demands and the MRB regime.

4.6.4.3. Non Drought Sequence 1980/81 -
The Chart vs. MRB (2006)

The relatively high flows experienced in 1980/81 result in generally reduced level of all pollutants in the estuary and an enhanced oxygen regime (Figures 4.35. 4.36) The increased demands and polluting loads for 2006 do result in higher concentrations in the estuary but the effect is small. There are no significant differences between the MRB and the traditional chart schemes i water abstraction and the absolute value of salinity at Syon Park is zero.

4.6.5. General effect on sediment deposition and suspension

There has been some concern expressed that changes in the residual flow over Teddington Weir might affect the location of areas of sediment deposition. In particular it has been suggested that the induced change in current velocities might cause a pronounced increase in the height of mud banks at various places above London Bridge. This question could not be answered by the use of the Thames Water Quality so a small statistical study was made to give some indication of the likely effect.

4.6.5.1. The results of a multiple regression analysis which relates suspended partulates to tidal range and daily mean river discharge is given in Table 4.2.

TABLE 4.2.

The coefficients in this table refer to an equation of the form $\log_e S = A + B \log_e(R) + C \log_e(Q)$ where S = suspended solids, R = tidal range and Q = river flow.

Sampling Zone No.	Coefficients A	B	C	Correlation squared
2	-0.32^{ns}	1.06**	0.34***	.17
3	2.49**	0.14^{ns}	0.09^{ns}	.01
4	2.98***	0.68	-0.24**	.08
5	1.95*	0.13***	-0.40***	.28
6	2.43^{ns}	2.06***	-0.64***	.39
7	3.39^{ns}	1.42***	-0.51***	.33
8	2.49***	2.13***	-0.53***	.60
9	1.54**	2.30***	-0.33***	.41
10	1.82***	2.13***	-0.28***	.40

The tidal range would remain the same as at present under the MRB policy but river discharge would be reduced for moderate flows and increased for low flow conditions. These flow changes are very small in comparison to the total annual variations in flow upon which the correlations are based so, taking the relationships on face value they would have little impact on total suspended particulates in the water column.

4.6.5.2. The negative correlation between river discharge (Q) and suspended particulates (S) requires some explanation because, in general one would expect higher flows to induce higher suspended solids. Close inspection of the data suggests a possible explanation . There is a general tendency for suspended particulates to increase from Teddington Weir to London Bridge. As flows decrease the zone of maximum turbidity tends to migrate up the estuary towards the weir (Bale et al In press) For an observer at one location this will give the impression of turbidity increasing even if at the same time there is a small reduction in suspended partulates in the estuary as a whole.

4.6.5.3. Areas of net deposition or resuspension will be associated with the seasonal drift of the zone of the turbidity maximum. In view of the very small flow changes associated with two policies it is very unlikely that the areas of deposition would be moved by any significant distance up or down the estuary under the MRB policy.

4.6.5.3. During low run off conditions the tidaly induced currents are much larger than those induced by residual river flow. Resuspension of sediment is therefore largely influenced by the tidal range during droughts. The small changes in river flow induced by water abstraction would therefore not significantly affect the sediment budget for the estuary.

4.7 CONCLUSIONS

4.7.1. The simulation model of the Thames Estuary developed for this study adequately represents the observed water quality of the estuary for the purpose of comparing alternative strategies of water abstraction.

4.7.2. The effects of improved water treatment since 1975/76 are clearly demonstrated by the simulated reduction in the average oxygen sag which is most pronounced in the Beckton/Crossness reach of the estuary. Commensurate decreases in biochemical oxygen demand and ammonia are observable, although nitrate levels continue to rise by about 1% per annum up-estuary of London Bridge.

CLASSICAL DROUGHT SEQUENCE 1975/76

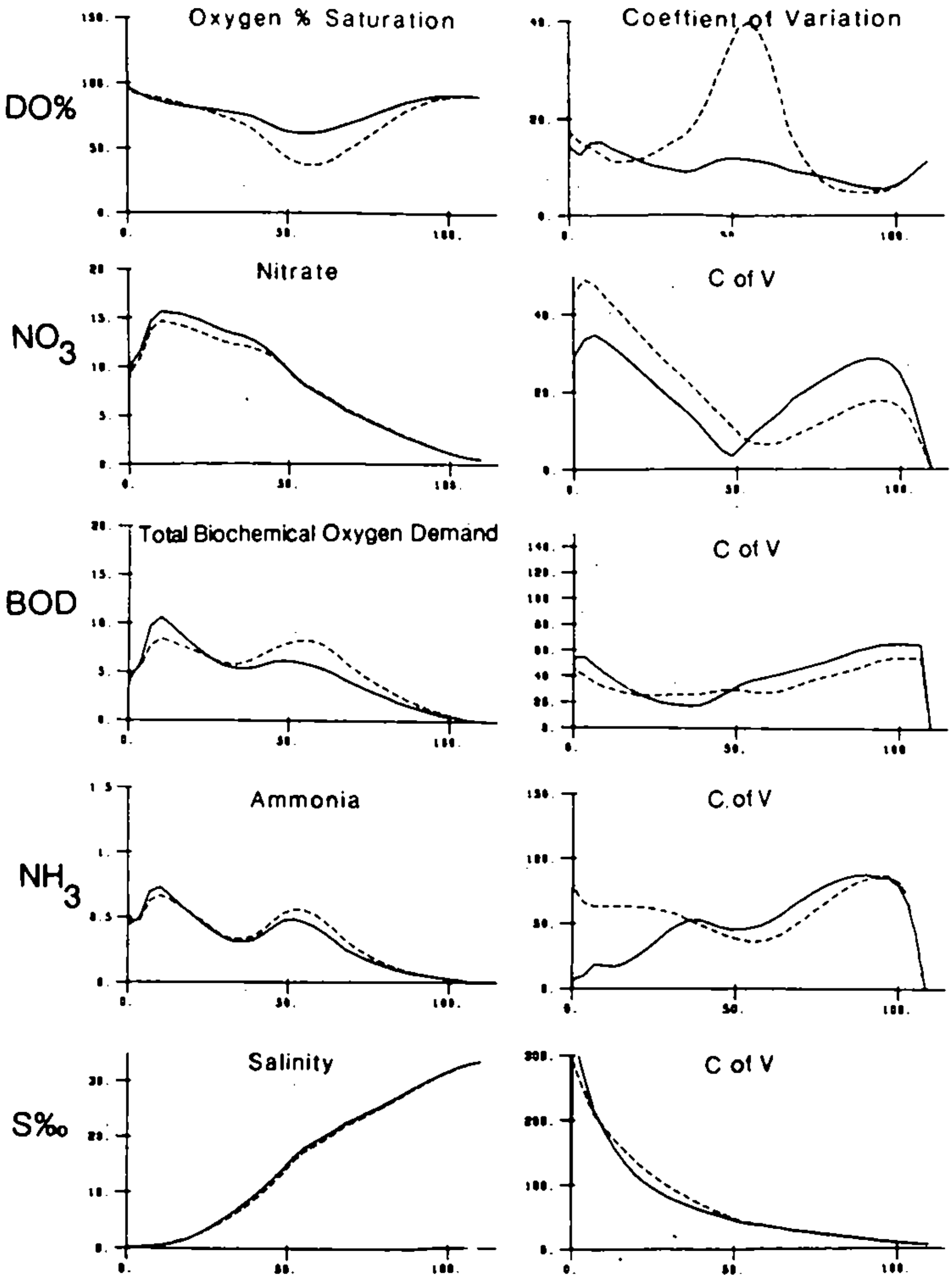


Figure 4.6

Comparison of predicted average axial concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rules according to current chart vs.1975/76 demands, loads and chart.

Current chart ———— 1975/76 - - - - - -

CLASSICAL DROUGHT SEQUENCE 1975/76

SLICE 4 SYON PARK

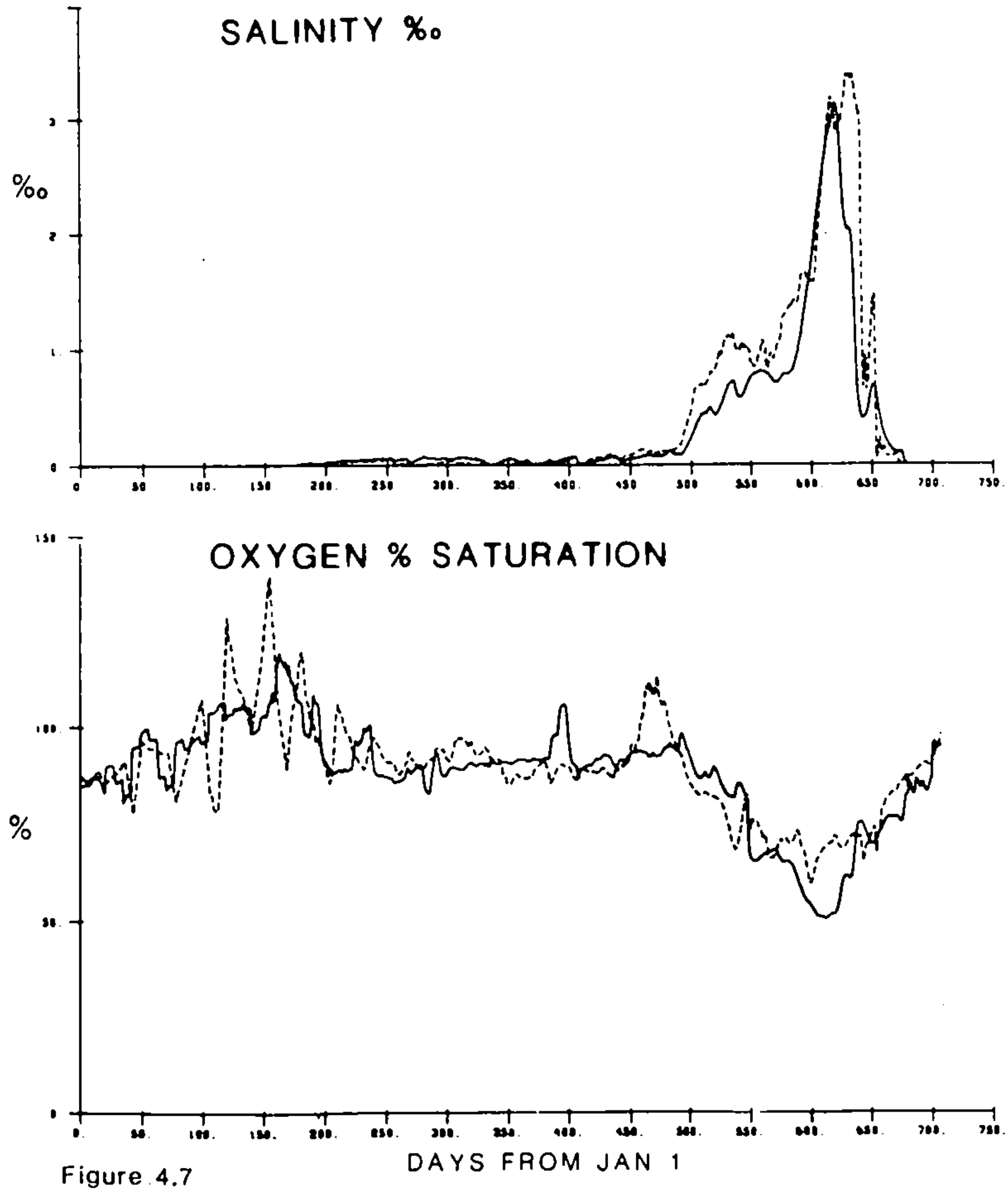


Figure 4.7

Comparison of predicted concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rates according to the chart vs. 1975/76 demands, loads and chart

Current Chart ——————— 1975/76 — — — — — —

CLASSICAL DROUGHT SEQUENCE 1975/76

SLICE 11 LONDON BRIDGE

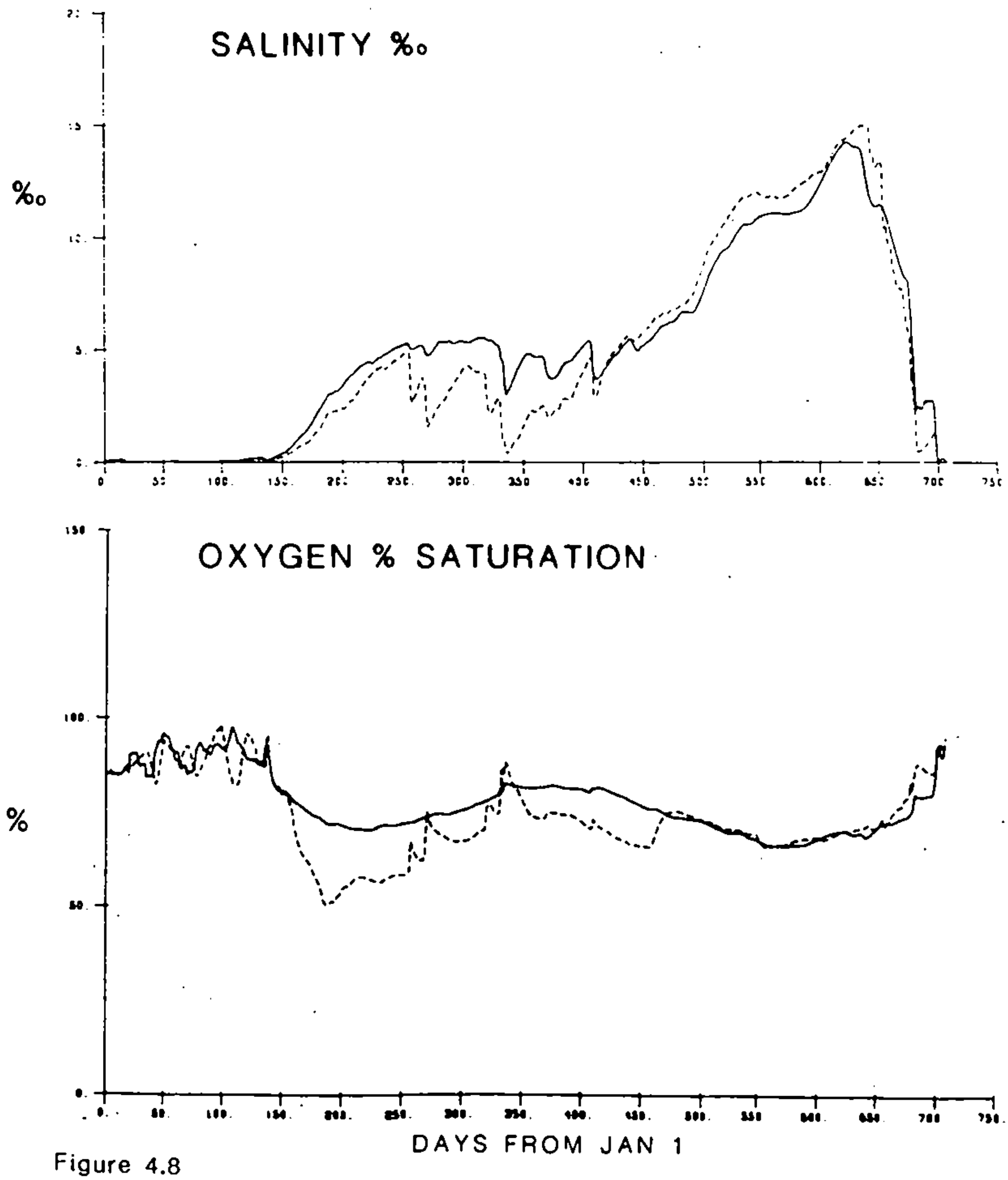


Figure 4.8

Comparison of predicted concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rates according to the chart vs. 1975/76 demands, loads and chart

Current Chart ———— 1975/76 - - - - - -

CLASSICAL DROUGHT SEQUENCE 1975/76

SLICE 18 BARKING CREEK

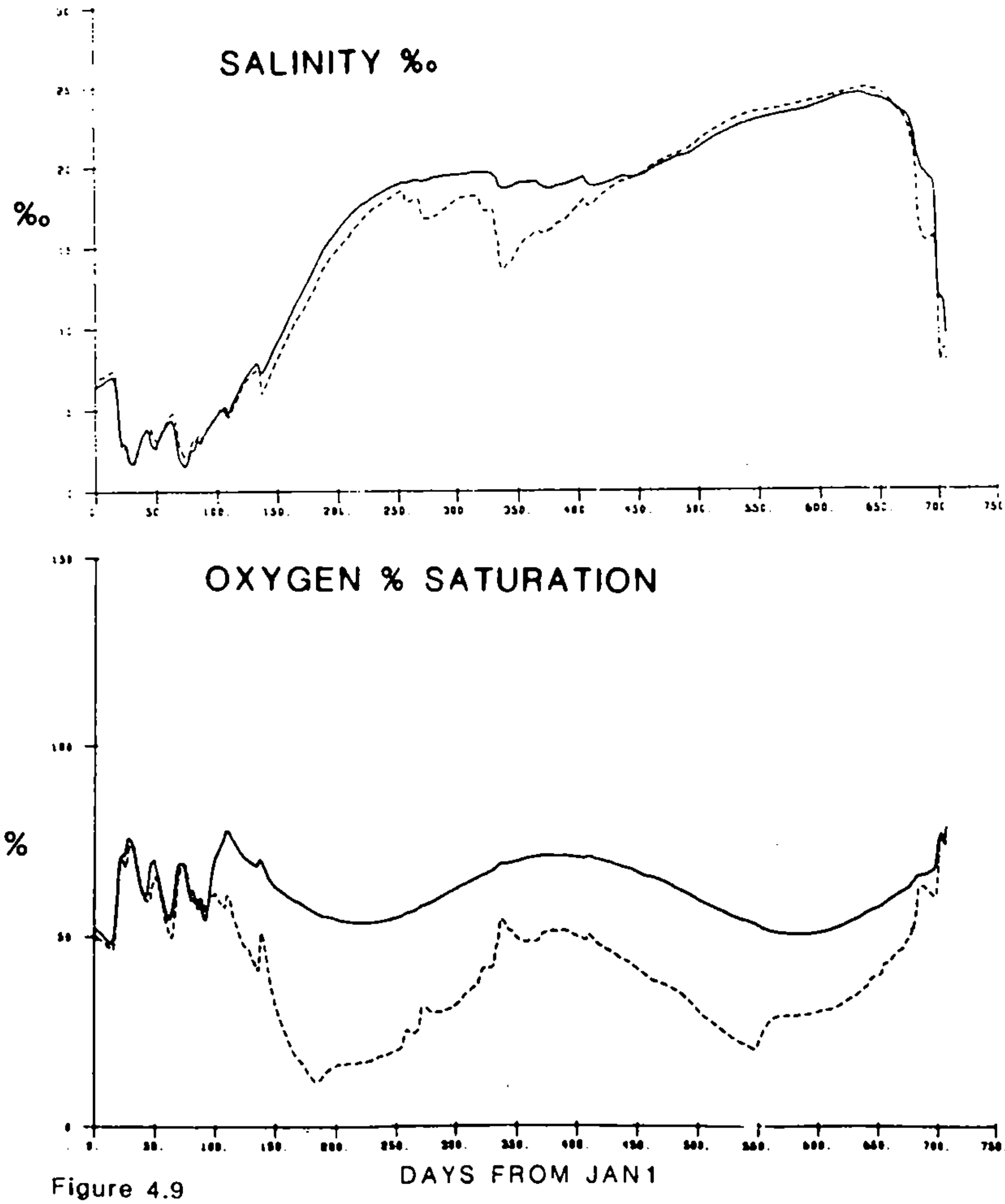


Figure 4.9

Comparison of predicted concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rates according to the chart vs. 1975/76 demands, loads and chart

Current Chart ———————   1975/76 – – – – – – –

CLASSICAL DROUGHT SEQUENCE 1975/76

SLICE 28 HOLEHAVEN CREEK

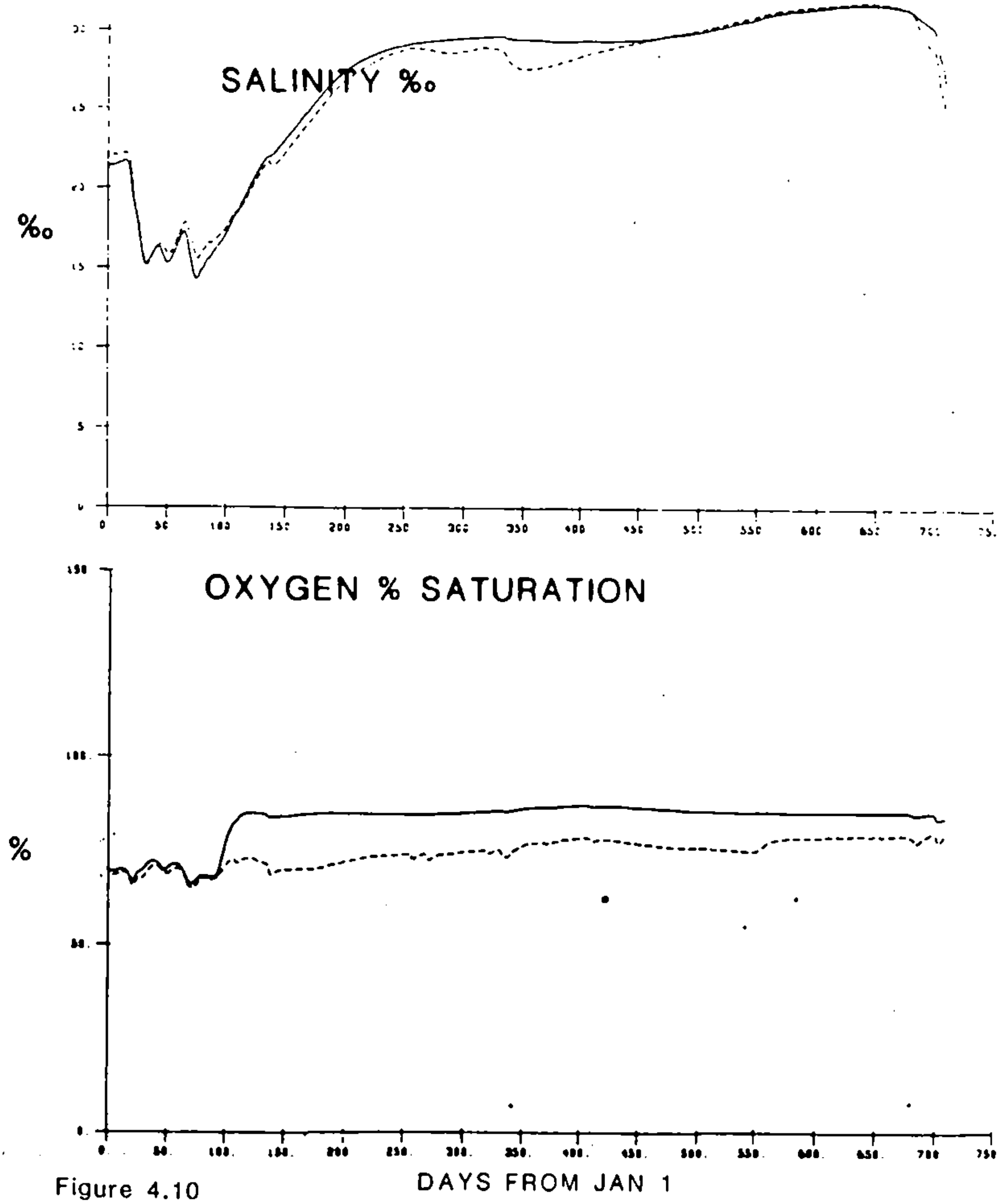


Figure 4.10

Comparison of predicted concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rates according to the chart vs 1975/76 demands, loads and chart

Current Chart ———————   1975/76 — — — — — —

CLASSICAL DROUGHT SEQUENCE 1975/76

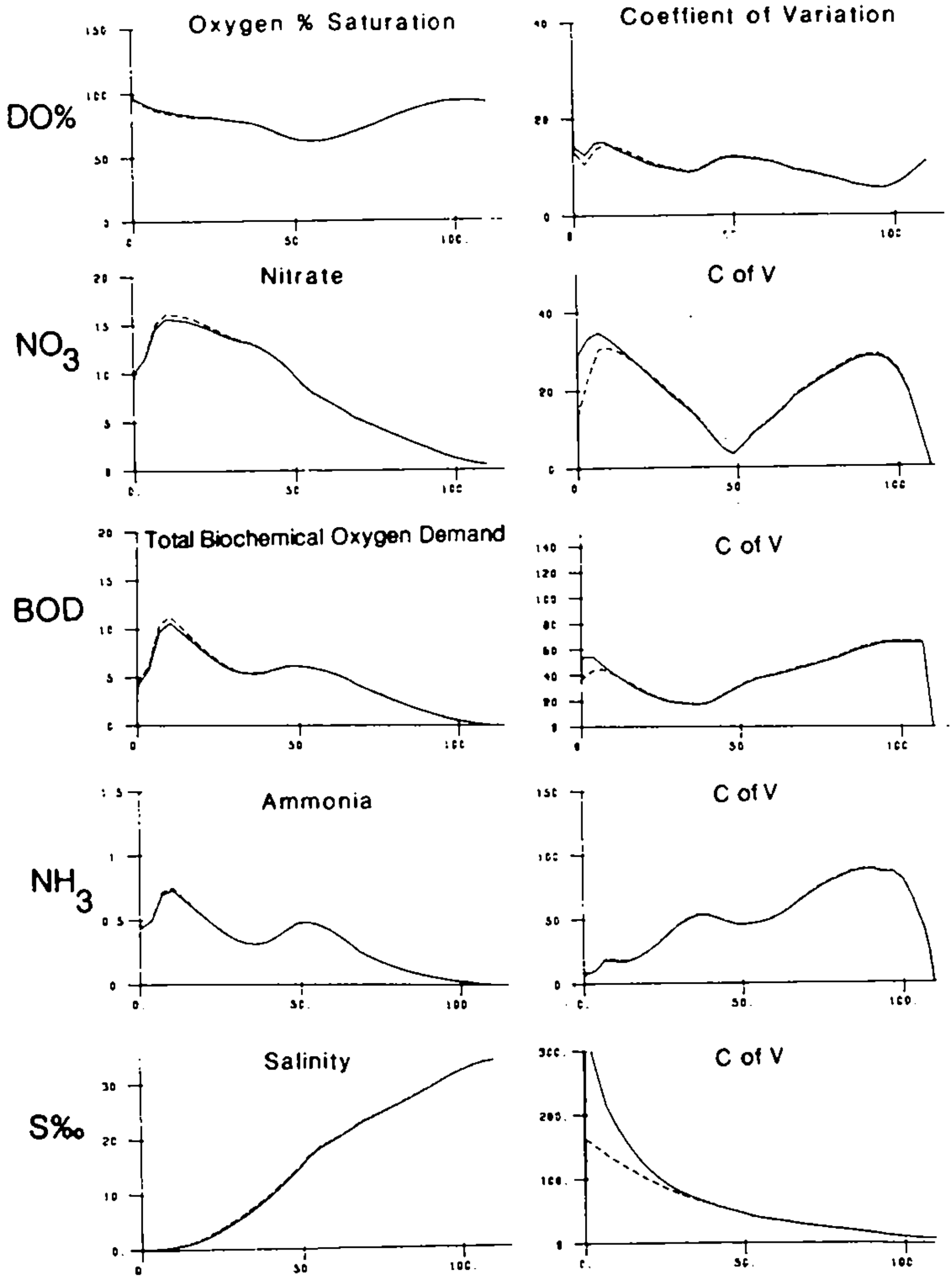


Figure 4.11

Comparison of predicted average axial concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rules according to current chart vs. the Maximum Resource Benefit Policy (MRB)

Current chart ________ MRB Policy --------

CLASSICAL DROUGHT SEQUENCE 1975/76

SLICE 2 TEDDINGTON

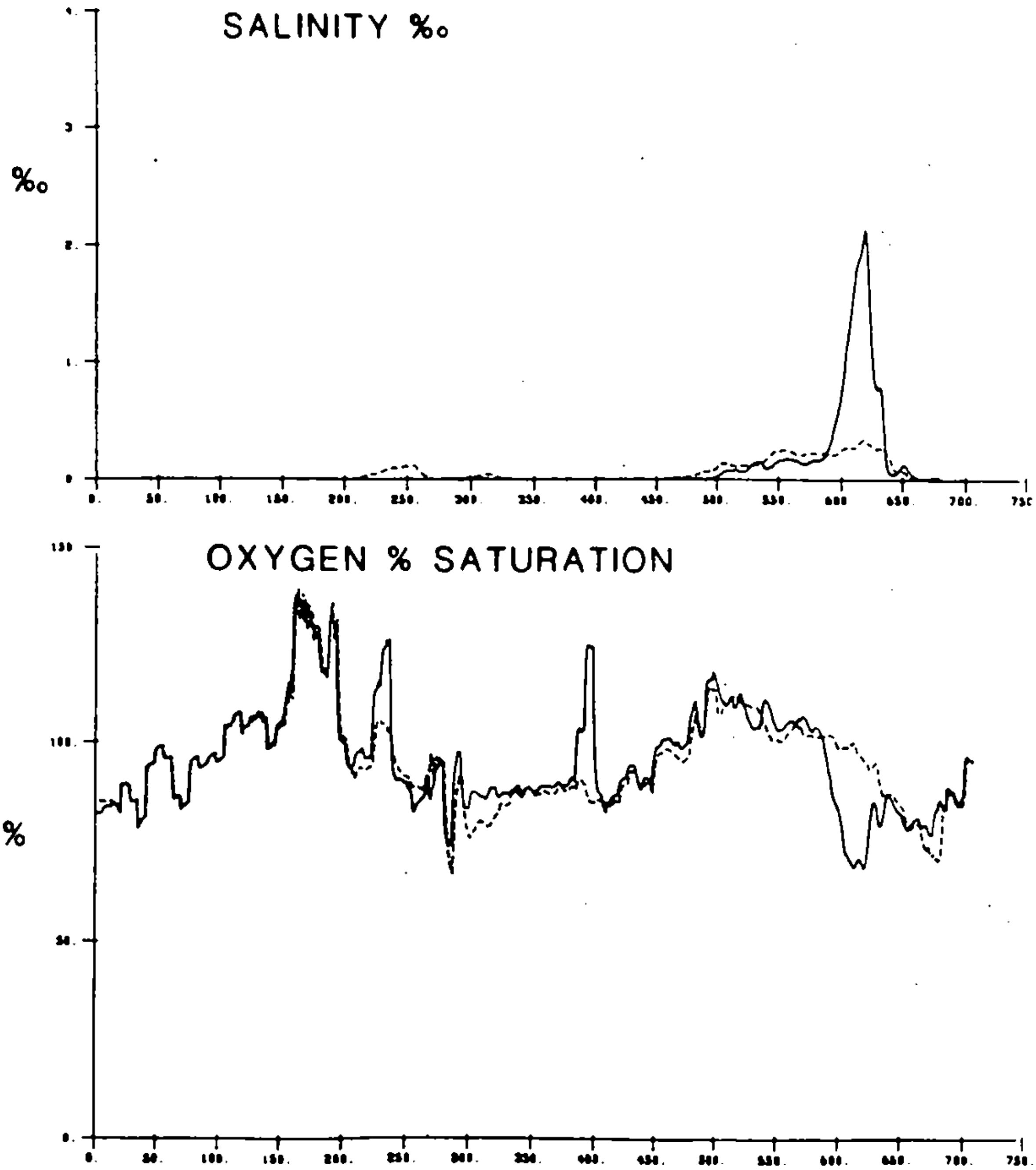


Figure 4.12 DAYS FROM JAN 1

Comparison of predicted concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rates according to the chart vs MRB Policy

Current Chart ——— MRB Policy - - - - - - -

CLASSICAL DROUGHT SEQUENCE 1975/76

SLICE 4 SYON PARK

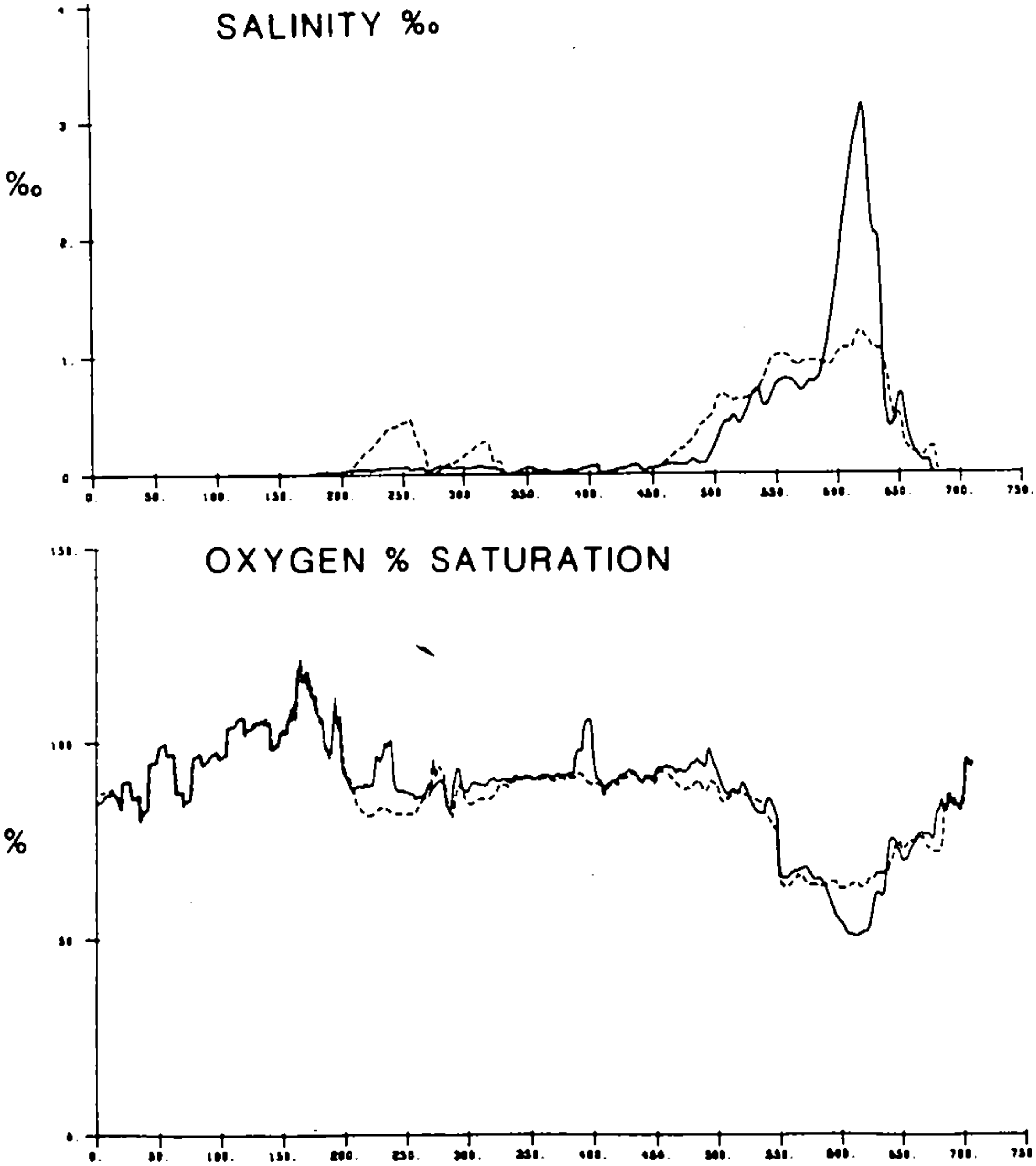


Figure 4.13

Comparison of predicted concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rates according to the chart vs MRB Policy

Current Chart ——— MRB Policy - - - - - - -

CLASSICAL DROUGHT SEQUENCE 1975/76

SLICE 11 LONDON BRIDGE

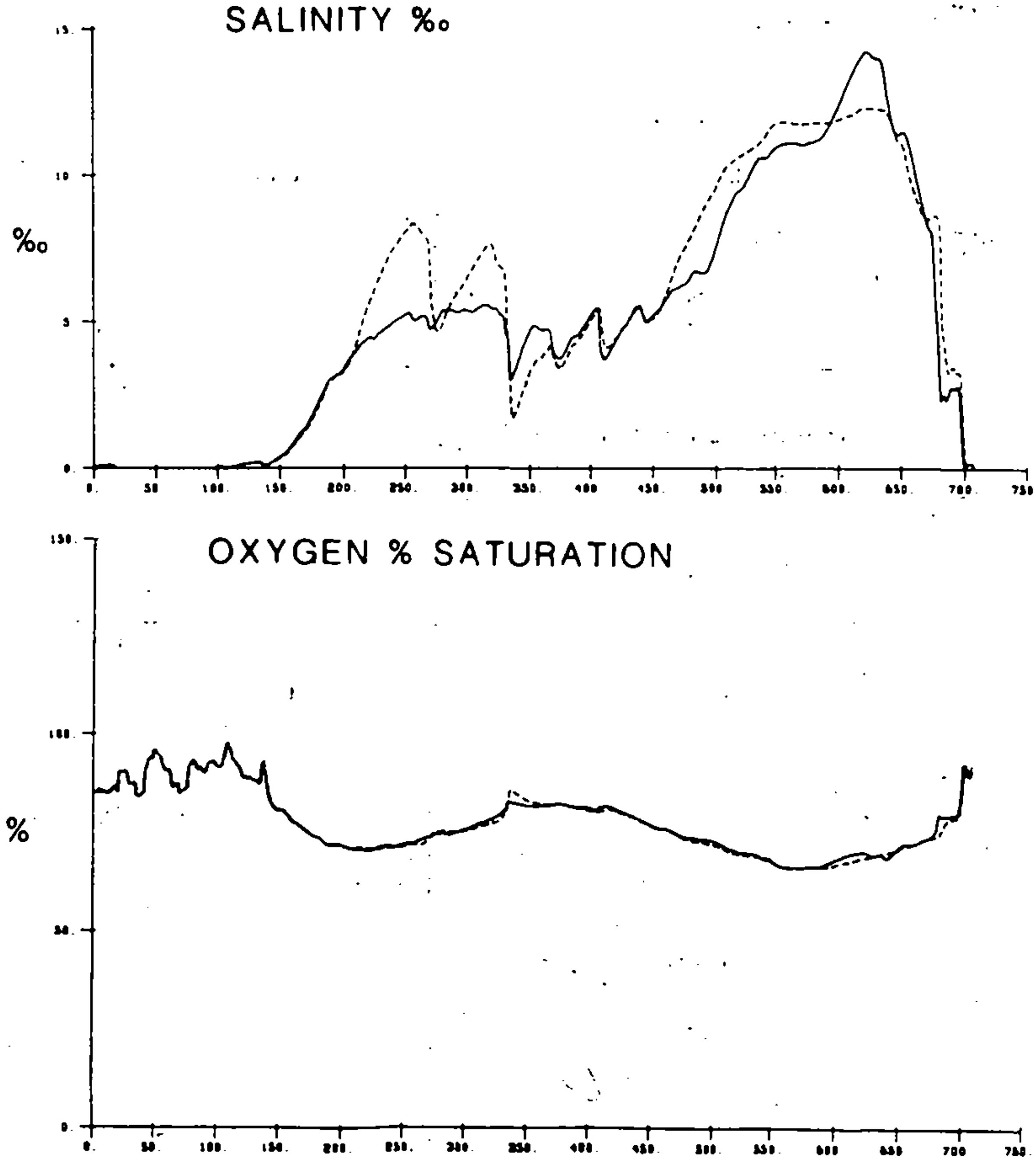


Figure 4.14 DAYS FROM JAN 1

Comparison of predicted concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rates according to the chart vs MRB Policy

Current Chart ———— MRB Policy - - - - - -

CLASSICAL DROUGHT SEQUENCE 1944/45

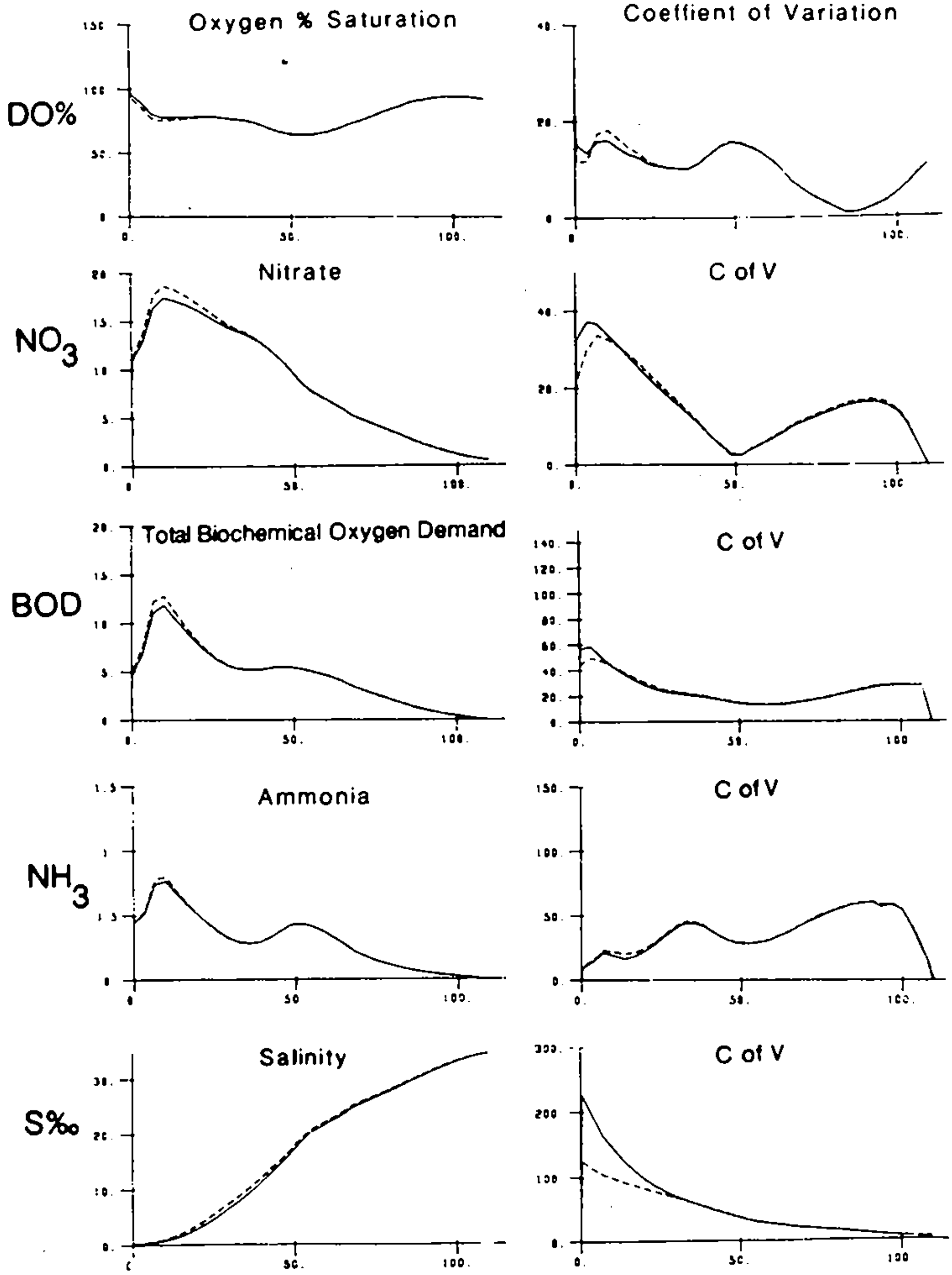


Figure 4.15

Comparison of predicted average axial concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rules according to current chart vs. the Maximum Resource Benefit Policy (MRB)

Current chart ________ MRB Policy — — — — —

CLASSICAL DROUGHT SEQUENCE 1944/45

SLICE 2 TEDDINGTON

SALINITY ‰

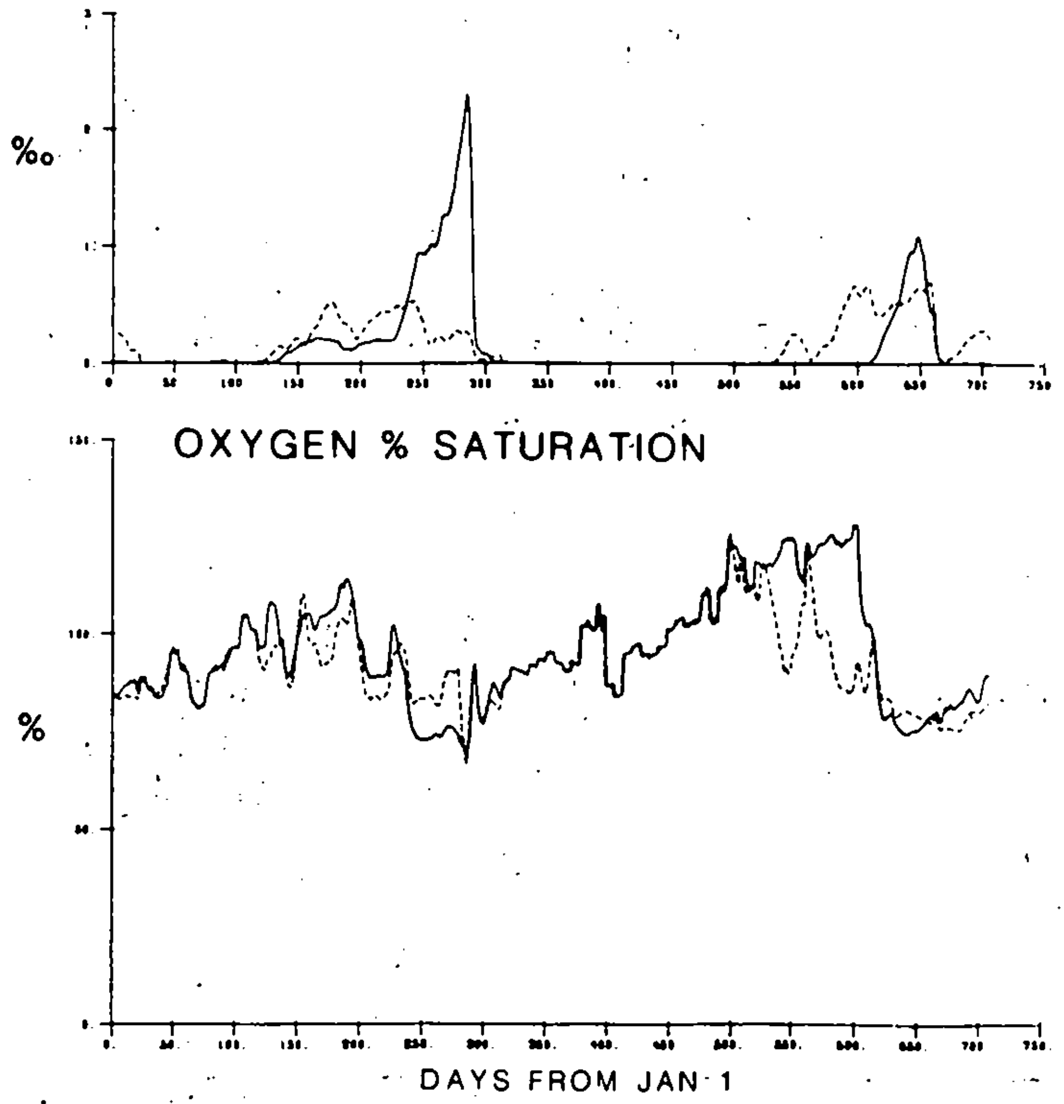


Figure 4.16

Comparison of predicted concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rates according to the chart vs. MRB Policy

Current Chart ———— MRB Policy - - - - - -

CLASSICAL DROUGHT SEQUENCE 1944/45

SLICE 4 SYON PARK

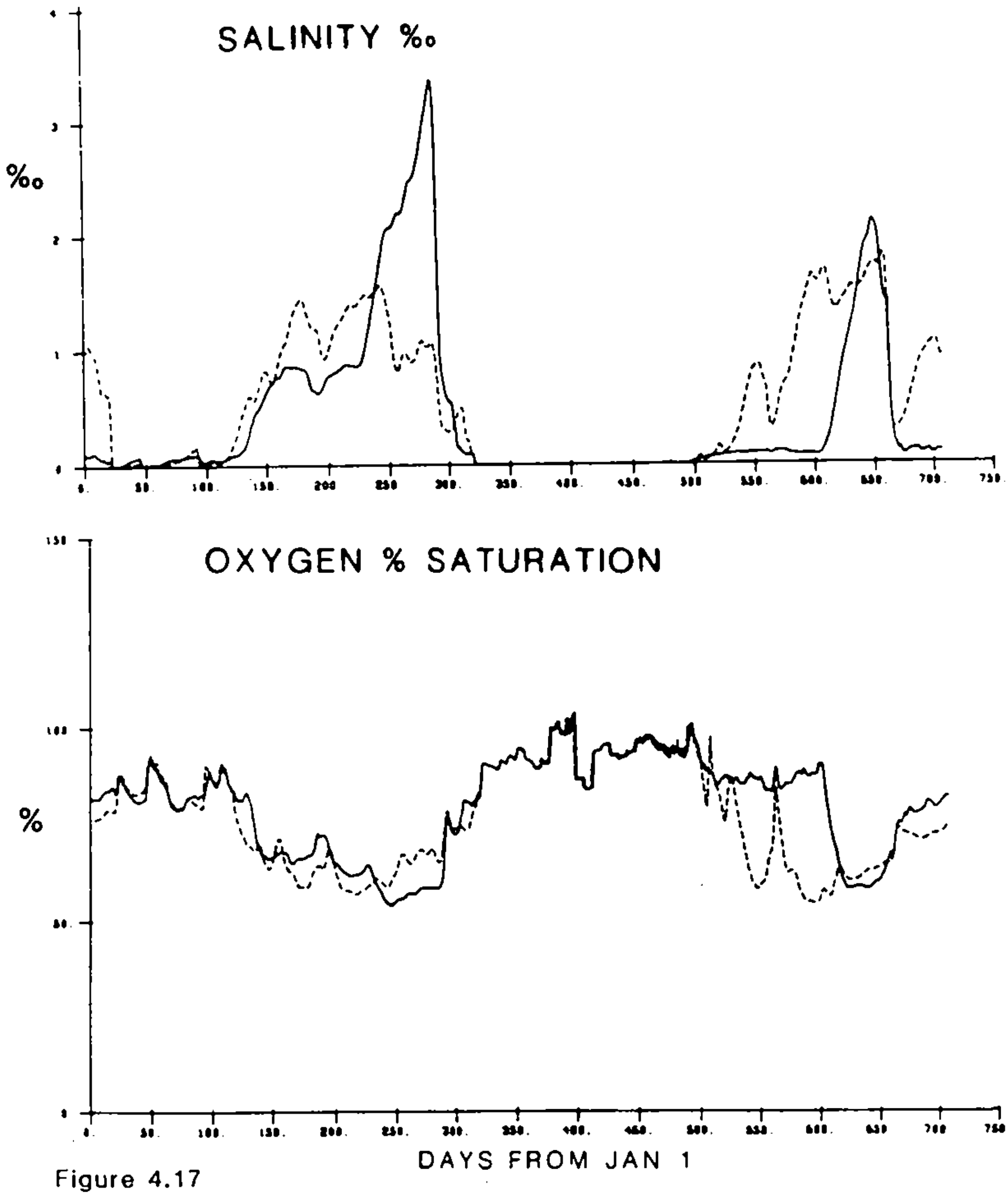


Figure 4.17
Comparison of predicted concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rates according to the chart vs MRB Policy

Current Chart ———— MRB Policy - - - - - - -

CLASSICAL DROUGHT SEQUENCE 1944/45

SLICE 11 LONDON BRIDGE

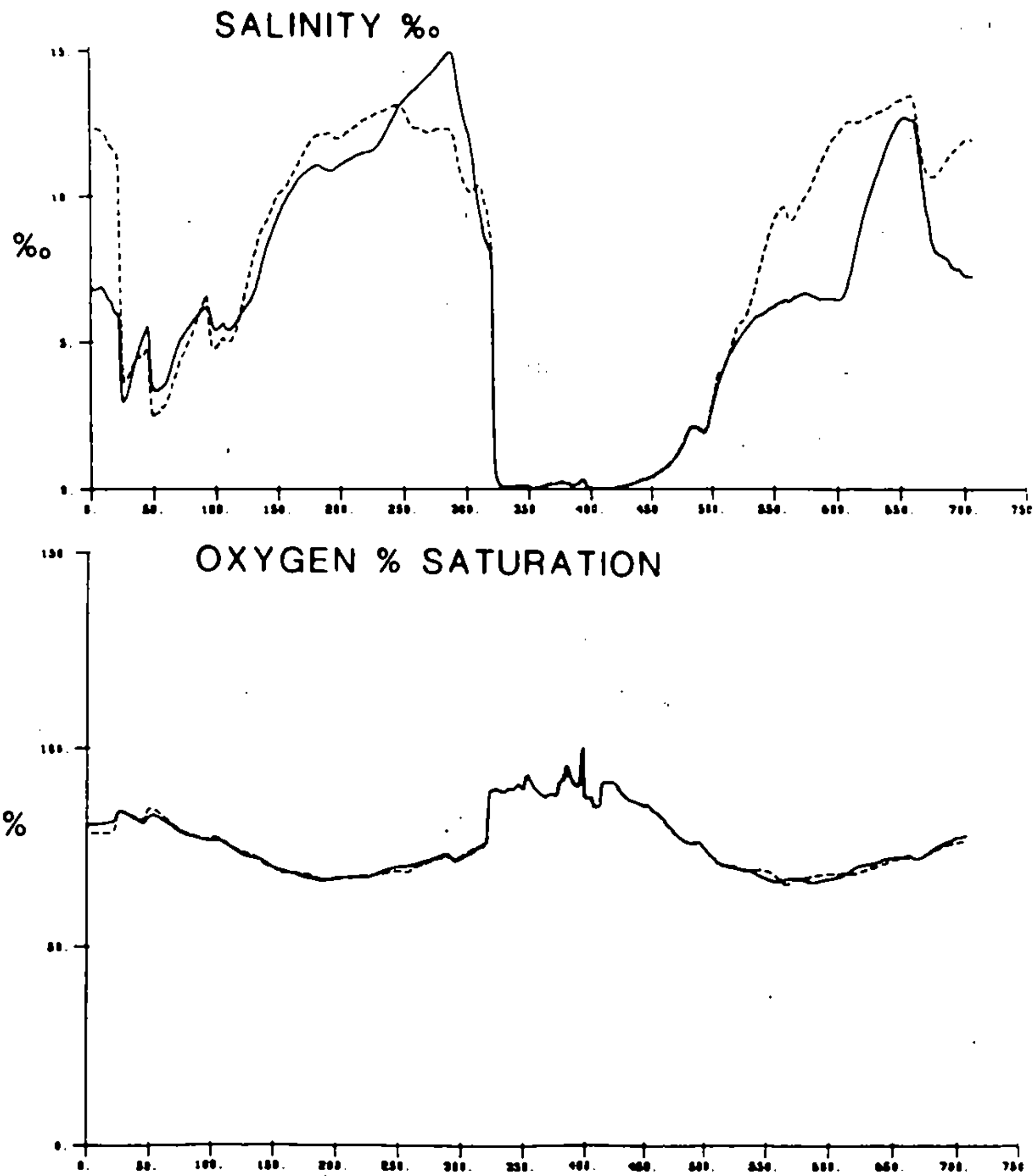


Figure 4.18 DAYS FROM JAN 1

Comparison of predicted concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rates according to the chart vs. MRB Policy

Current Chart ——— MRB Policy - - - - - -

NON DROUGHT SEQUENCE 1952/53

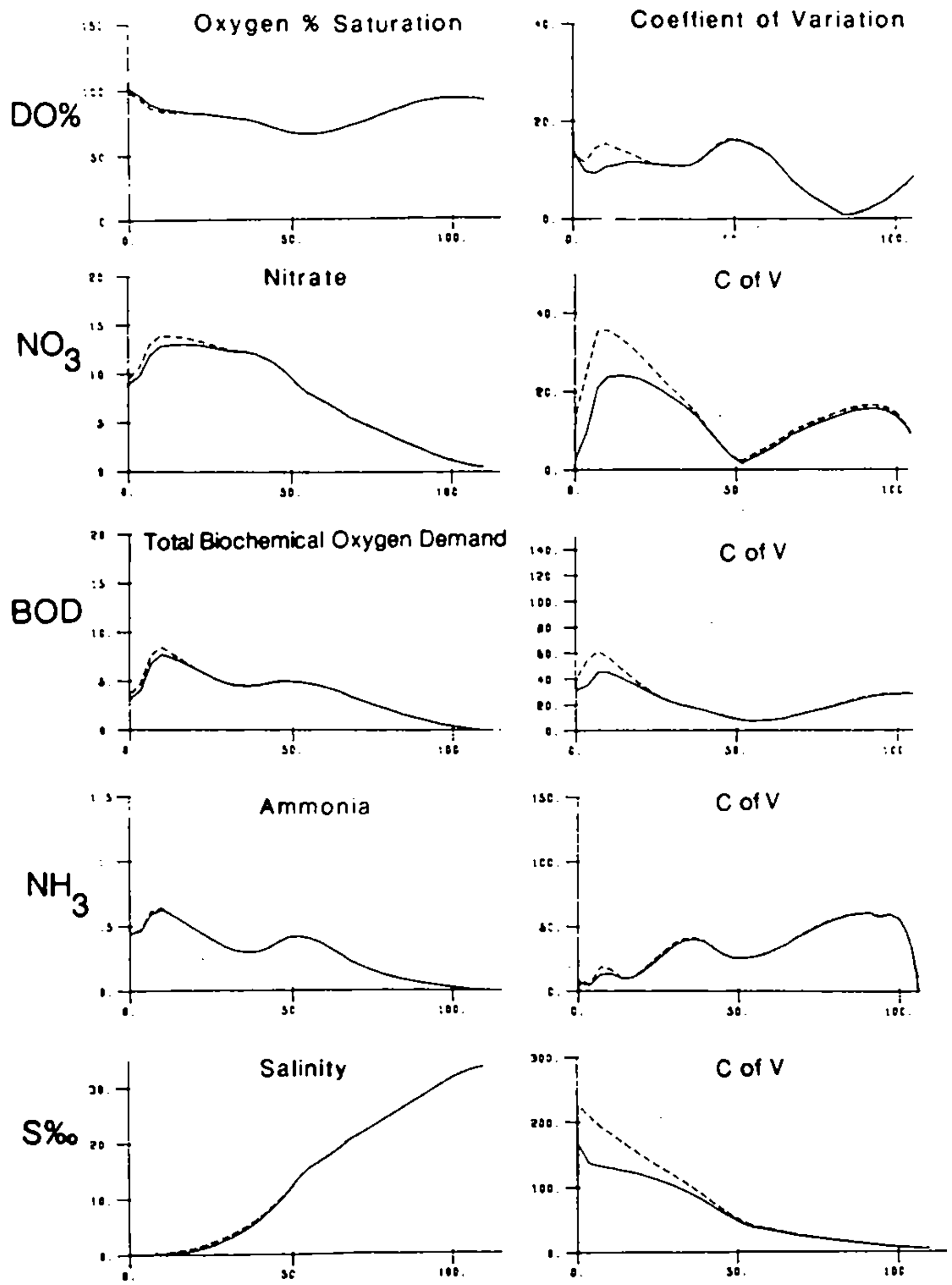


Figure 4.19
Comparison of predicted average axial concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rules according to current chart vs. the Maximum Resource Benefit Policy (MRB)

Current chart ________ MRB Policy - - - - - -

NON DROUGHT SEQUENCE 1952/53

SLICE 4 SYON PARK

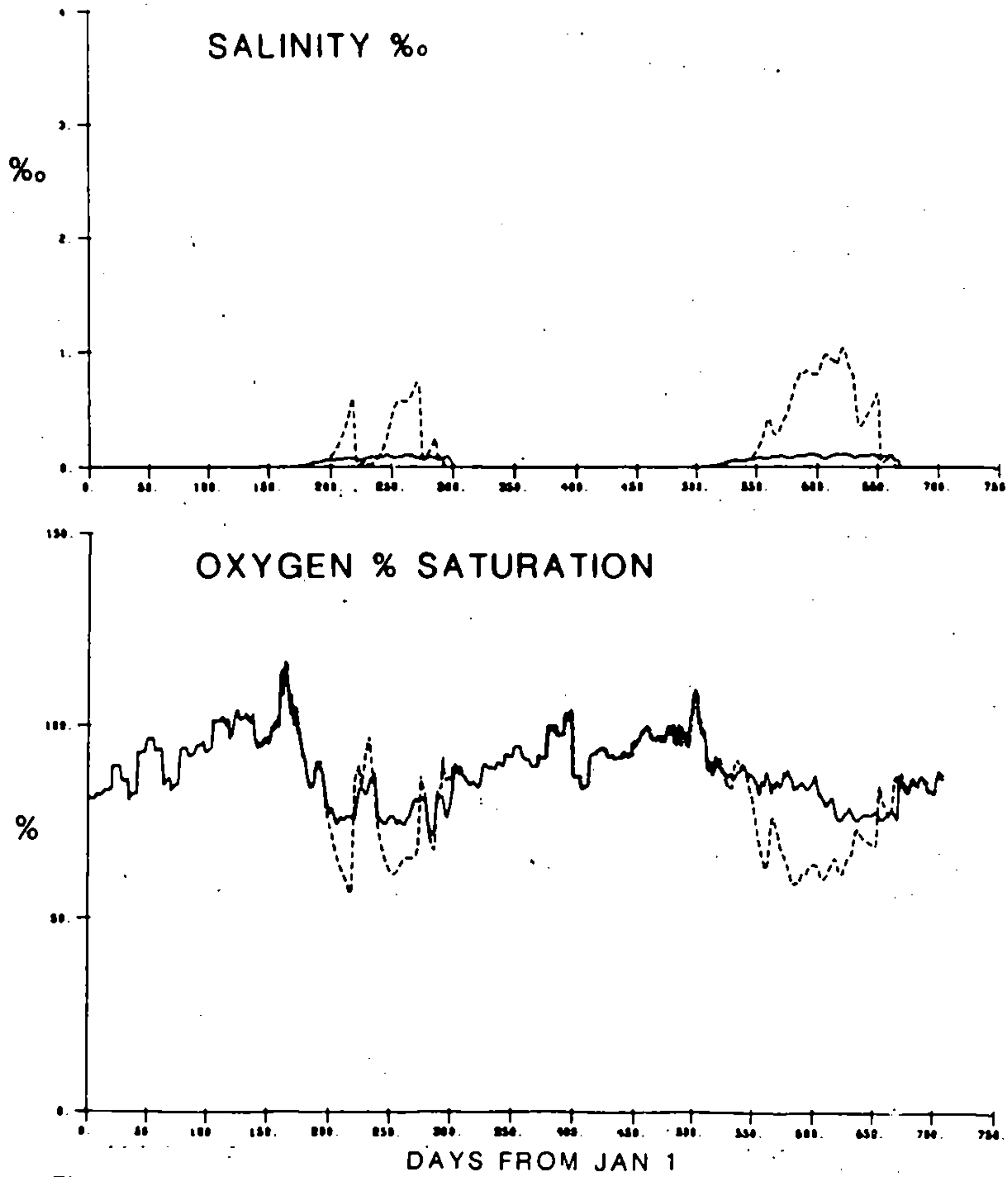


Figure 4.20

Comparison of predicted concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rates according to the chart vs. MRB Policy

Current Chart ———— MRB Policy - - - - - -

NON DROUGHT SEQUENCE 1952/53

SLICE 11 LONDON BRIDGE

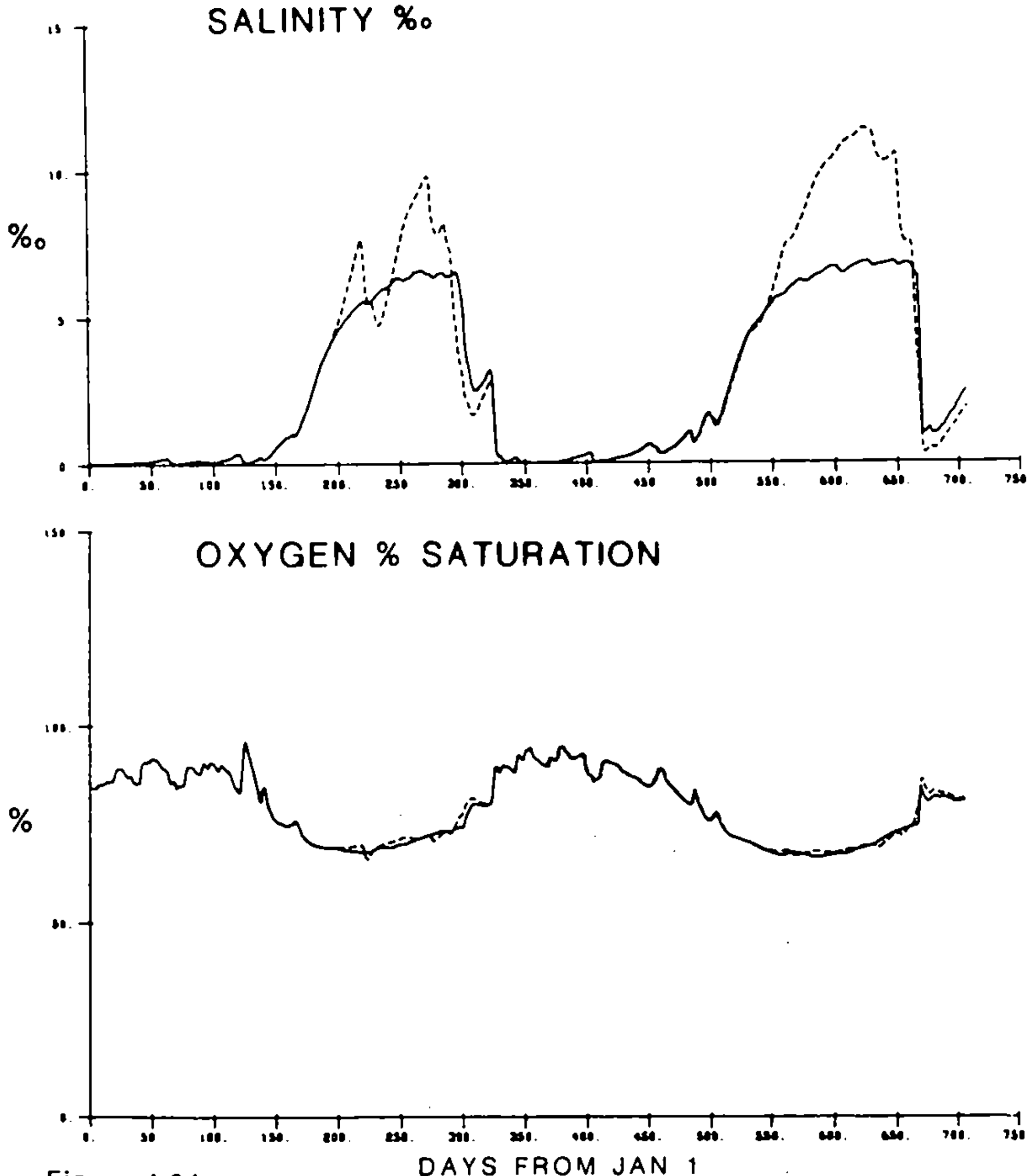


Figure 4.21

Comparison of predicted concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rates according to the chart vs. MRB Policy

Current Chart ——————— MRB Policy - - - - - - -

NON DROUGHT SEQUENCE 1980/81

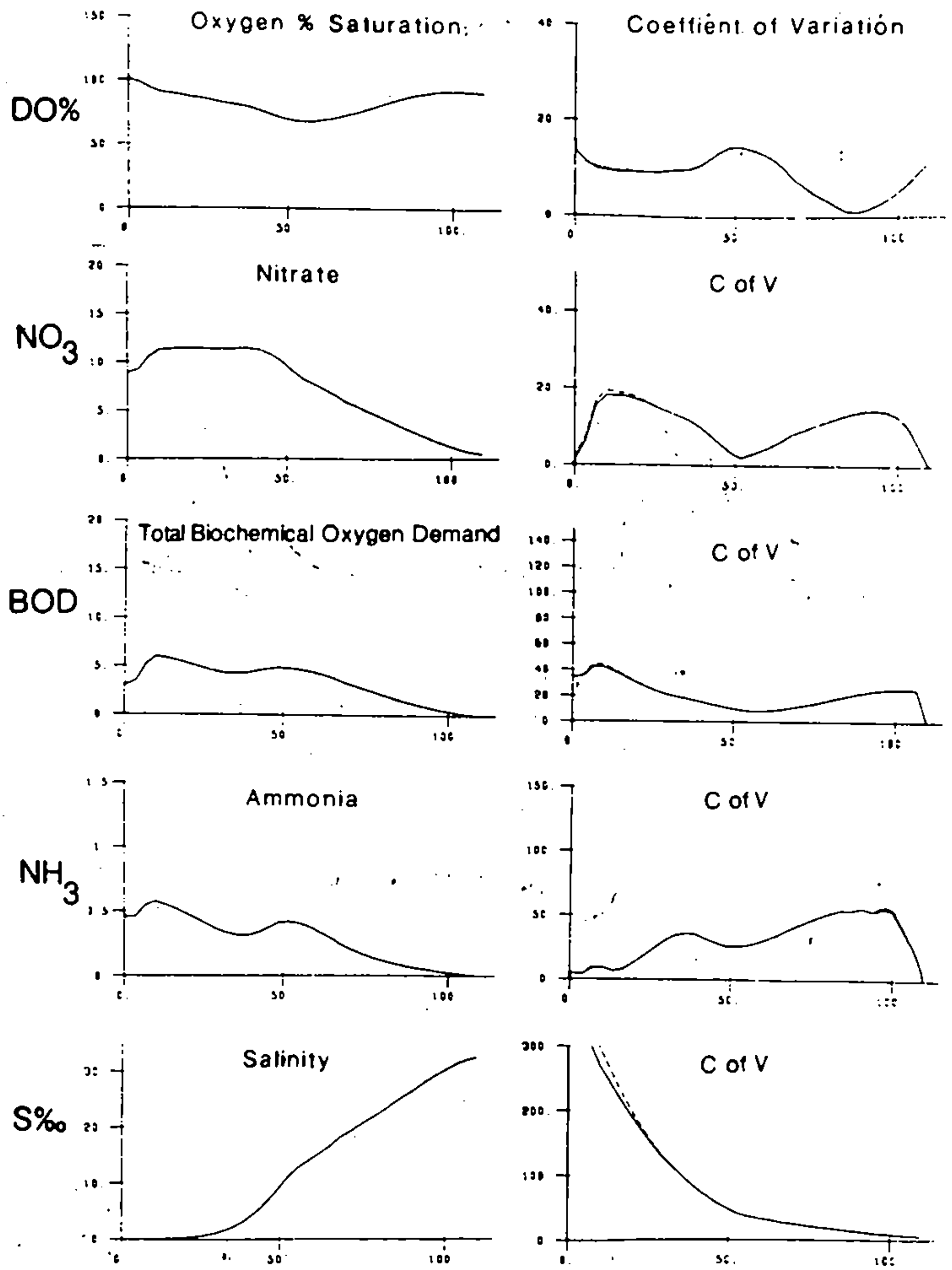


Figure 4.22

Comparison of predicted average axial concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rules according to current chart vs. the Maximum Resource Benefit Policy (MRB)

Current chart ———— MRB Policy – – – – –

NON DROUGHT SEQUENCE 1980/81

SLICE 11 LONDON BRIDGE

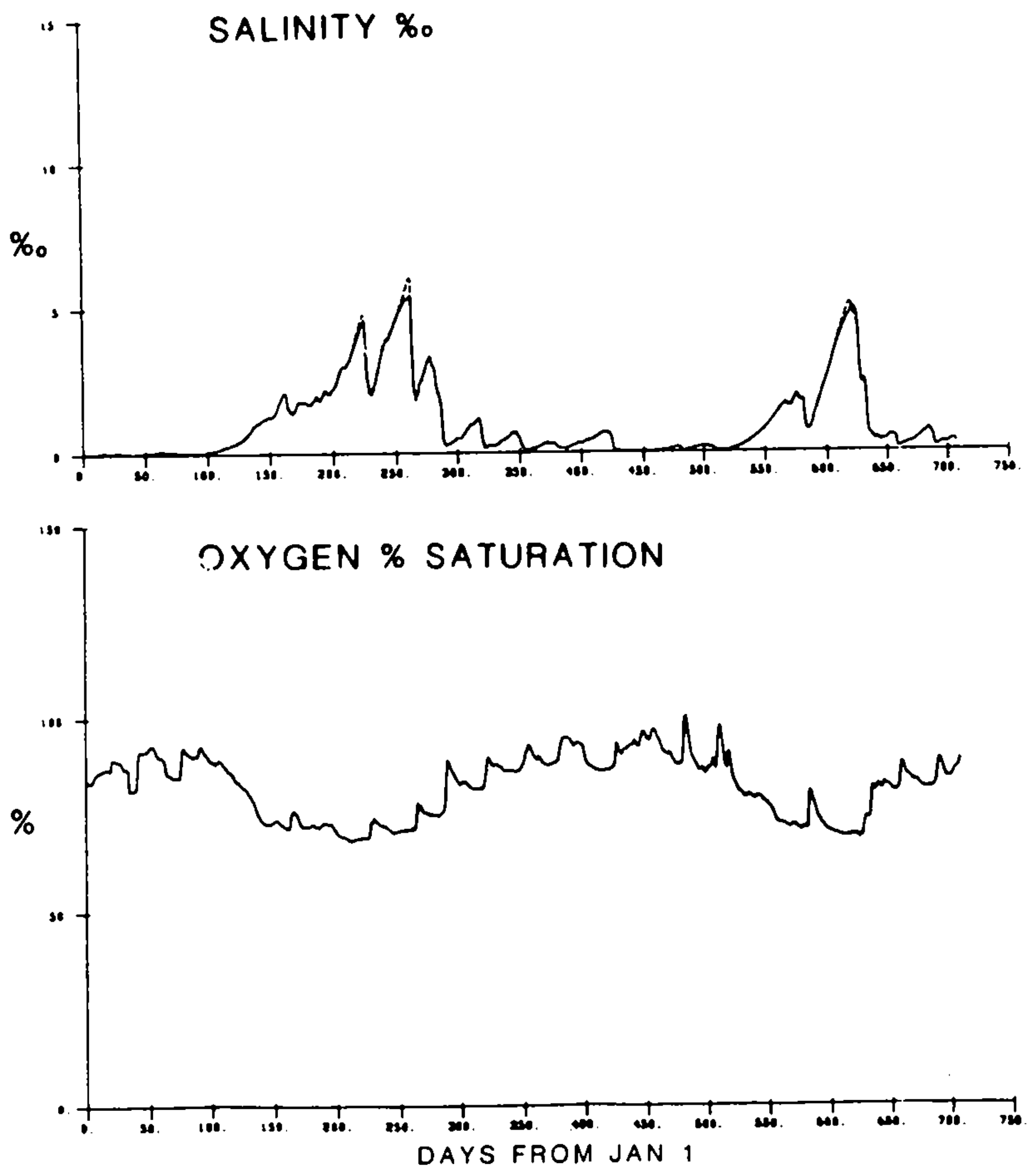


Figure 4.23
Comparison of predicted concentrations for the Thames under current demands and polluting loads, using present Teddington residual flow rates according to the chart vs. MRB Policy

Current Chart ———— MRB Policy - - - - - - -

CLASSICAL DROUGHT SEQUENCE 1975/76

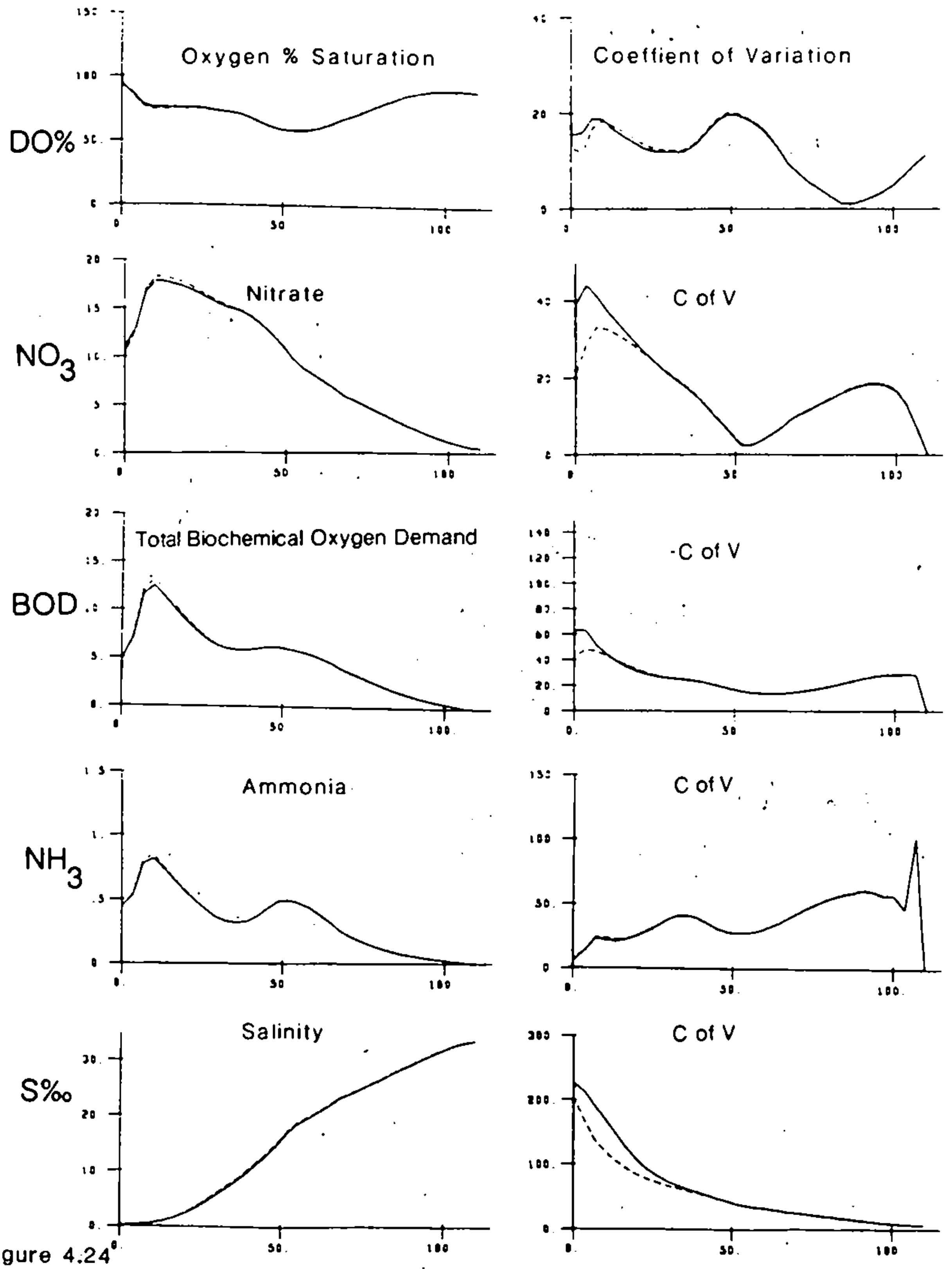


Figure 4.24

Comparison of predicted average axial concentrations for the Thames under "2006" demands and polluting loads, using "2006" Teddington residual flows according to current chart vs. the Maximum Resource Benefit Policy (MRB)

Current chart ______ MRB Policy ------

CLASSICAL DROUGHT SEQUENCE 1975/76

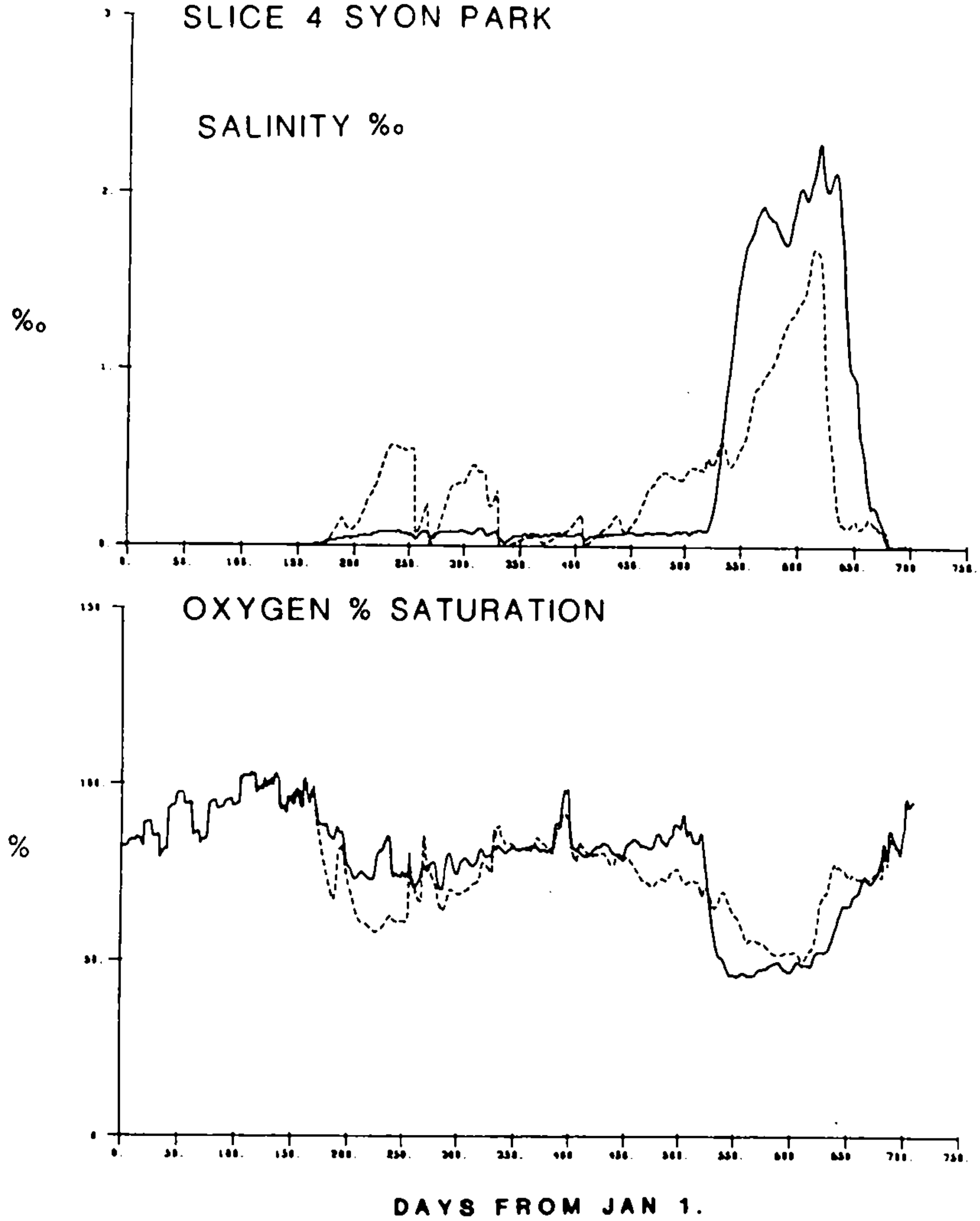


Figure 4.26

Comparison of predicted concentrations for the Thames under "2006" demands and polluting loads, using "2006" Teddington residual flows according to the Chart vs."MRB Policy.

The Chart ———— **- - - - - - - - "MRB"**

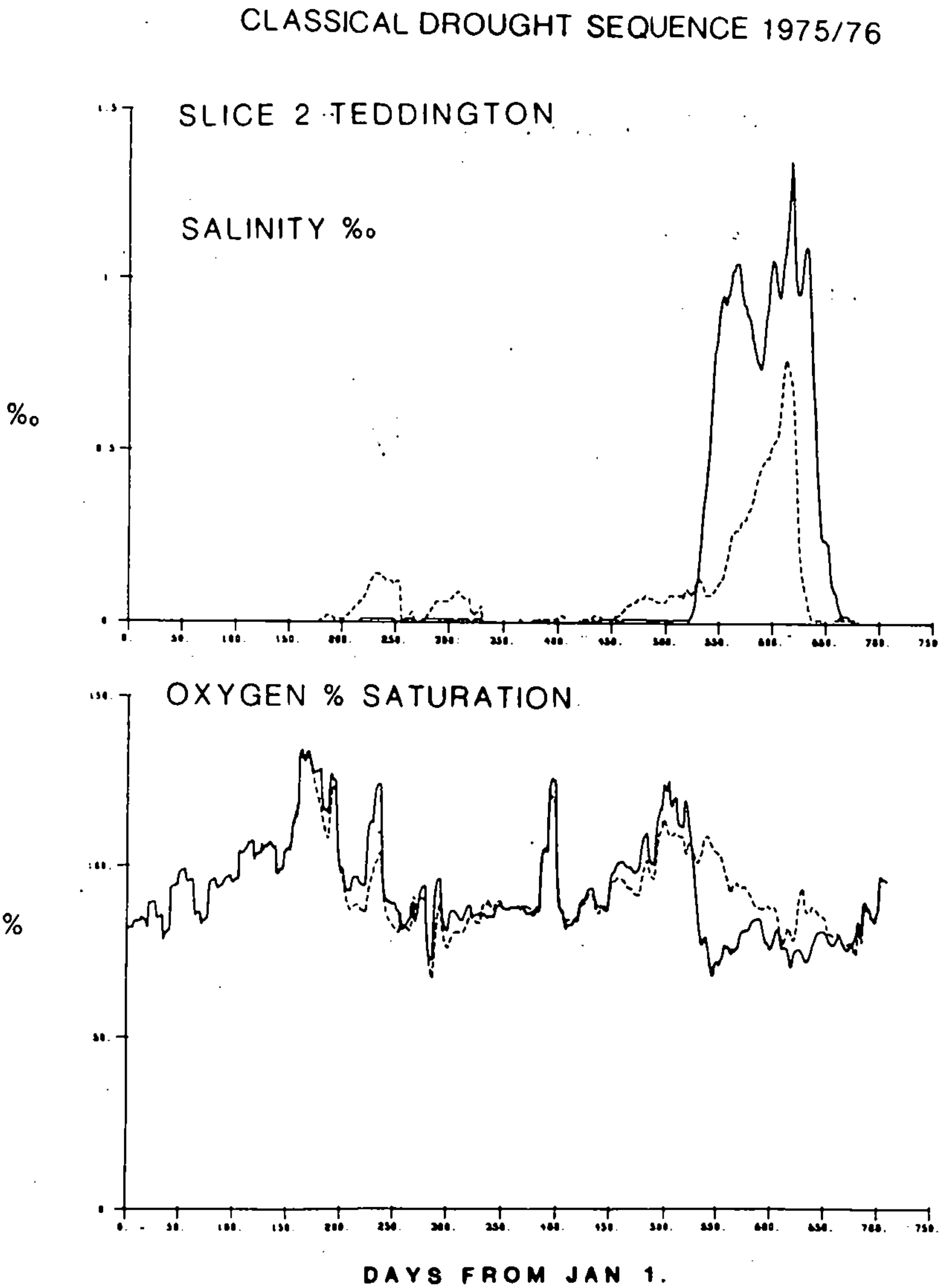


Figure 4.25

Comparison of predicted concentrations for the Thames under "2006" demands and polluting loads, using "2006" Teddington residual flows according to the Chart vs."MRB Policy.

The Chart ————   – – – – – – **"MRB"**

CLASSICAL DROUGHT SEQUENCE 1975/76

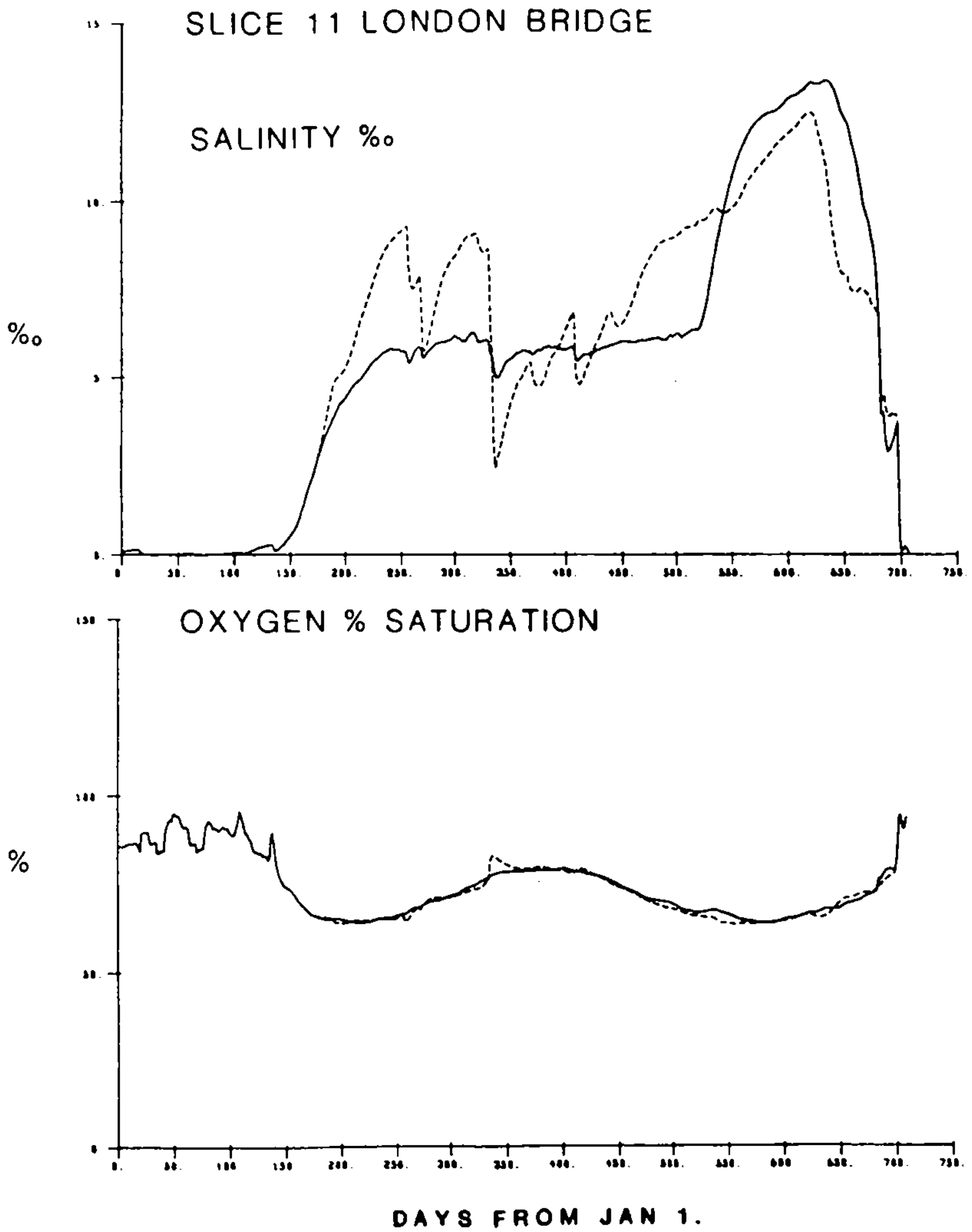


Figure 4.27

Comparison of predicted concentrations for the Thames under "2006" demands and polluting loads, using "2006" Teddington residual flows according to the Chart vs."MRB Policy.

The Chart ———— ------- **"MRB"**

CLASSICAL DROUGHT SEQUENCE 1944/45

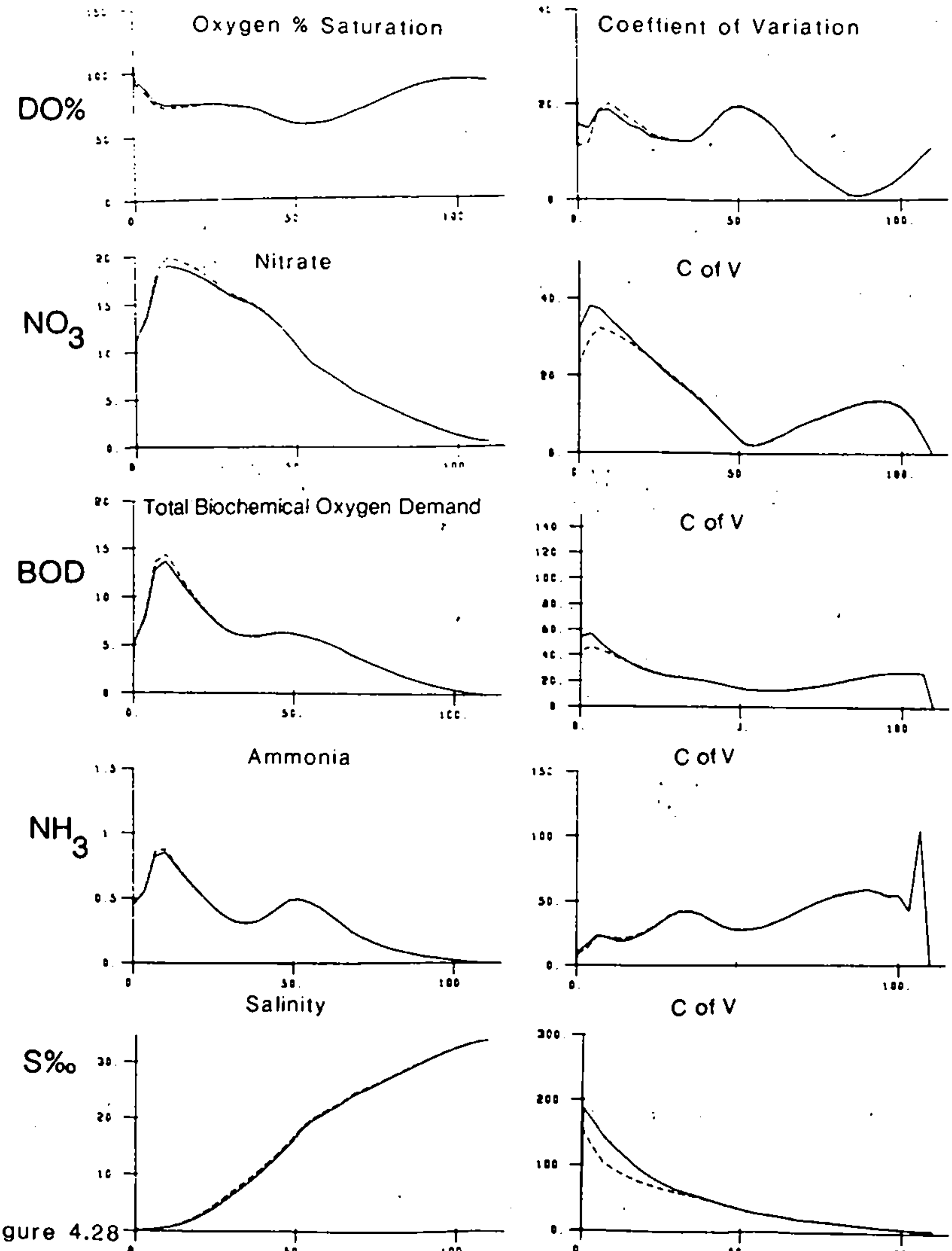


Figure 4.28
Comparison of predicted average axial concentrations for the Thames under "2006" demands and polluting loads, using "2006" Teddington residual flows according to current chart vs. the Maximum Resource Benefit Policy (MRB)

Current chart ________ MRB Policy — — — — —

CLASSICAL DROUGHT SEQUENCE 1944/45

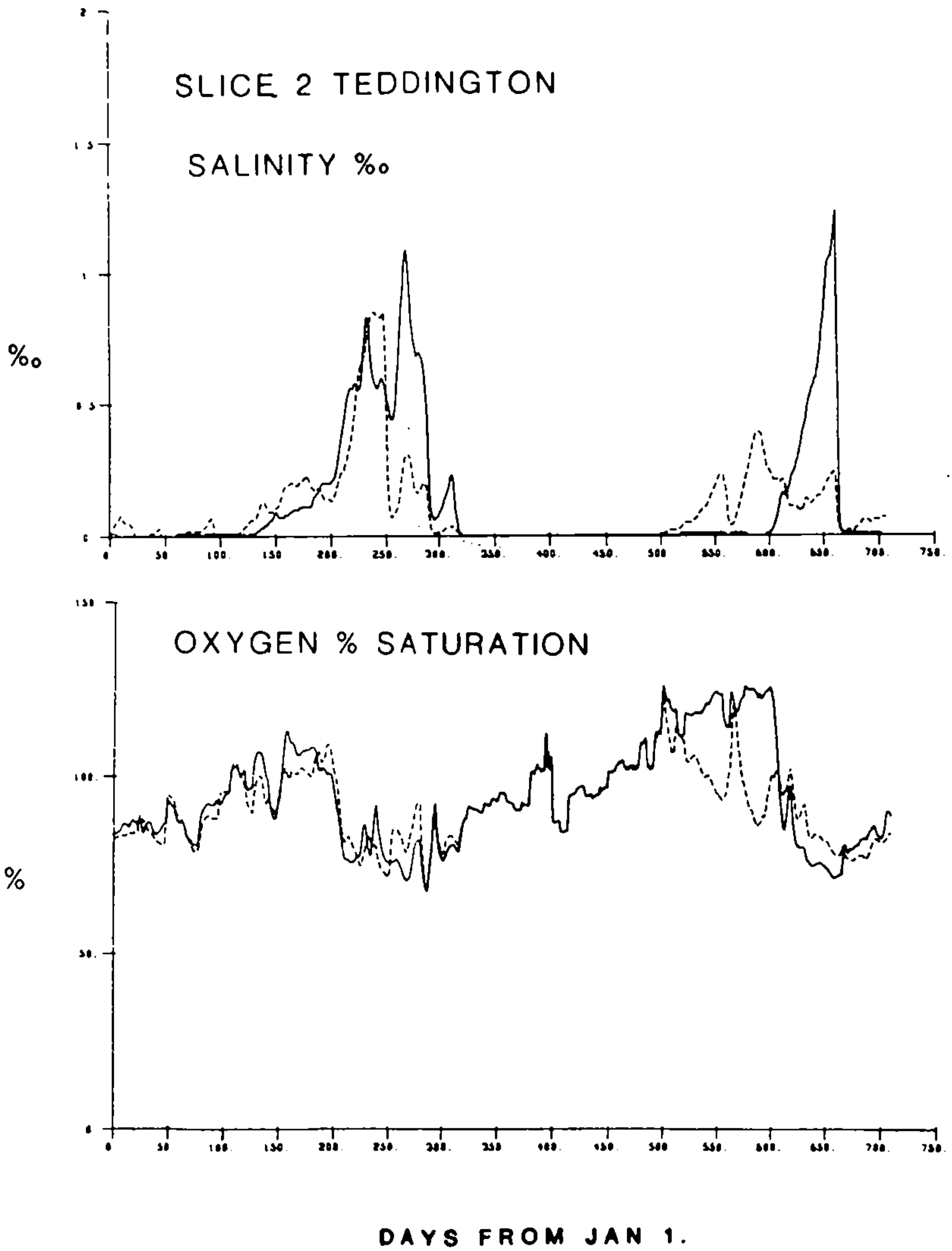


Figure 4.29

Comparison of predicted concentrations for the Thames under "2006" demands and polluting loads, using "2006" Teddington residual flows according to the Chart vs. "MRB Policy.

The Chart ————— - - - - - - - "MRB"

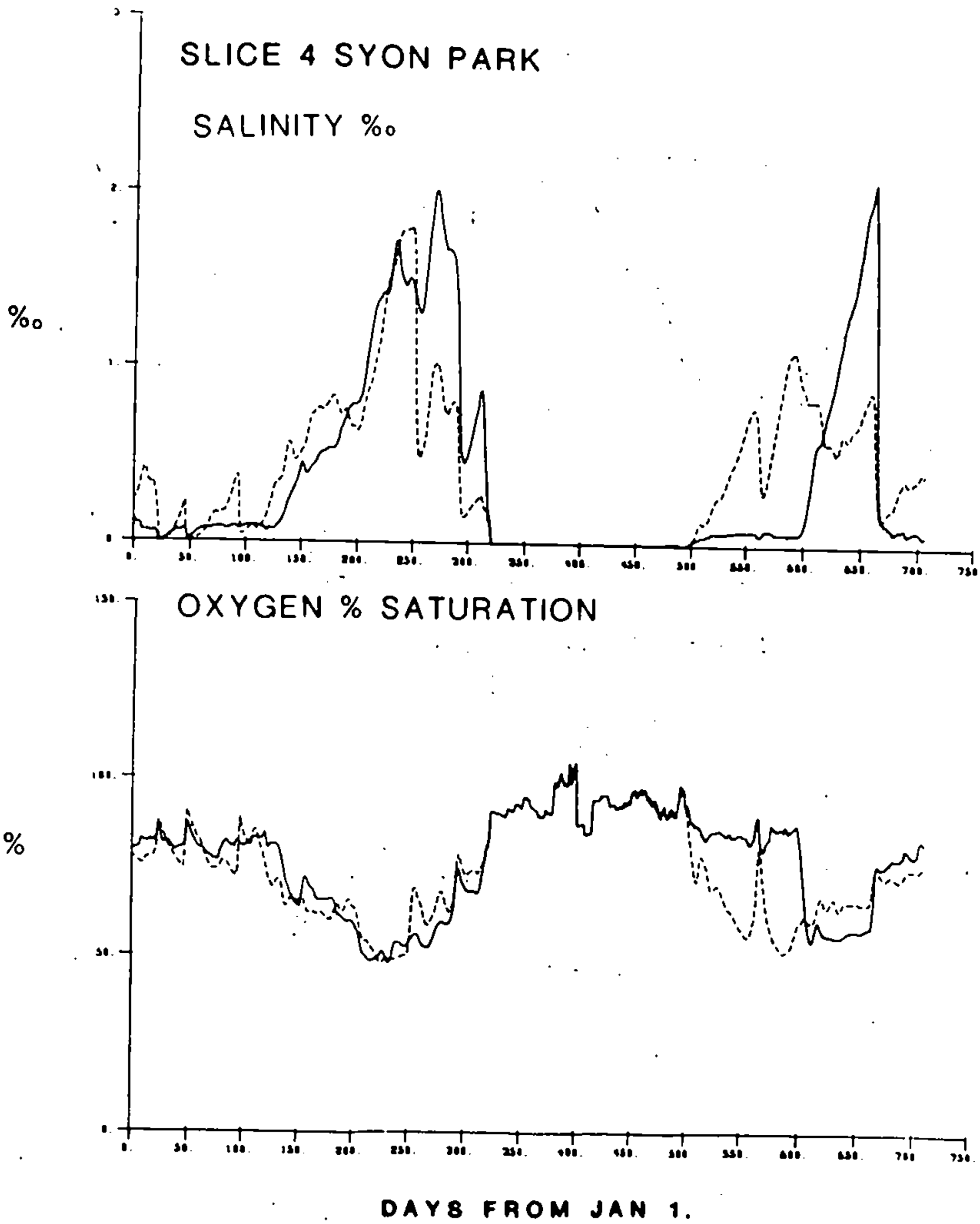


Figure 4.30

Comparison of predicted concentrations for the Thames under "2006" demands and polluting loads, using "2006" Teddington residual flows according to the Chart vs. "MRB Policy.

The Chart ———— ------- **"MRB"**

CLASSICAL DROUGHT SEQUENCE 1944/45

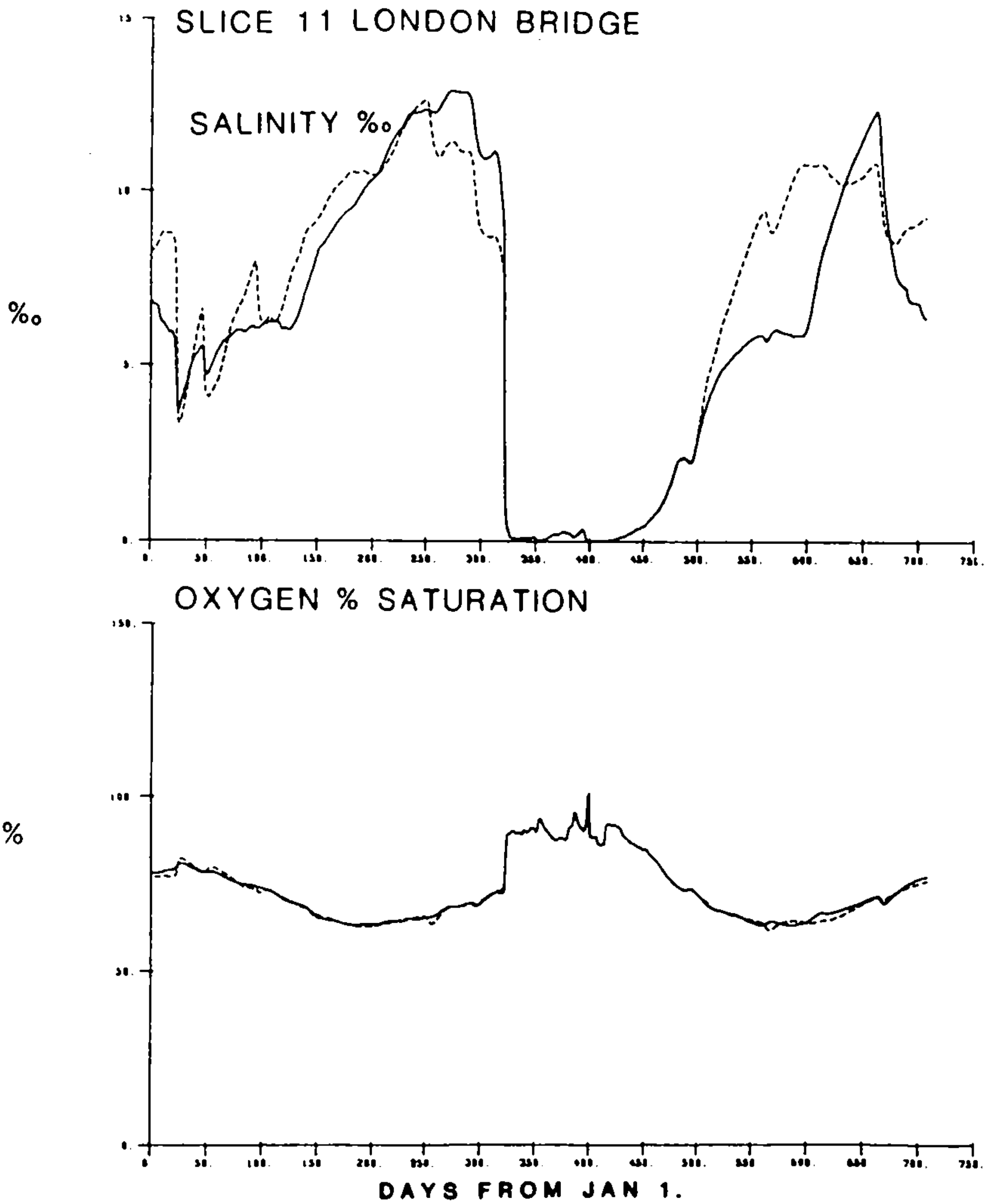


Figure 4.31

Comparison of predicted concentrations for the Thames under "2006" demands and polluting loads, using "2006" Teddington residual flows according to the Chart vs."MRB Policy.

The Chart ——————— - - - - - - - - **"MRB"**

NON DROUGHT SEQUENCE 1952/53

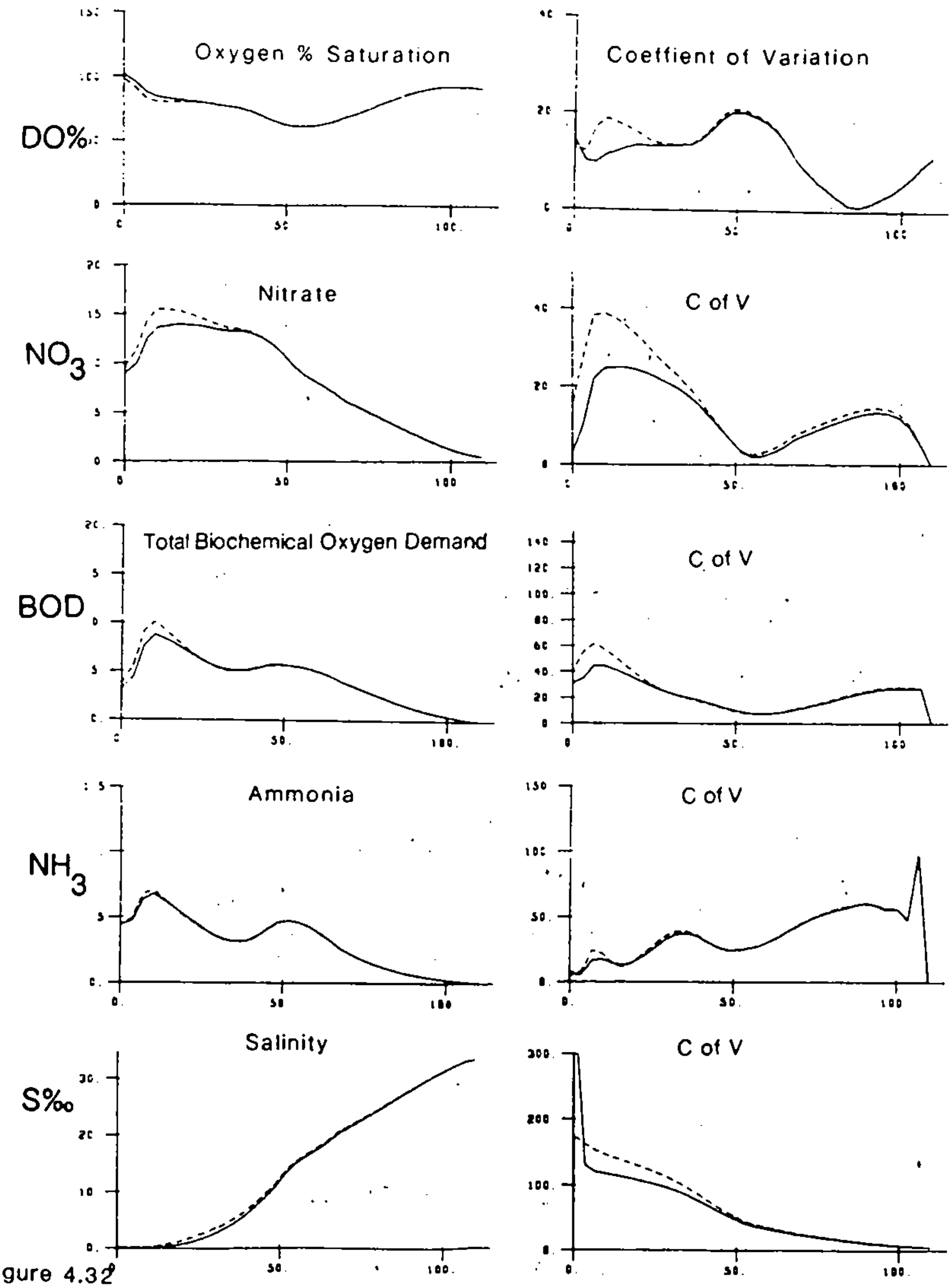


Figure 4.32

Comparison of predicted average axial concentrations for the Thames under "2006" demands and polluting loads, using "2006" Teddington residual flows according to current chart vs. the Maximum Resource Benefit Policy (MRB)

Current chart ———— MRB Policy - - - - - -

NON DROUGHT SEQUENCE 1952/53

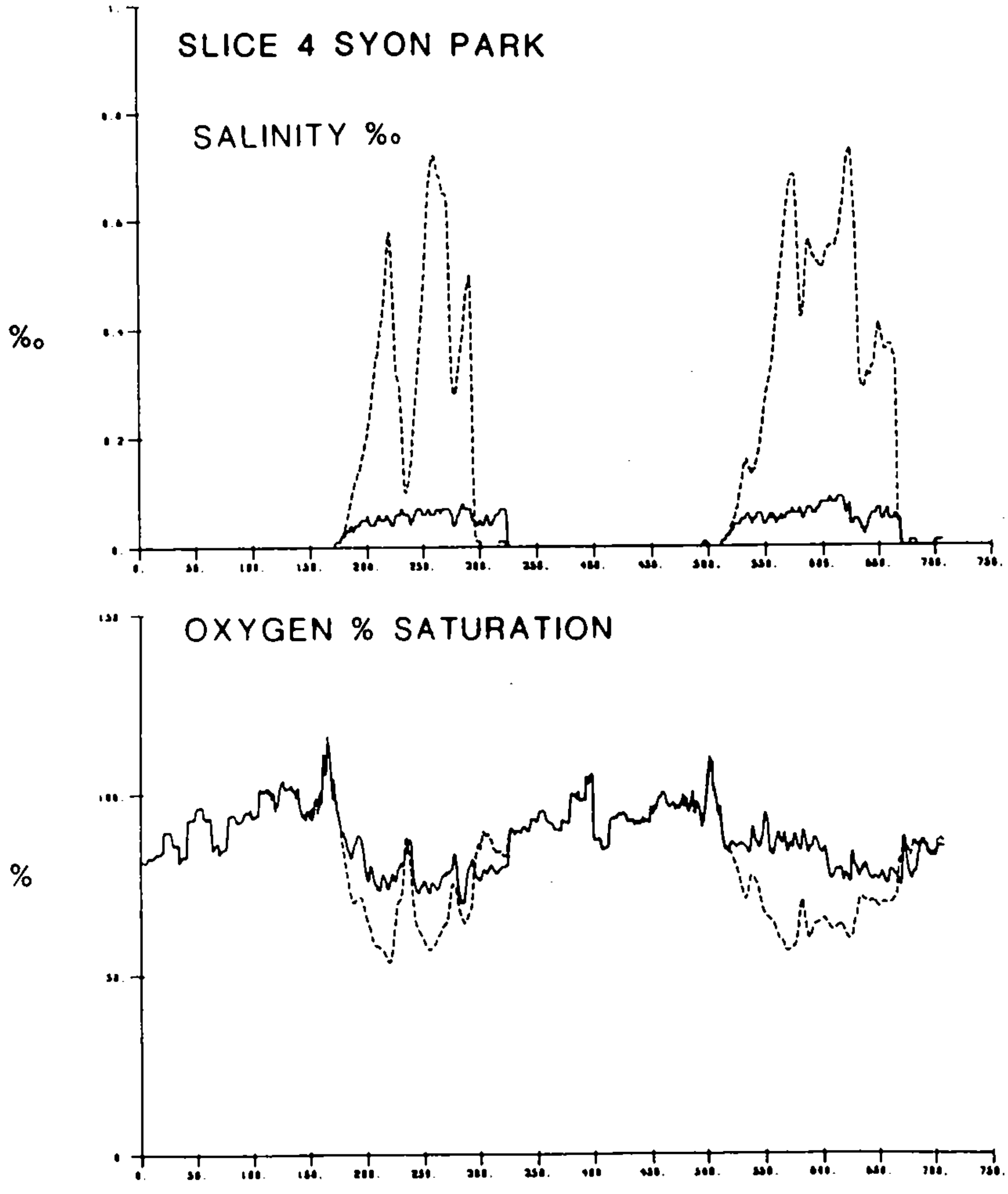


Figure 4.33

Comparison of predicted concentrations for the Thames under "2006" demands and polluting loads, using "2006" Teddington residual flows according to the Chart vs."MRB Policy.

The Chart ———— - - - - - - - "MRB"

NON DROUGHT SEQUENCE 1952/53

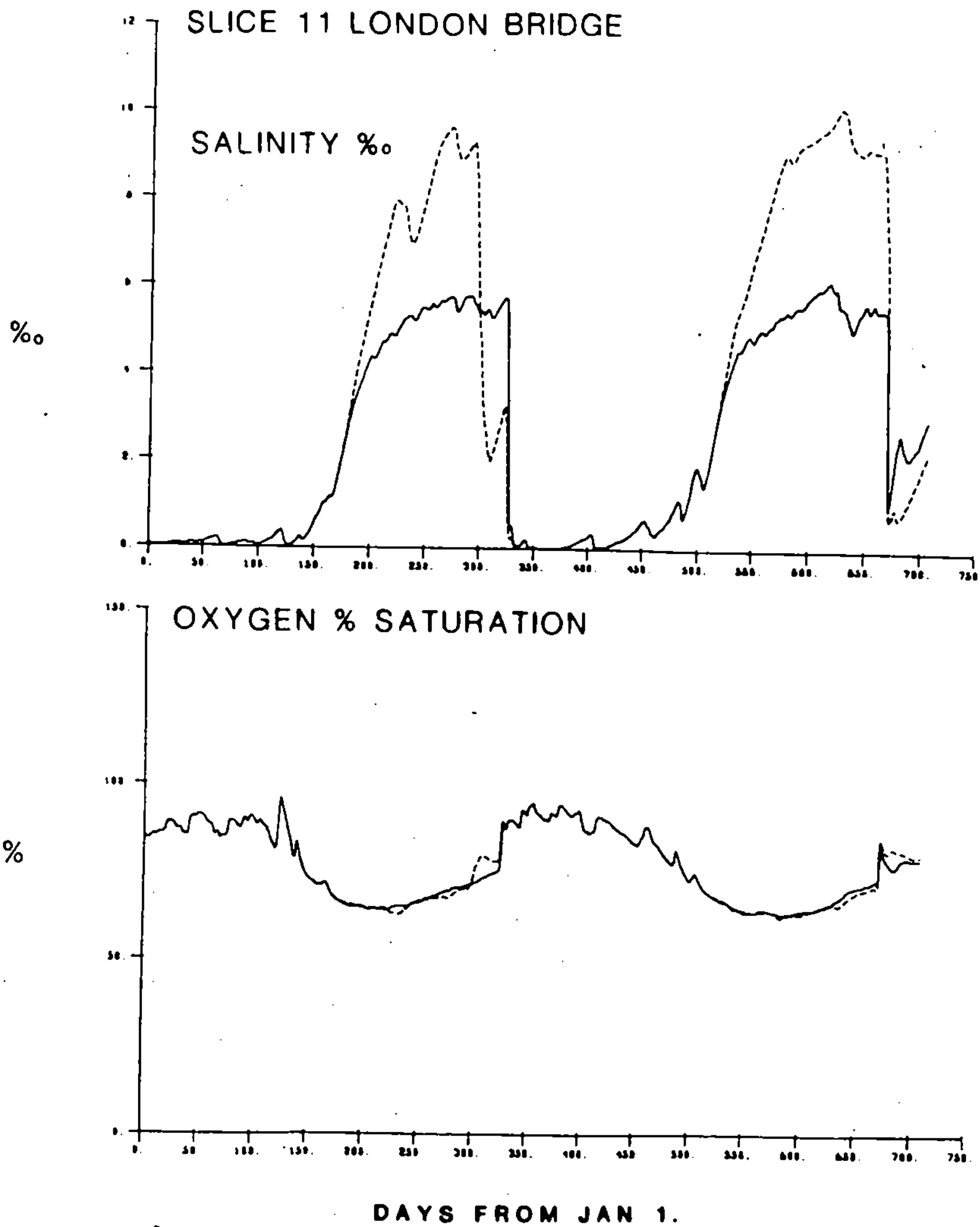


Figure 4.34

Comparison of predicted concentrations for the Thames under "2006" demands and polluting loads, using "2006" Teddington residual flows according to the Chart vs."MRB Policy.

The Chart ———————— - - - - - - - - "MRB"

NON DROUGHT SEQUENCE 1980/81

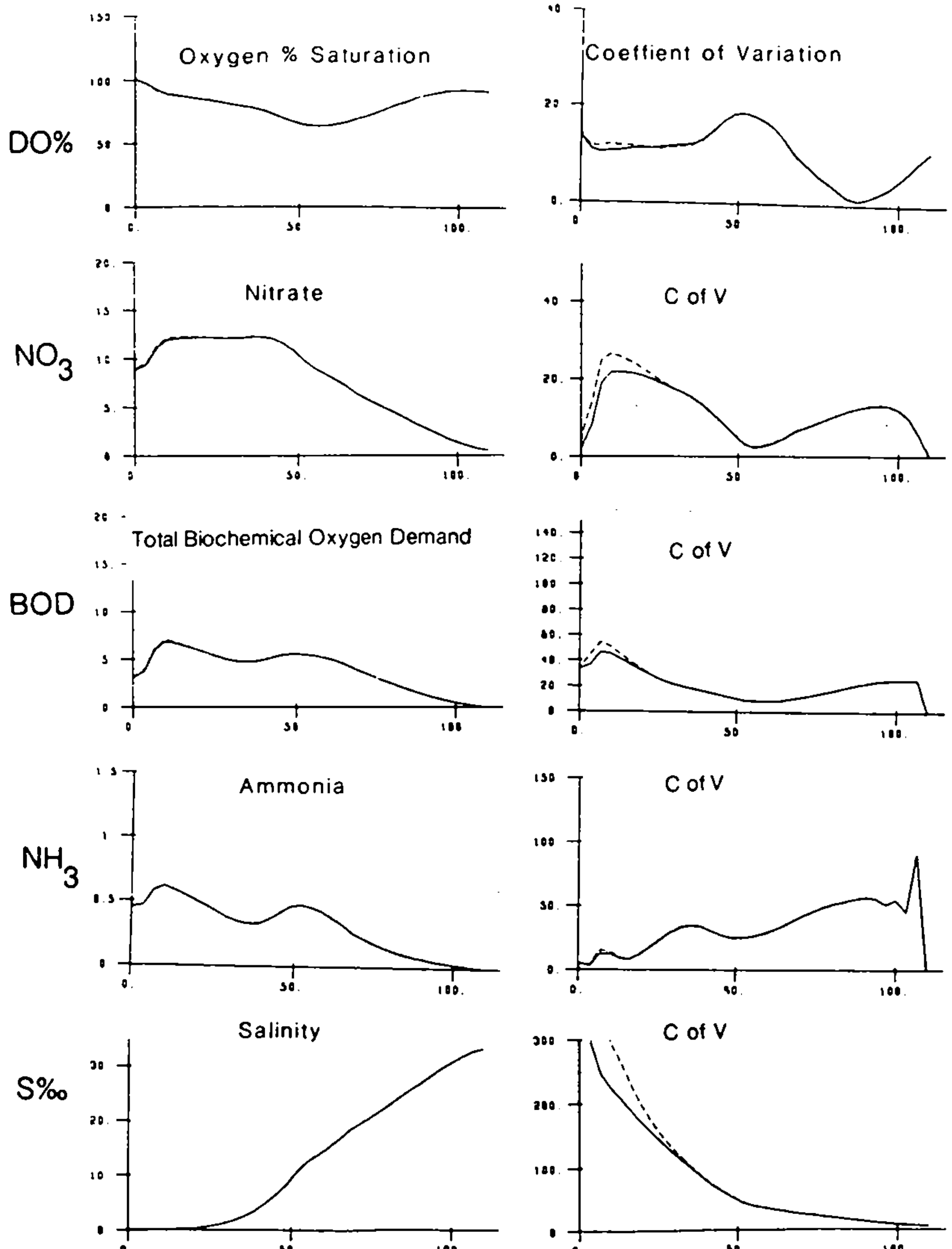


Figure 4.35
Comparison of predicted average axial concentrations for the Thames under "2006" demands and polluting loads, using "2006" Teddington residual flows according to current chart vs. the Maximum Resource Benefit Policy (MRB)

Current chart ———— MRB Policy - - - - - -

NON DROUGHT SEQUENCE 1980/81

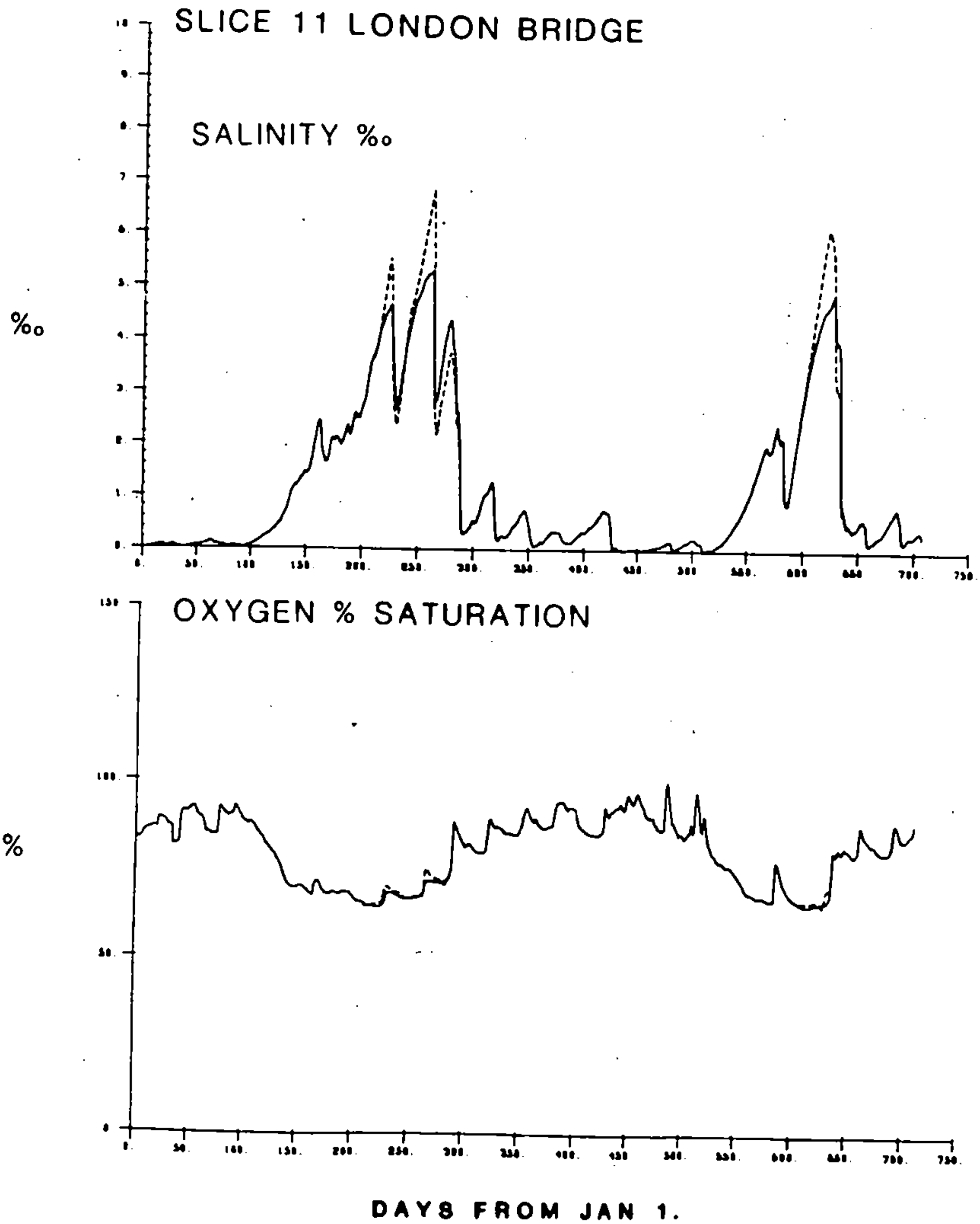


Figure 4.36

Comparison of predicted concentrations for the Thames under "2006" demands and polluting loads, using "2006" Teddington residual flows according to the Chart vs."MRB Policy.

The Chart ———————   – – – – – – – "MRB"

4.7.3. The deleterious effects of changing from the present "Chart" method of water abstraction to a maximum resource benefit (MRB) policy are very small compared to the improvements caused by the recent introduction of modern water treatment plant at a number of discharge sites.

4.7.4. In general the effect of introducing the MRB policy is to produce a slightly inferior water quality in the early stages of a drought sequence (e.g. 1975/76, 1944/45) balanced by a slightly improved quality at the height of the drought. None of these impacts are large and they are only evident upestuary of London Bridge.

4.7.5. Under less severe drought conditions (e.g. 1952/53) the inferior water quality produced earlier in the year is not compensated for when the full drought does not materialise. In this way a relative deteriaration would occur in the estuary during such sequences but at a time when water quality is relatively high.

4.7.6. When the system experiences very high river run-off conditions (e.g. 1980/81) there are no significant differences in water quality caused by the two methods of abstraction. This is true throughout the Thames Estuary from Teddington to Shoebury Ness.

4.7.7. For both regimes a slight reduction in water quality is predicted under the increased water demand anticipated for the year 2006. This deterioration is almost entirely due to the increased sewage discharges that, it is assumed, would accompany the increased water use. The salinity distribution does not change significantly because the extra water abstracted is replaced by increased sewage effluent, particularly that from the rivers Mole and Hogsmill.

4.7.8. The relative effects of the two regimes are very similar under the increased (2006) demands as under present (1984) demands. No serious deterioration in estuarine water quality is evident that might be important to the well being of the ecosystem as a whole. The gradual but continual increase in the highest mean nitrate concentration over the years might be of concern if it were to be used as drinking water but brackish water is a very unlikely source for this. There is no known deleterious effect of these levels of nitrate on estuarine ecosystems.

4.7.9. Changes in the instanoneous flow regime in the estuary would be only marginally affected by the adoption of the MRB policy since the velocities induced by tidal movements are much greater than those caused by residual river flows. Similarly the resultant suspended particulants in the water column would be very similar under the two regimes.

4.7.10. The location of the turbidity maximum, which relates to the positions of the greatest resuspension and deposition (Bale et al, In press) would change only very slightly if the MRB policy were adopted, and this change would be very small compared to the natural movements experienced within any annual cycle.

4.8 REFERENCE

Bale A.J., Morris, A.W. and Howland R.J.M. (in press) Seasonal sediment movement in the Tamar Estuary. Oceanol Acta.

Barrett M.J., Mallowney B.M. and Casapieri P. (1978). The Thames Model: Ann Assessment Prog. Wat. Tech. 10, 409-416.

Department of Scientific and Industrial Research (1964) Effects of polluting discharges on the Thames Estuary Water Pollution Research Technical Paper No 11 H.M. Stationary Office, London pp 609.

Hydraulic Research, (1983) Tolo Harbour water quality model QUEST2. A three dimensional water quality and ecosystem model. Ex 1140 pp 175.

Maskell J.M. and Odd, N.V.M. (1977) A mathematical model of the oxygen balance of a well mixed estuary. Hydraulics Research Station, Wallingford. Reprint IT171.

Mallowney B.M. (1982a) Mathematical Modelling. In The Quality of the Humber Estuary (Ed A.L.H. Gameson) Yorkshire Water Authority 82-88.

Mallowney B.M. (1982b) The Moving - Segment Steady State Model of dissolved oxygen in the Humber Tidal System. Report for the Humber Estuary Technical Working Party. WRC Report No. 366-M.

Radford P.J. (1980) Severn Tidal Power. Predicted effects of proposed tidal power schemes upon the Severn Estuary Ecosystem. Vol.1 Water Quality Institute for Marine Environmental Research. Department of Energy contract agreement No E/5A/CON/5B/1598/51/036.pp55.

Radford P.J. and Joint I.R. (1980) The application of an ecosystem model to the Bristol Channel and Severn Estuary. Institute of Water Pollution Control. Annual Conference. Water Pollution Control 79,244-254.

Radford P.J. Uncles R.J. and Morris A.W. (1981) Simulating the impact of technological changes on dissolved cadmium distributions in the Severn Estuary. Water Research 15 1045-1052.

Sexton J.R., Cook D.J. and Jones A.E.(1979) Thames Water Resources Model, An Introduction publ. Thames Water.

Uncles R.J. and Radford, P.J. (1980) Seasonal and spring-neap tidal dependence of axial dispersion coefficients in the Severn - a wide vertically mixed estuary J.Fluid Mechanics, 98 703-736.

Whitehead P.G. and Williams R.J. (1982) A dynamic nitrogen balance model for river systems. In Effects of Waste Disposal on Groundwater and Surface Water. proc.symp IAHS p.89-99.

APPENDIX 4.1. The Thames Model. An Assessment.

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Prog. Wat. Tech. 1978, Vol. 10, Nos 5/6, pp. 409-416. Pergamon Press. Printed in Great Britain.

THE THAMES MODEL: AN ASSESSMENT

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Abstract
Mathematical methods, developed in the late 1950s, for predicting the effects of pollution on the Thames Estuary are reviewed, and the adequacy of the methods assessed by comparing observed and predicted distributions of dissolved oxygen, and ammoniacal and oxidized nitrogen over a period in which there has been a marked reduction in polluting load.

INTRODUCTION

A principal aim of the Thames investigations carried out by the Water Pollution Research Laboratory (now the Water Research Centre, Stevenage Laboratory) some years ago was the development of a mathematical model to allow the effects of polluting discharges to be predicted for any combination of conditions that might arise in the future (1). The model - which was validated by comparing predictions with the observed distributions of dissolved oxygen, and ammoniacal and oxidized nitrogen for each three-monthly period from 1950 to 1961 - was subsequently adopted as a basis for control of pollution by the Port of London Authority, and a variant of it is now used routinely as a management tool by the Thames Water Authority.

In this paper the mathematical methods are reviewed and their adequacy in predicting water quality is assessed by comparing observed and calculated distributions of dissolved oxygen and of ammoniacal and oxidized nitrogen over a period during which the polluting load has been progressively reduced.

REPRESENTATION OF MIXING AND MOVEMENT OF THE WATER

The original model was based on a concept (2) which envisaged longitudinal mixing as a statistical redistribution process whereby during each tidal cycle some of the water contained between any two cross-sections, an infinitesimal distance apart, was spread uniformly upstream and some uniformly downstream over a distance equal to about half the tidal excursion, while some remained in (or returned to) the original position. Subsequently, it was found (3) that very similar numerical results are obtained using the simulation adopted by other workers in which longitudinal mixing is regarded as analogous to a diffusion process. As the diffusion model is somewhat simpler mathematically, it was adopted for the Thames.

For the purposes of the model, a detailed description of which has been given by Mollowney (4), the estuary is imagined to be divided by vertical partitions (extending from one bank to the other) into a series of segments which oscillate upstream and downstream with the tidal velocity, such that the volume of water landwards of a given segment boundary is constant. The fresh-water flow, which is assumed to be steady and in equilibrium with the salinity distribution, is represented by a volume flow through each segment into the one immediately downstream, allowance being made in individual segments for the entry of water from tributaries and sewage and industrial effluents. Longitudinal mixing is simulated by assuming a continuous exchange of water between adjacent segments by equal and opposite flows as indicated in Fig. 1 by F_1.

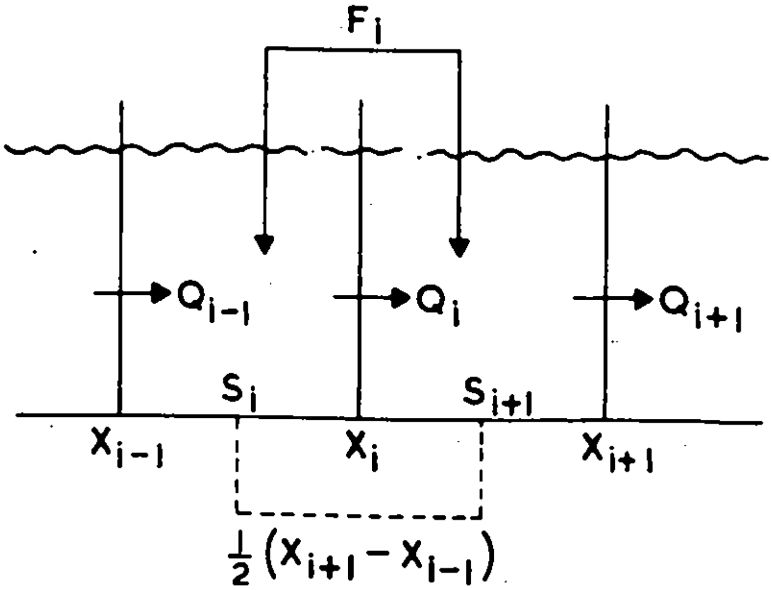

Fig. 1. Representation of longitudinal mixing and fresh-water flow in model.

The exchange flow, F_i, is essentially a finite-difference approximation for the dispersion term in the convective-diffusion equation with

$$F_i = 2E_iA_i/(x_{i+1} - x_{i-1}), \tag{1}$$

where E_i is an effective dispersion coefficient and A_i is the cross-sectional area at x_i. Values of the exchange flow are determined by fitting the model to a known longitudinal salinity distribution. Thus if S_i is the average salinity in the segment x_{i-1} to x_i during a tidal cycle, F_i is given by

$$F_i = Q_iS_i/(S_{i+1} - S_i), \tag{2}$$

where Q_i is the total land-water flow at x_i.

The distribution of a pollutant is found by considering the changes in mass occurring in each segment during a tidal period. Thus if C_i is the concentration in the segment x_{i-1} to x_i, the mass balance in the i-th segment (volume ΔV_i) is given by

$$Q_{i-1}C_{i-1} - Q_iC_i + F_{i-1}(C_{i-1} - C_i) + F_i(C_{i+1} - C_i)$$
$$- k_i\Delta V_iC_i + M_i = 0. \tag{3}$$

where k_i is a first-order decay constant for the substance, and M_i the mass rate of addition of the substance to the i-th segment. Although the ebb and flow of the tide is not explicitly represented, the estuary is conceived as a series of segments oscillating upstream and downstream with the tidal velocity, and account can be taken of the fact that an effluent discharged continuously is spread over a distance equal to the tidal excursion during a tidal oscillation. This is achieved by distributing the inputs in suitable proportions among those segments which could be affected by particular outfalls during a tidal cycle. The segments affected can be identified by considering the maximum and minimum volume landwards of an outfall during a tidal cycle.

OXYGEN BALANCE

The distribution of dissolved oxygen is calculated by considering the changes in mass occurring in each segment during a tidal cycle taking into account the hydrolysis of organic nitrogen, oxidation of ammonia and carbonaceous material, reaeration at the water surface, and the reduction of nitrate when oxygen concentration is limiting. Assumptions are:

1. The rates of oxidation of carbon compounds are proportional to their concentration.

2. The organic nitrogen is converted to ammonia at the same rate as the carbon is oxidized.

3. The rate of oxidation of ammonia is proportional to its concentration.

4. Oxidized nitrogen is destroyed at a very slow rate proportional to its concentration.

However, when dissolved oxygen falls below 5 per cent of saturation, assumptions 3 and 4 are replaced by 5 and 6 below.

5. The rate of oxidation of ammonia is initially decreased, and if, despite this, the concentration of oxygen remains below 5 per cent of saturation, the oxidation of ammonia ceases, and oxidized nitrogen is reduced to molecular nitrogen at a rate sufficient to maintain the oxygen content at 5 per cent.

6. The concentration of oxygen is allowed to fall to zero when the reserves of nitrate are exhausted.

The basis for these assumptions, which cannot be justified in detail, is discussed in Ref. 1a.

During the course of the Thames investigations (1b) it was found empirically that the course of oxidation of a wide range of wastes including sewage and sewage effluent, industrial waste waters, and river water, could be represented by a composite exponential derived on the assumption that the organic matter undergoing oxidation consists of two components referred to as 'fast' and 'slow' carbon respectively to distinguish the relative rates of oxidation. The equation used was

$$y = E_c\left[1 - (1 - p)\exp(-kt) - p\exp(-kt/5)\right], \tag{4}$$

where y is the oxygen uptake in time t, k the standard rate-constant of 0.23 d^{-1} at 20°C, p the proportion of the organic material oxidized at the slower rate, and E_c the total oxygen uptake. The value of E_c when multiplied by the rate of discharge of effluent was called the effective carbonaceous load and represents the mass of oxygen that would be taken up if the material remained in the estuary for a sufficient period. If some of the material escapes to the sea before being oxidized, it does not affect the polluting load, as this loss is taken into account in the calculation of the distribution of carbonaceous material. The relationship between E_c and the 5-day BOD at 20°C, B, found by setting t = 5 days in Equation 4, is

$$E_c = B/(0.69 - 0.48\,p). \tag{5}$$

The value of p depends upon the type and nature of the discharge and was determined experimentally for each of the major sources of pollution during the survey of the Thames. The values found for settled sewage (BOD 150 mg/l), sewage effluent (BOD 25 mg/l) and the water entering the estuary from the River Thames at Teddington were 0, ½, and 6/7, respectively.

Since it has been assumed that the hydrolysis of organic nitrogen to ammonia proceeds at the same rate as the oxidation of carbonaceous material, the effective oxygen demand due to organic nitrogen (1c) has been obtained by multiplying the ultimate oxygen demand by E_c/U_c, where U_c is the theoretical carbonaceous ultimate oxygen demand equal to 2.67 C, C being content of organic carbon. The effective oxygen demand due to oxidizable nitrogen is thus taken as

$$E_N = 4.57\left(N_{amm} + \frac{E_c}{U_c}N_{org}\right), \tag{6}$$

where N_{amm} and N_{org} are the contents of ammoniacal and organic nitrogen respectively.

Average values of the polluting load entering the estuary from the four main sewage works are shown for selected periods in Table 1.

TABLE 1. Effective oxygen demand load (tonnes/day) for four major sewage discharges to Thames Estuary

Period	Effective load			
	Carbonaceous	Nitrogeneous		Total
		Ammoniacal	Organic	
1950–53	453	274	89	816
1960–62	331	242	69	642
1969	227	164	36	427
1970	213	164	33	415
1971	187	142	29	358
1972	174	112	26	312
1973	176	93	28	297
1974	153	73	25	251
1975	132	68	22	222

The oxygen balance in each segment is represented by 7 equations: those for 'fast' and 'slow' organic carbon, 'fast' and 'slow' organic nitrogen, ammonia, nitrate, and dissolved oxygen. Setting

$$G_{ij} = -(F_{i-1} + Q_{i-1}) \text{ for } j = i-1$$
$$= F_{i-1} + F_i + Q_i \qquad j = i$$
$$= -F_i \qquad j = i+1$$
$$= 0 \text{ otherwise,}$$

Equation 3 becomes

$$-\Sigma_j G_{ij} C_j - K_i\, \Delta V_i C_i + M_i = 0. \tag{7}$$

The distributions of 'fast' carbon (FC), 'slow' carbon (SC), 'fast' nitrogen (FN), and 'slow' nitrogen (SN) are independent of oxygen concentration and of each other, and are calculated by solving equations of the form of Equation 7 using appropriate rate-constants and polluting loads. The distributions of ammoniacal nitrogen (AM), and oxidized nitrogen (NO) are calculated simultaneously with that of dissolved oxygen (DO).

The equation governing the mass balance of ammoniacal nitrogen in the i-th segment is

$$-\Sigma_j G_{ij}\, \underline{AM}_j - K_{\underline{AM}i}\, \Delta V_i\, \underline{AM}_i + M_{\underline{AM}i} + U_i/4.57$$
$$+ K_{\underline{FN}i}\, \Delta V_i\, \underline{FN}_i + K_{\underline{SN}i}\, \Delta V_i\, \underline{SN}_i = 0, \tag{8}$$

where the underlining in terms such as $\underline{AM}_i$ indicates a single parameter (in this case the concentration of ammoniacal nitrogen in the i-th segment) not the product of two parameters (A and M_i); similarly $K_{\underline{AM}i}$ and $M_{\underline{AM}i}$ are the oxidation rate-constant and rate of input respectively of ammoniacal nitrogen for the i-th segment. The term U allows for restricted oxidation of ammonia when the concentration of oxygen is limiting, the numerical constant being the oxygen consumed in the oxidation of unit mass of ammonia. The last two terms in the equation represent the formation of ammonia from the hydrolysis of 'fast' and 'slow' organic nitrogen.

The equation for oxidized nitrogen is

$$-\Sigma_j G_{ij}\, \underline{NO}_j - K_{\underline{NO}i}\, \Delta V_i\, \underline{NO}_i + M_{\underline{NO}i} - U_i/4.57 - W_i/2.86 + K_{\underline{AM}i}\, \Delta V_i\, AM_i = 0 \tag{9}$$

where the final term represents the formation of oxidized nitrogen by the oxidation of ammonia. The term W_i allows for the reduction of nitrate to satisfy carbonaceous oxygen demand when oxygen is limiting. The 2.86 is a conversion factor and represents the oxygen made available by the reduction of unit mass of nitrate to nitrogen.

The equation for dissolved oxygen is

$$- \sum_j Q_{ij} \underline{DO}_j + f_i \underline{SA}_i (\underline{DS}_i - \underline{DO}_i)$$

$$+ H_{\underline{DO}i} - K_{\underline{PC}i} \Delta V_i \underline{PC}_i - K_{\underline{SC}i} \Delta V_i \underline{SC}_i - 4.57 K_{\underline{AM}i} \Delta V_i \underline{AM}_i$$

$$+ U_i + W_i = 0 \qquad (10)$$

where f_i is the exchange coefficient for oxygen at the air/water interface, $\underline{SA}_i$ is the average surface area of the i-th segment during a tidal cycle, and $\underline{DS}_i$ is the saturation value for dissolved oxygen.

The method of calculation is to assume initially that there will be no restriction on the oxidation of ammonia, and no reduction of nitrate, that is $U = W = 0$. If the minimum calculated concentration of dissolved oxygen is not less than 5 per cent of saturation, the calculated distribution is the solution required. If values less than 5 per cent occur in any segment, the following iterative procedure is used to obtain the required solution.

1. For each value of $\underline{DO}$ less than 5 per cent, $\underline{DO}$ is set to 5 per cent and Equation 10 is solved for U (assuming $W = 0$). This corresponds to the value of U required to raise $\underline{DO}$ to 5 per cent.

2. A value of U_{max} is also found by setting $K_{\underline{AM}} = 0$ in Equation 8. If $U < U_{max}$, this stage is repeated for the next segment.

3. If $U > U_{max}$, U is set equal to U_{max} (corresponding to $K_{\underline{AM}} = 0$) and the rest of the deficit is assumed to be made up by W (the reduction of nitrate to nitrogen).

4. A value of W_{max} is then found (the amount of oxygen made available by reducing all the nitrate) from Equation 9. If $W < W_{max}$ these steps are repeated for the next segment.

5. If $W > W_{max}$, W is set equal in W_{max} (corresponding to $\underline{NO}_i = 0$) and $\underline{DO}$ is no longer forced to 5 per cent but is allowed to find its own level.

An additional term, which was included in the original model (1d) to allow for reduction of sulphate under anaerobic conditions, is no longer required because of the greatly improved condition of the estuary (5,6).

Since each change in one of the variables in Equations 3 to 10 produces changes in all the others, steps 1 to 5 have to be repeated a number of times in an iterative procedure to obtain convergence of the solution. Each complete iteration involves carrying out steps 1 to 5 for each segment, and then solving Equations 3 to 10 again using the values of U and W found.

The rate-constants for the various processes concerned in the oxygen calculations are functions of temperature satisfying an equation of the form

$$K_\theta = K_{15} \left(1 + \frac{a}{100}\right)^{\theta - 15}, \qquad (11)$$

where K_θ is the value of the constant at temperature θ, K_{15} is the value at $15^\circ C$, and a is a temperature coefficient. Values of K_{15} and a are listed in Table 2.

TABLE 2. Values of K_{15} and α

Process	FC	SC	FN	SN	AM	NO
K_{15}	0.183	0.036	0.183	0.036	0.26	0.01
α	5.0	5.0	5.0	5.0	8.8	0

It was concluded on the basis of comparisons between observed and predicted distributions of dissolved oxygen, and of ammoniacal and oxidized nitrogen, for each three-monthly period from 1950 to 1961, that the value of 1.7 per cent per $^{\circ}C$ for the temperature coefficient of the rate-constant of nitrification formerly used was too low, and for subsequent calculations a value of 8.8 per cent per $^{\circ}C$, more consistent with that found for the growth rate of Nitrosomonas and Nitrobacter, has been used(7). The constant, which allows for a loss of nitrate from the water under aerobic conditions - due to its reduction at the sediment/water interface or within suspended particulate matter - has also been changed: a rate of loss of 2 per cent per day was formerly assumed. The rate-constants associated with fast and slow organic carbon and fast and slow organic nitrogen are those formerly used.

OBSERVED AND PREDICTED CONDITION

In Fig. 2 a comparison is made between the observed and predicted distributions of dissolved oxygen and ammoniacal and oxidized nitrogen for each three-monthly period of 1969, 1971, 1973, and 1975 (but excluding nitrogen forms for 1975). The factors considered to change from quarter to quarter are the polluting loads from the larger sewage works, from storm sewage, and from tributaries, the distribution of temperature, and the fresh-water flow. A uniform value of 5.5 cm/h for the exchange-coefficient for oxygen at the air/water interface has been assumed throughout the estuary in each quarter. Observed data are the averages of weekly surveys throughout the estuary. The sampling positions of the individual data (which relate to various tidal states) were reduced to a common tidal state before averaging. The method adopted (1e) was to replace the actual sampling position by the one occupied by the water at half-tide - defined as the instant when the upstream volume is the mean value for the average tidal cycle. Thus low-water samples are, in effect, moved upstream, and high-water samples downstream by about half the tidal excursion.

The agreement is seen to be reasonably good, the general shapes of the calculated curves are similar to the distributions of observed data, the various maxima and minima occurring in the same part of the estuary for both observed and calculated results. Moreover, conditions of fresh-water flow and temperature in first and third quarters differ widely, but the shapes of the predicted curves follow the observed distributions with a fair degree of accuracy. For example, the average fresh-water flow in the first quarter of 1969 was 169 m^3/s and in the third quarter 26 m^3/s, and the corresponding temperatures in the reach extending from 30 to 70 km seawards of the tidal limit at Teddington Weir were 8.2 and 21.8$^{\circ}C$.

There are some systematic discrepancies, the most apparent being the tendency to over-estimate oxygen concentrations in the upper reaches. Each calculated curve assumes a steady fresh-water flow at Teddington; if allowance is made for flow variations within a quarter, the predicted oxygen content is lowered in the upper reaches (1e) - the effect is generally most pronounced in the first and fourth quarters. Discrepancies are also likely to arise from the use of a single value of the exchange coefficient for oxygen, due to the fact that no account has been taken of the effects of photosynthesis by phytoplankton, and to the assumptions concerning rates of oxidation when the oxygen content is low. The approximate nature of these assumptions may also account in part for the severe disagreement - notably in the third quarter of 1971 - found between the observed and predicted distributions of ammoniacal nitrogen. The agreement between the observed and predicted nitrogen forms in 1975 (not shown) is similar to that in 1973. The predicted distributions of oxidized nitrogen are of the correct general shape, but there is some tendency to under-estimate the concentrations in the first, second, and fourth quarters of the year. No doubt some improvement could be effected by relating loss of nitrate to the type and nature of the sediments and by allowing for seasonal variations in temperature.

Despite these limitations, it may be seen that the model, which has the merit of comparative simplicity, has led to realistic predictions of water quality during a period where there has been a progressive reduction in polluting load, (Table 1).

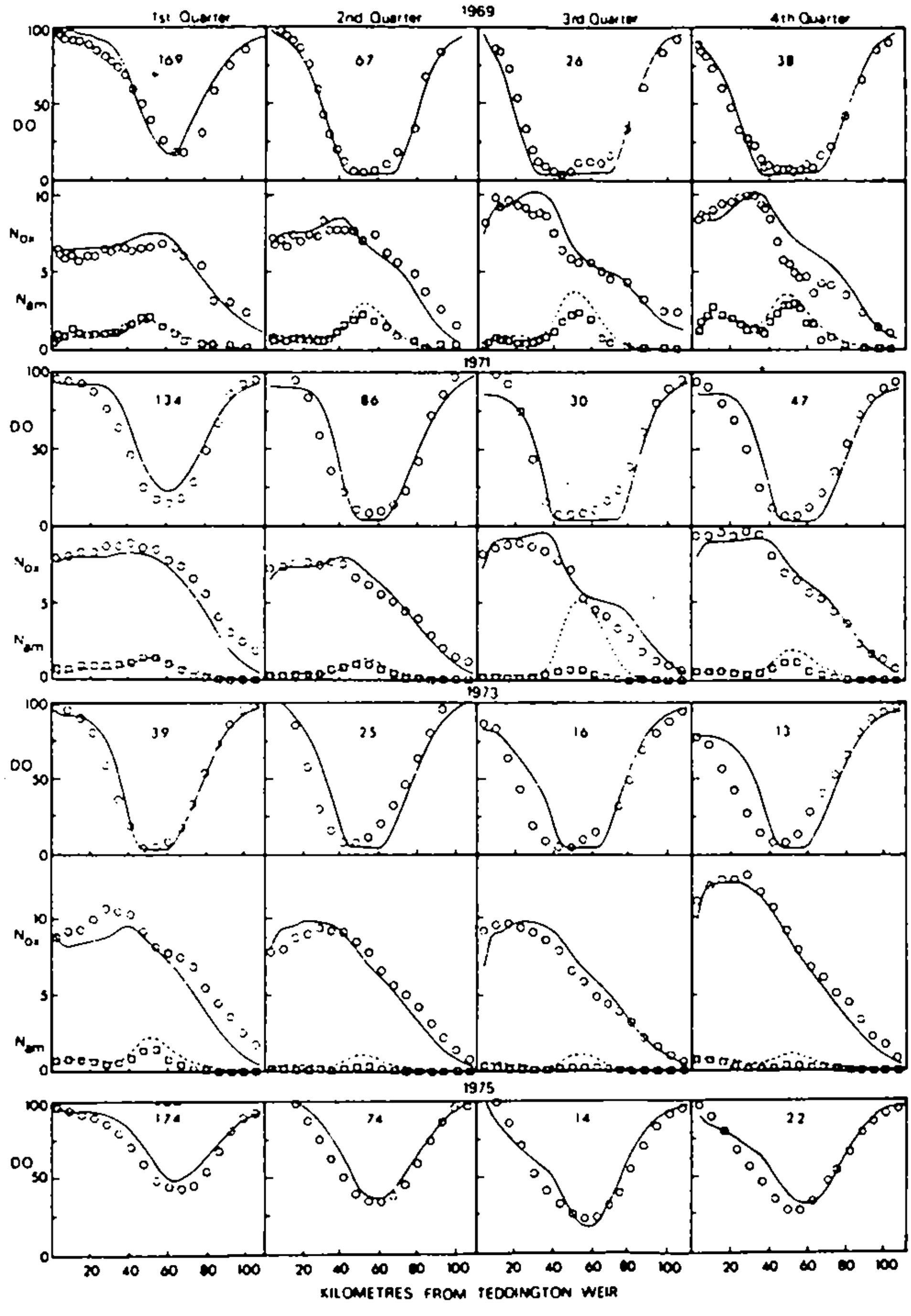

Fig. 2. Observed and calculated quarterly distributions of dissolved oxygen, DO (per cent saturation), ammoniacal nitrogen, N_{am} (mg/l), and oxidized nitrogen, N_{ox} (mg/l).

Encircled points relate to observed distributions of dissolved oxygen and oxidized nitrogen and squares to observed distributions of ammoniacal nitrogen. Continuous curves, calculated distributions of dissolved oxygen and oxidized nitrogen; broken curves, calculated distributions of ammoniacal nitrogen.

Figures are average fresh-water flows (m^3/s) passing Teddington Weir.

Acknowledgements

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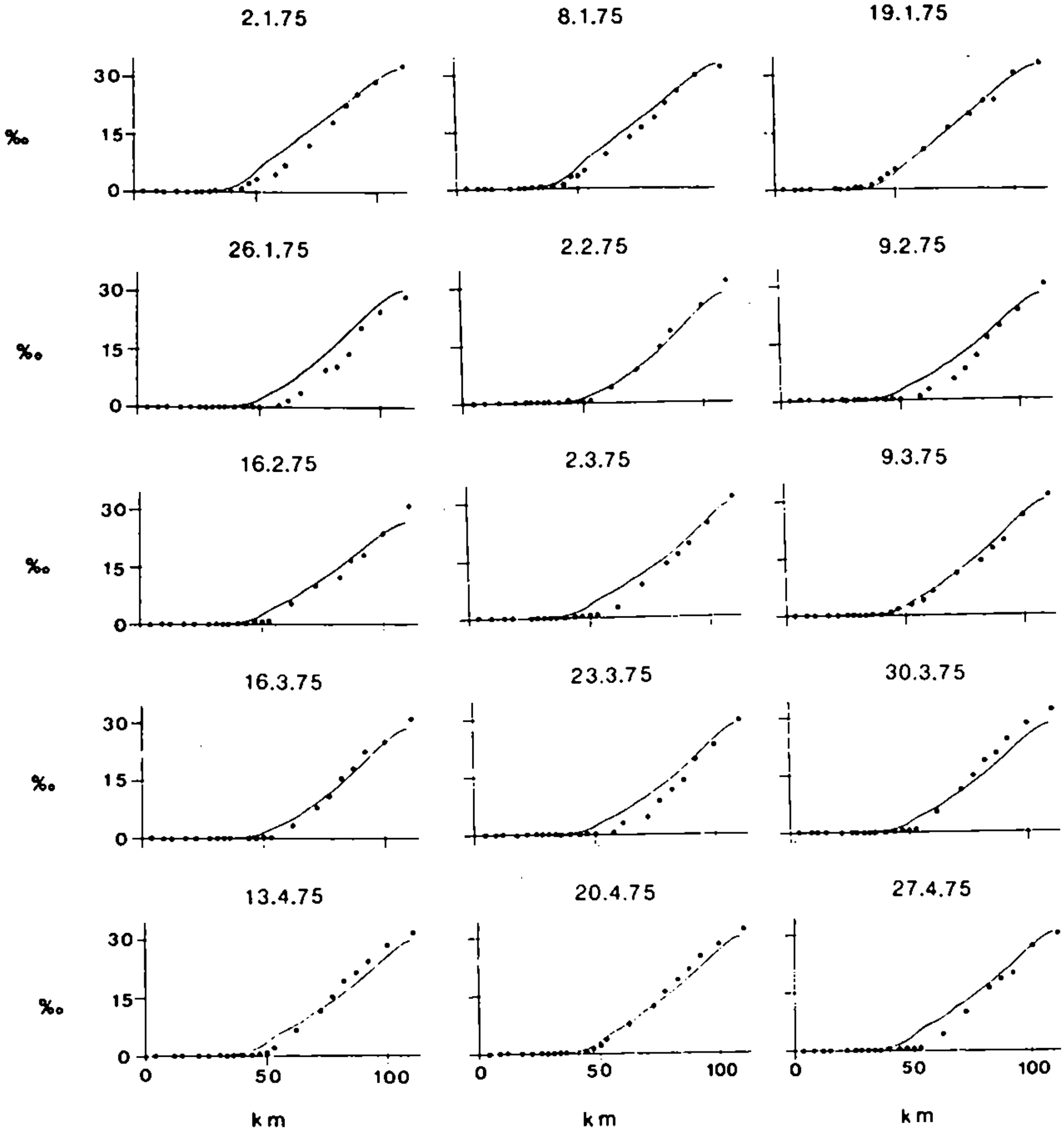
REFERENCES

1. Department of Scientific and Industrial Research. Effects of polluting discharges on the Thames Estuary. Water Pollution Research Technical Paper No. 11, H. M. Stationery Office, London (1964). a, p. 489; b, chap. 6; c, p. 227; d, p. 476; e, p. 485.
2. Preddy, W. S. The mixing and movement of water in the estuary of the Thames. J. mar. biol. Ass. U.K., 33, 645-662 (1954).
3. Water Pollution Research 1966. H. M. Stationery Office, London (1967), pp.24-26.
4. Mollowney, B. M. One-dimensional models of estuarine pollution. In Water Pollution Research Technical Paper No. 13. Mathematical and hydraulic modelling of estuarine pollution. H. M. Stationery Office, London, (1973), pp.71-84.
5. Gameson, A. L. H., Barrett, M. J., and Shewbridge, J. S. The aerobic Thames Estuary. In Proc. 6th Int. Conf. Wat. Pollut. Res., Jerusalem, 1972: Advances in Water Pollution Research, Pergamon Press, Oxford and New York (1973).
6. Gameson, A. L. H. and Alwyne Wheeler. Restoration and recovery of the Thames Estuary. Proc. Int. Symp. Recovery of Damaged Ecosystems, Bladesburg, Va, USA, March 1975. University Press of Virginia, Charlottesville (1977).
7. Knowles, G., Downing, A. L., and Barrett, M. J. Determination of kinetic constants for nitrifying bacteria in mixed culture, with the aid of an electronic computer. J. gen. Microbiol., 38, 263-278 (1965).

APPENDIX 4.2. Validation plots of salinity distributions.

CLASSICAL DROUGHT SEQUENCE 1975/76

SALINITY ‰



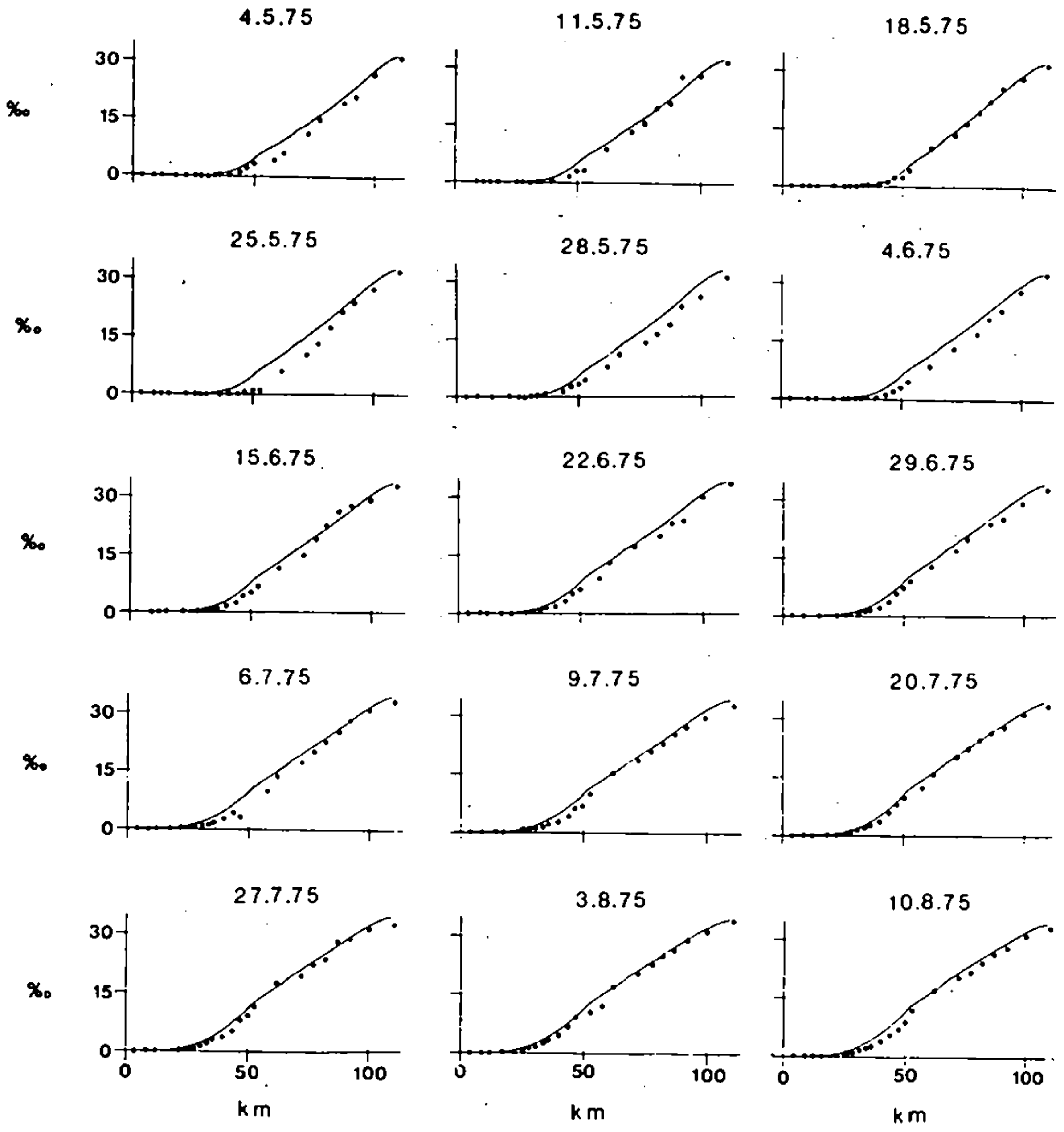
from Teddington

APPENDIX 4.2

Figure 1 Observed (•) vs. Simulated (continuous lines) 'mid-tide' axial salinity distributions (‰) from Teddington (0 km) to Shoebury Ness (111 km) for each sampling occasion in 1975/76.

CLASSICAL DROUGHT SEQUENCE 1975/76

SALINITY ‰



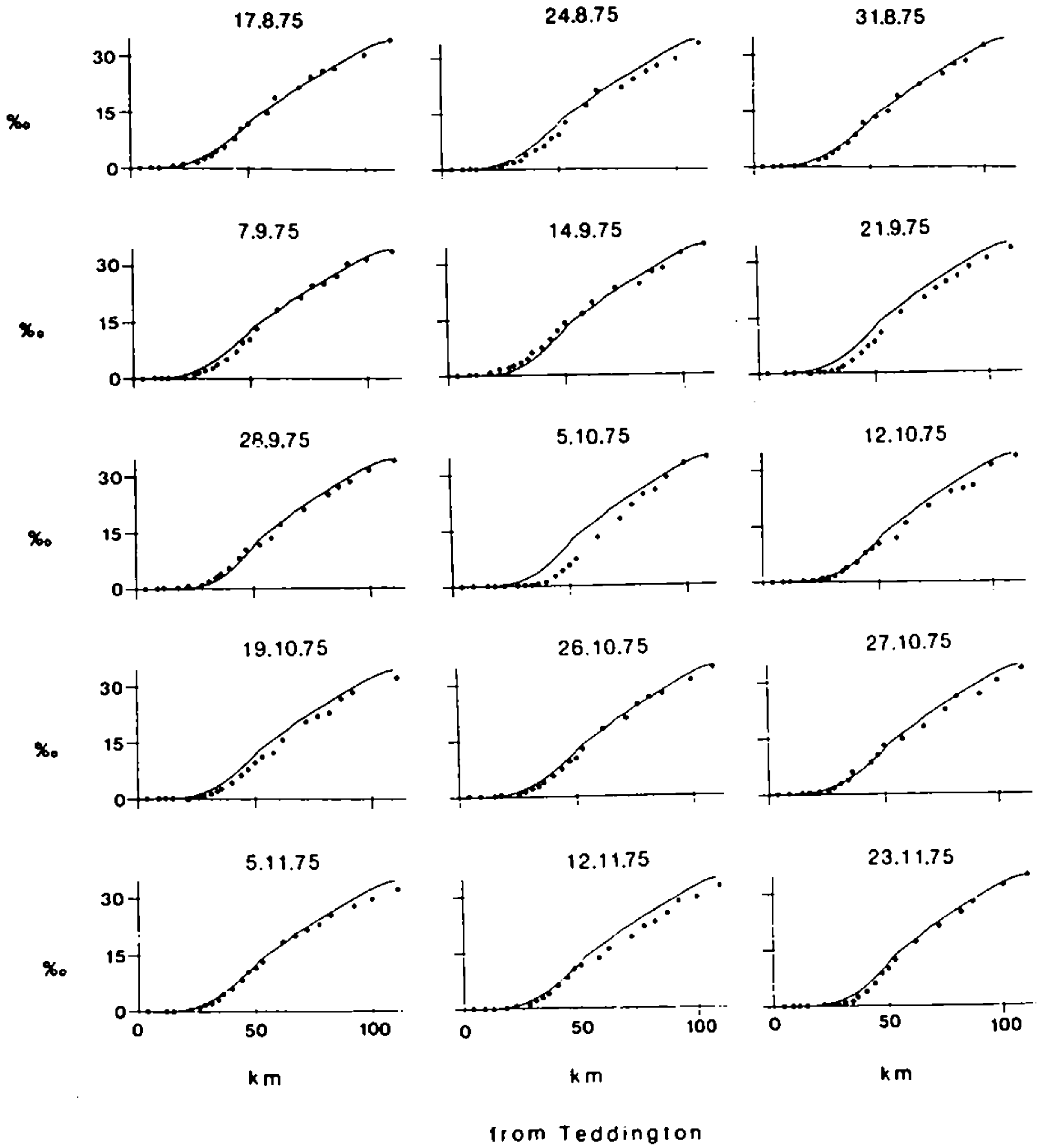
from Teddington

APPENDIX 4.2

Figure 2 Observed (•) vs. Simulated (continuous lines) 'mid-tide' axial salinity distributions (‰) from Teddington (0 km) to Shoebury Ness (111 km) for each sampling occasion in 1975/76.

CLASSICAL DROUGHT SEQUENCE 1975/76

SALINITY ‰

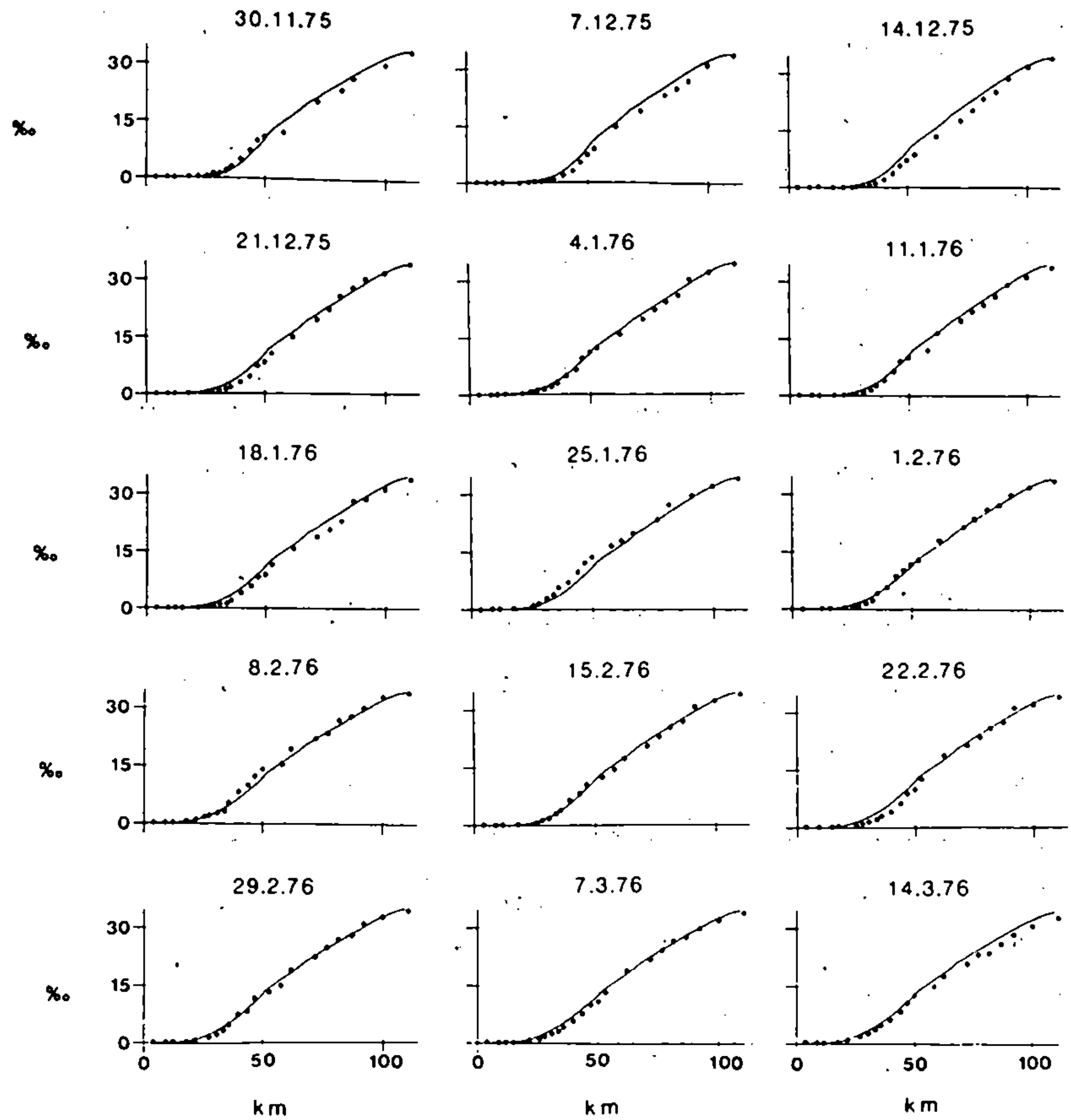


APPENDIX 4.2

Figure 3 Observed (•) vs. Simulated (continuous lines) 'mid-tide' axial salinity distributions (‰) from Teddington (0 km) to Shoebury Ness (111 km) for each sampling occasion in 1975/76.

CLASSICAL DROUGHT SEQUENCE 1975/76

SALINITY ‰



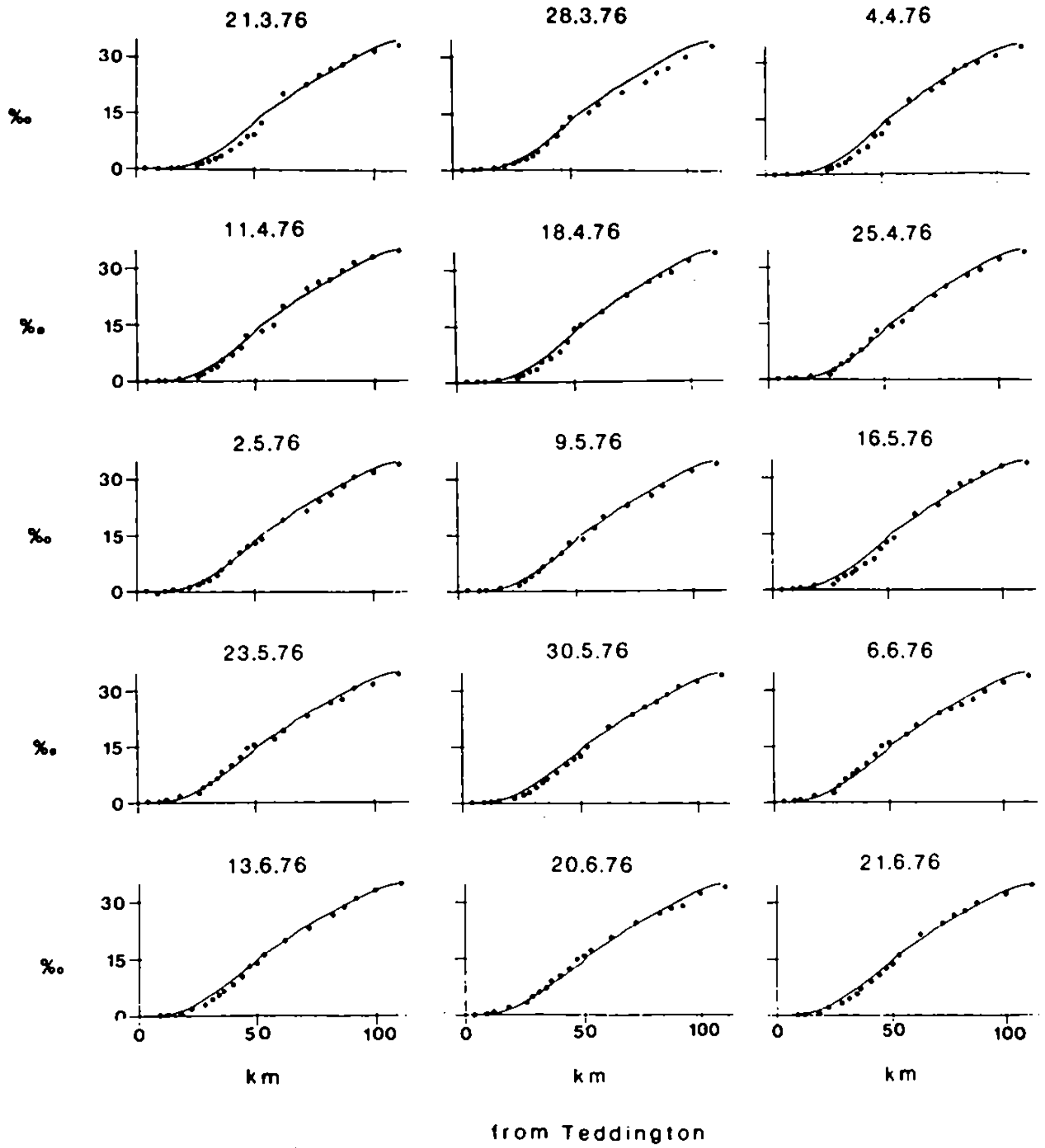
from Teddington

APPENDIX 4.2

Figure 4 Observed (•) vs. Simulated (continuous lines) 'mid-tide' axial salinity distributions (‰) from Teddington (0 km) to Shoebury Ness (111 km) for each sampling occasion in 1975/76.

CLASSICAL DROUGHT SEQUENCE 1975/76

SALINITY ‰

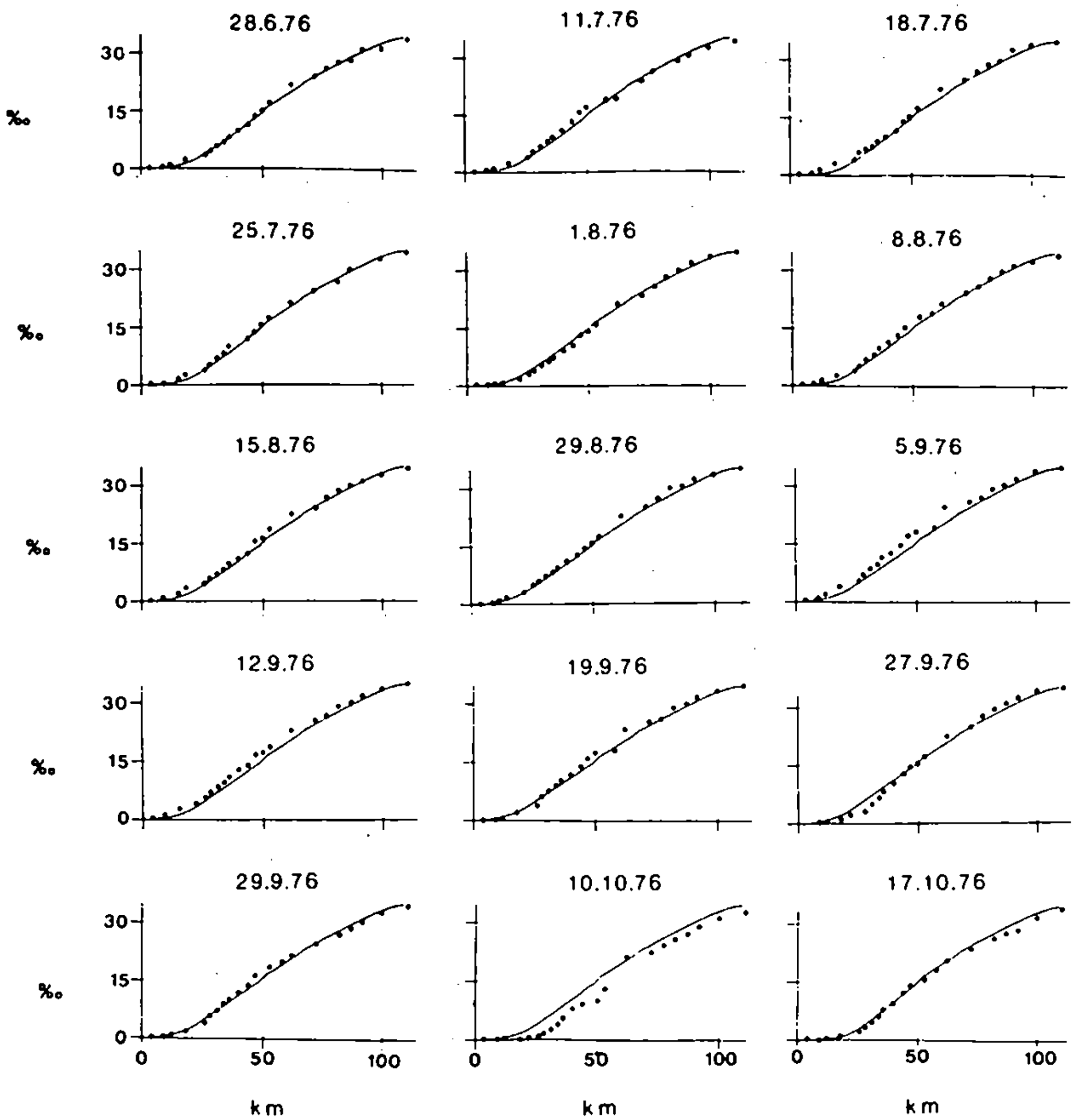


APPENDIX 4.2

Figure 5 Observed (•) vs. Simulated (continuous lines) 'mid-tide' axial salinity distributions (‰) from Teddington (0 km) to Shoebury Ness (111 km) for each sampling occasion in 1975/76.

CLASSICAL DROUGHT SEQUENCE 1975/76

SALINITY ‰



from Teddington

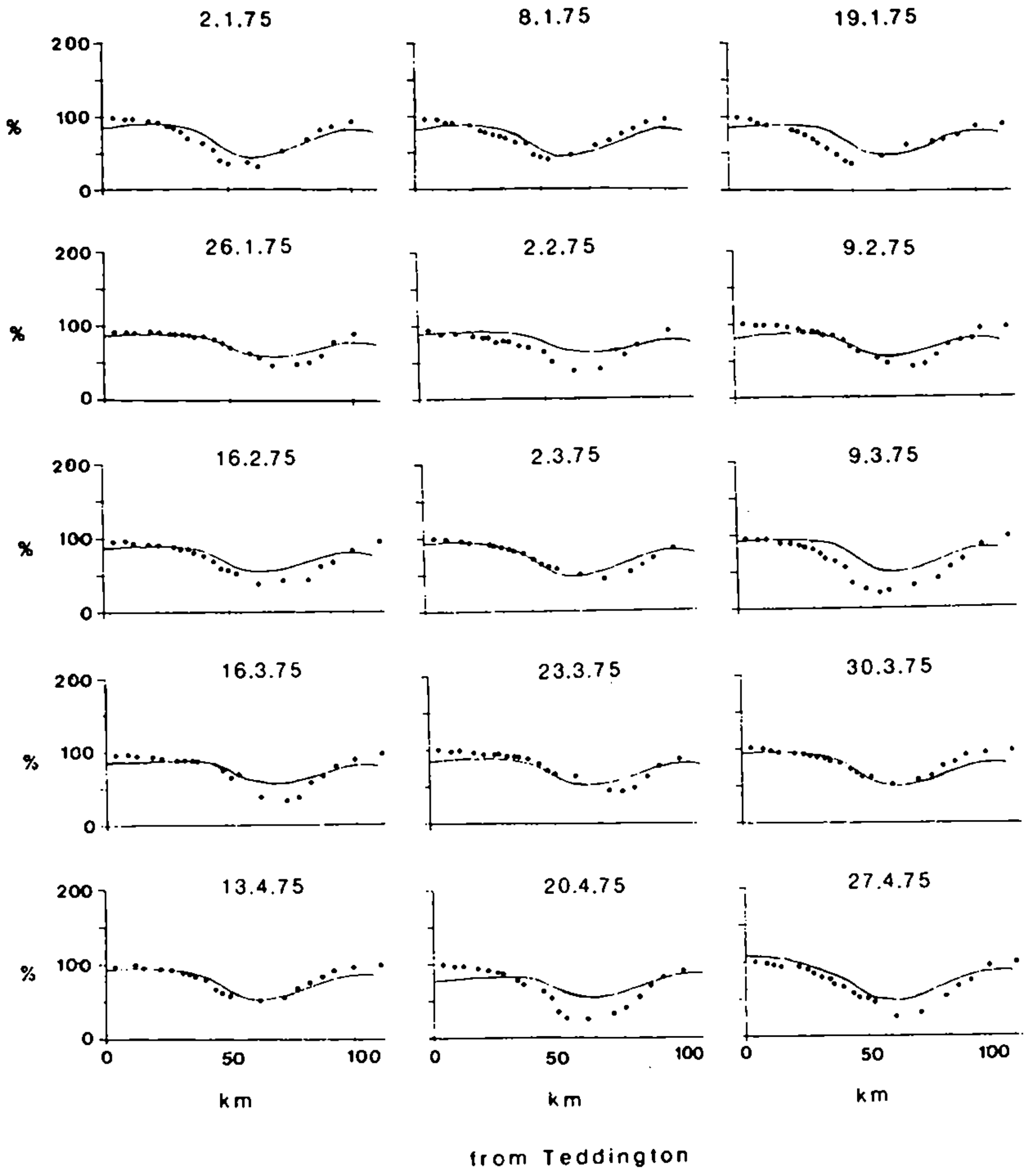
APPENDIX 4.2

Figure 6 Observed (•) vs. Simulated (continuous lines) 'mid-tide' axial salinity distributions (‰) from Teddington (0 km) to Shoebury Ness (111 km) for each sampling occasion in 1975/76.

APPENDIX 4.3. Validation plots of dissolved oxygen.

CLASSICAL DROUGHT SEQUENCE 1975/76

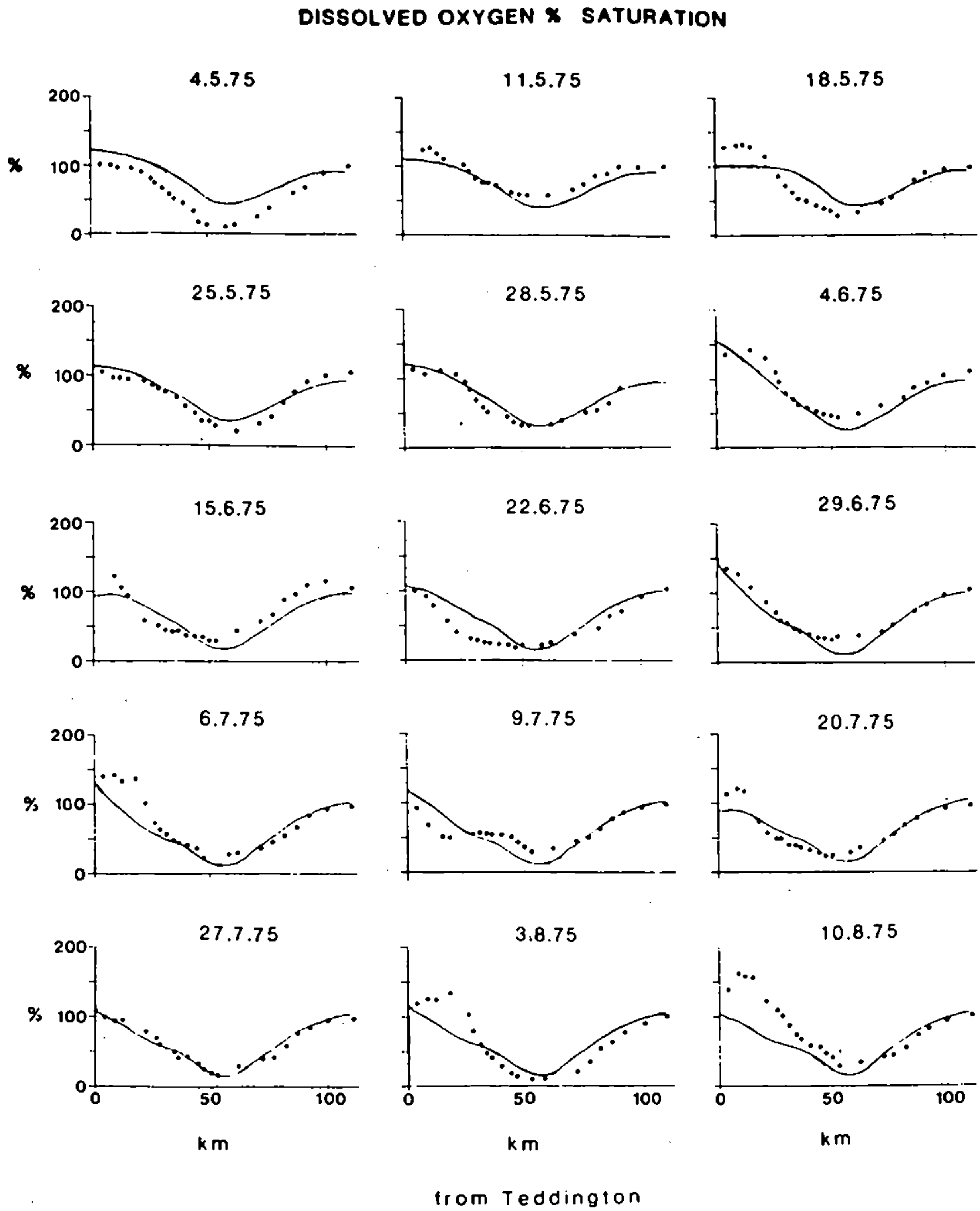
DISSOLVED OXYGEN % SATURATION



APPENDIX 4.3

Figure 1 Observed (•) vs. Simulated (continuous lines) 'mid-tide' axial oxygen distributions (% saturation) from Teddington (0 km) to Shoebury Ness (111 km) for each sampling occasion in 1975/76.

CLASSICAL DROUGHT SEQUENCE 1975/76

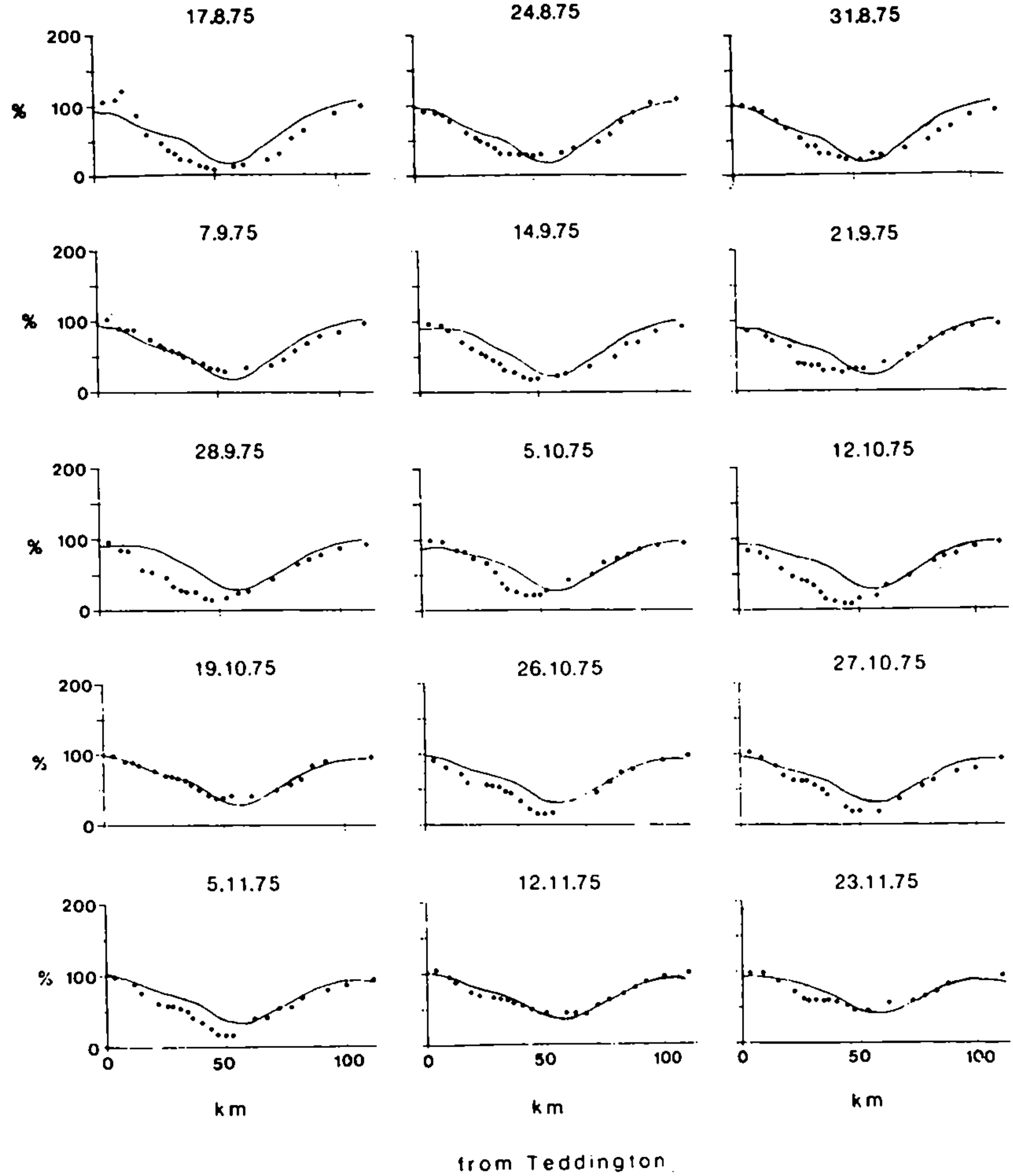


APPENDIX 4.3

Figure 2 Observed (•) vs. Simulated (continuous lines) 'mid-tide' axial oxygen distributions (% saturation) from Teddington (0 km) to Shoebury Ness (111 km) for each sampling occasion in 1975/76.

CLASSICAL DROUGHT SEQUENCE 1975/76

DISSOLVED OXYGEN % SATURATION



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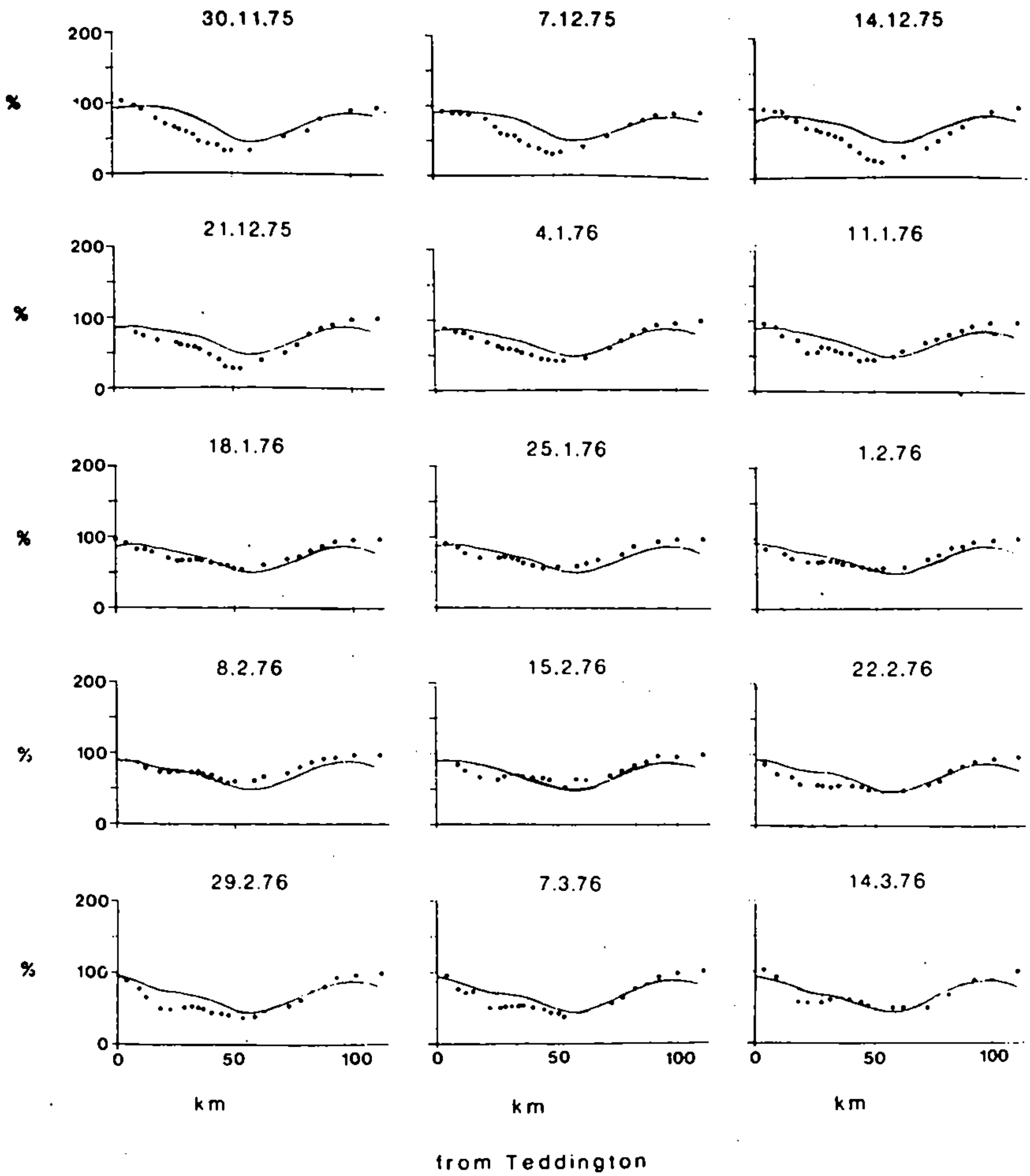
CLASSICAL DROUGHT SEQUENCE 1975/76

DISSOLVED OXYGEN % SATURATION



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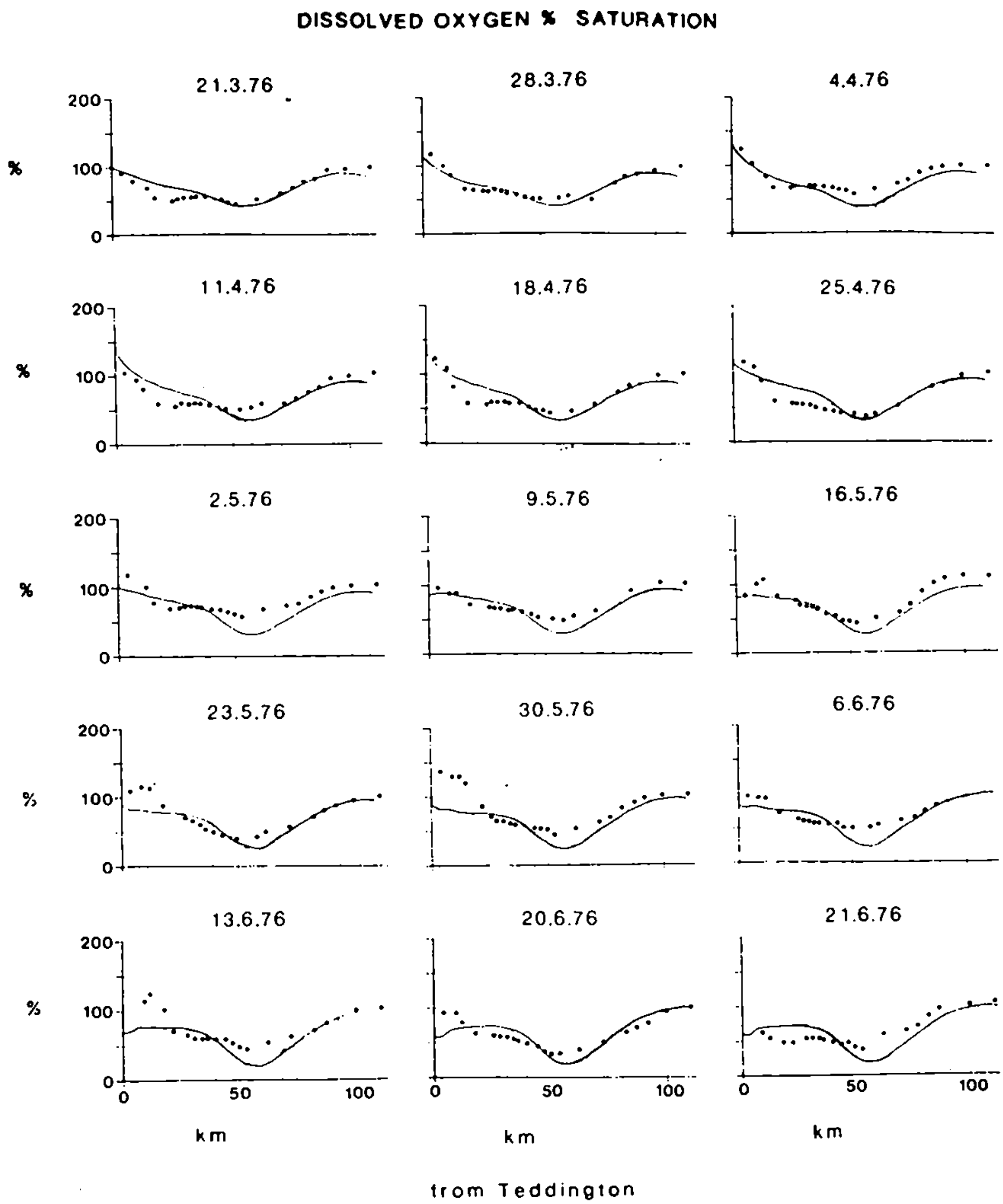
CLASSICAL DROUGHT SEQUENCE 1975/76



APPENDIX 4.3

Figure 5 Observed (•) vs. Simulated (continuous lines) 'mid-tide' axial oxygen distributions (% saturation) from Teddington (0 km) to Shoebury Ness (111 km) for each sampling occasion in 1975/76.

CLASSICAL DROUGHT SEQUENCE 1975/76

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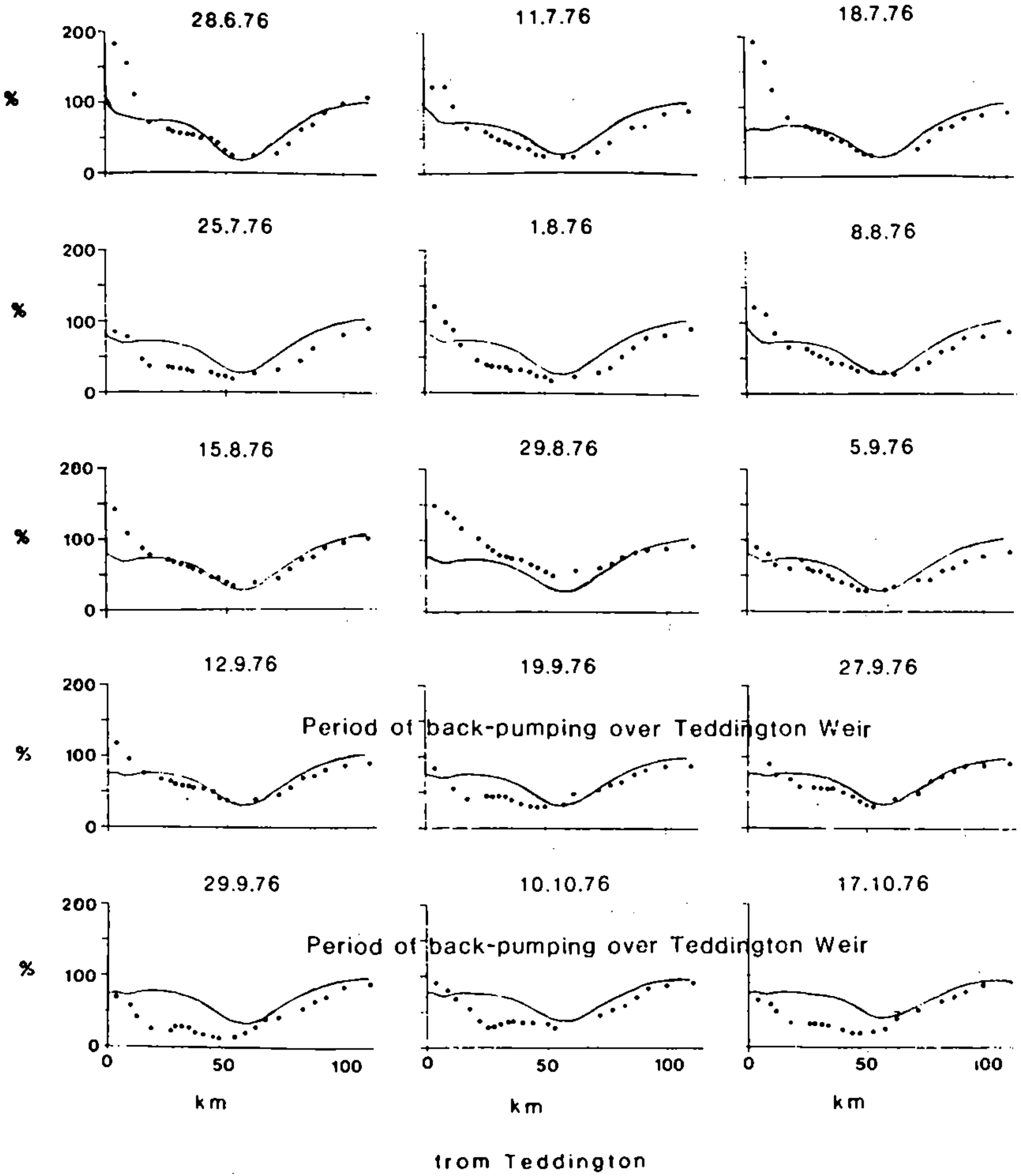

APPENDIX 4.3

Figure 6 Observed (•) vs. Simulated (continuous lines) 'mid-tide' axial oxygen distributions (% saturation) from Teddington (0 km) to Shoebury Ness (111 km) for each sampling occasion in 1975/76.

5. <u>EFFECTS ON THE BIOTA OF THE LOWER THAMES AND THE THAMES ESTUARY</u>

A.D. Berrie

5. EFFECTS ON THE BIOTA OF THE LOWER THAMES AND THE THAMES ESTUARY

5.1. INTRODUCTION

There is a scarcity of good information about the flora and fauna of the lower River Thames. Although over 800 publications have been identified which appear to have relevance to the hydrology, chemistry and biology of the river, few of these have provided useful information for this study. We would like to have known what species are present in the river, the numbers and rates of growth of all the main species, how these vary seasonally and from year to year and how this variation relates to physical and chemical conditions in the river. In the absence of this information it is difficult to make accurate predictions about the likely effects of small changes in environmental conditions.

Living organisms can survive within a range of environmental conditions. Under optimum conditions a species will flourish and under less favourable conditions it may decline because the conditions are more favourable for some other species. The rate and extent of change in the flora and fauna depend on the level of change in environmental conditions and on the period for which the change persists. In some cases the conditions may change far enough to eliminate a particular species. This is most likely to occur when conditions are at an extreme of the range found in the river and it is consequently important to know what extremes may occur now or under the Maximum Resource Benefit (MRB) policy. This is difficult given the nature of the physical and chemical data both about present conditions in the river and about the predicted conditions. Much of the information is based, understandably, on samples taken during normal working hours. When rapid changes are unlikely and where these are not related to the time of day, this may be adequate. However, levels of dissolved oxygen are known to vary on a daily cycle with the lowest values occurring around dawn. The nocturnal levels are the ones that may be critical for animals and little is known about them in the River Thames. Modelling the hydrology and chemistry of the river involves reducing the available data to some manageable form. This usually means that average values are used. These may be calculated on a daily, tidal, weekly or other basis but they all have the effect of concealing the extreme values that may occur for short parts of the period that is averaged. It is these extreme values that may be critical for biological predictions and, in this case, they are not known or predictable.

The models that have been constructed and the biological predictions based on them assume either present levels of demand for water supply purposes from the River Thames or a 17% increase in demand which has been predicted by TWA for the year 2006. If the demand increases beyond this level the predictions could be affected and the views expressed in this report should not be considered applicable.

This chapter has been prepared from contributions made by several scientists with expertise in different aspects of river ecology. Their names are shown beside the relevant headings.

5.2. ECOLOGICAL ASPECTS OF PREDICTED PHYSICAL AND CHEMICAL CHANGES (H. Casey and I.S. Farr)

5.2.1. Discharge and Velocity

The discharge at Teddington could be reduced in certain cases under MRB. Examples shown from the IH model are June/July 1945, August/September 1953, August/September 1975 and July 1957. In many of these cases the flow is reduced substantially under low flow conditions.

Changes in water velocities may influence a river ecosystem in a number of ways. A reduction in velocity and turbulence may allow separation of an anoxic layer in the water column. It may also allow sedimentation of the suspended load and siltation of benthic habitats. Increased velocity may cause scouring of sediment, increase in suspended load, reduction in light penetration and increase in biochemical oxygen demand (BOD).

Reduction in velocity increases retention of pollution in a section and may lead to reduced levels of dissolved oxygen (DO). The IH model suggests little significant change in water velocities at higher discharges following deregulation. However, for a recurrence of the 1952-53 drought sequence, velocities between day 550 and 640 are half those under the present regime for reaches 4 and 5. Even under the existing regulations velocities would be extremely low and the effects of the change in terms of additional sedimentation and reduced turbulence is unlikely to be great. Any effects due to extended retention of effluent are taken into account in the DO/BOD models.

5.2.2. Water quality

The major water quality problems appear to be in the section above Teddington Weir. Values from the period 2.1.74 - 29.12.82 show day 900 with a DO concentration down to c. 3 mg l^{-1} and a very high ammonia concentration of 4.6 mg l^{-1}. Day 1000 in this period shows an ammonium concentration of 5 mg l^{-1}. Later peaks of ammonia are up to 1.6 mg l^{-1}.

The IH model shows reach 5 above Teddington Weir to have consistently the highest mean BOD. The model also shows mean DO levels above or close to saturation for all reaches for all simulations of recurrence of drought sequences under both the Chart and MRB regimes. Only minor changes are indicated with different management. Such mean values are, however, of little use in assessing the impact of water quality on the ecology. Of greater significance are the minimum quality values and, in particular, the lowest DO.

Given a recurrence of the 1944-45 drought sequence the model predicts a substantial reduction in DO in reaches 3, 4 and 5 under deregulation. Similarly, a reduction in DO in reaches 4 and 5 is predicted given a recurrence of the 1975-76 drought. However the major reduction in quality (potentially fatal for most fish and many invertebrates) predicted for the 1944-45 sequence might be discounted following improvements in effluent treatment. The reduction in minimum DO from 7.75 to 6.60 mg O_2 l^{-1} in section 4 and from 6.50 to 5.80 mg O_2 l^{-1} in reach 5, for the recurrence of the 1975-76 drought does not appear to be of major significance to the ecology. In addition the high mean DO values predicted for all reaches indicate the partly empirical nature of some factors in the oxygen demand equations used in the models. There would appear to be a bias towards daytime surface water values of DO which may have been used to derive the factors in the equations. Those would be influenced by phytoplankton production of DO. Substantial oxygen sag is likely during darkness and at depth during low turbulence conditions and these will become more extreme as flows fall to the predicted 2006 levels.

Under the predicted conditions for the year 2006, a large proportion of the total flow of the river at Teddington during periods of low flow will be made up of sewage effluent. Water quality in the river will be strongly influenced by the quantity and quality of these effluents and there could be increases in the already high level of nitrates.

DO values given by the IMER model for the sections below Teddington Weir appear to show no ecologically significant changes in water quality following deregulation of flows over the weir. However the same uncertainties over the effects of increased demand and effluent production and the diel cycle, compounded by uncertainties associated with tidal flows, apply.

5.3. BACTERIAL ECOLOGY (J.H. Baker)

The population size of the suspended bacteria in rivers is largely influenced by prolonged rainy periods and during periods of steady or declining discharge bacterial numbers are more or less constant (Baker & Farr 1977). Therefore no large effect on the size of the suspended bacterial population is anticipated under MRB.

Under the existing system a substantial discharge is maintained over Teddington Weir during the spring and early summer of a drought year, but this discharge is greatly reduced later in the summer. Thus the residence time of the water above Teddington is substantially greater during late summer compared with earlier in the year. There are both advantages and disadvantages to increased residence time of river water. The disadvantages are mainly associated with the reduced ability to flush out noxious materials which are both harmful to the natural river biology and expensive to remove from drinking water supplies. The quality of the water both above and below the weir has been greatly improved since the Statutorily Protected Flow was introduced (Doxat 1977). Nevertheless it is still true to say in general terms that the more water that passes down the Thames the cleaner it will be.

The advantages of increased retention are associated with self-purification. Gannon et al. (1983) showed that faecal coliform bacteria disappeared from a river impoundment in Michigan, USA. Sedimentation was identified as the most likely cause. A similar phenomenon probably occurs when discharge is reduced over Teddington Weir. Although this report is not particularly concerned with faecal coliform organisms they may be assumed to indicate the behaviour of other non-indigenous bacteria. Another aspect of self-purification is the degradation of pollutants by bacteria. So long as the pollutant remains in the water column and oxygen is not limiting, degradation by bacteria of any carbonaceous material will continue and the degree of degradation will be proportional to the residence time.

With regard to benthic bacterial processes, increasing discharge will increase the rate at which microbial products are removed from the sediment surface. However, increased discharge also means a decrease in sedimentation rate and thus the supply of substrates to the sediment is decreased. Of particular importance is the denitrification process during which bacteria reduce nitrate. Because the nitrate is in dissolved rather than particulate form its supply to the benthic deposits is dependent on diffusion and turbulence rather than sedimentary processes. Nevertheless denitrification is an anaerobic process and is more likely to be controlled by the degree of aeration of the sediment than by the rate of substrate supply. Since aeration is inversely proportional to particle size, which in turn is directly related to current velocity, denitrification is likely to be correlated with decreasing discharge.

The MRB regime would probably result in the discharge being more constant over Teddington Weir during a drought with no sudden extended reduction. Although any of the processes discussed above may be pushed slightly one way or the other, the overall effect on the bacteriology of the River Thames is likely to be negligible.

5.4. ATTACHED ALGAE (A.F.H. Marker)

There is a total absence of quantitative data on the attached algae of the River Thames and very few studies of any description exist. The most comprehensive study was carried out between 1980 and 1983 by Dr. D. John of the British Museum (Natural History) who has kindly made available in draft form the results of his qualitative survey.

The Thames is nutrient rich and capable of supporting luxuriant growths of benthic algae to the extent that they could become a nuisance, be unsightly and detract from the amenities available to the public. The reasons why this has not occurred in the lower Thames, so far, is fundamental in predicting changes that could take place in the future.

The chemical data provided by TWA, their data on the phytoplankton of reservoirs and the analysis of Thames phytoplankton in this report confirm the eutrophic nature of the lower Thames. The upper reaches of the Thames also appear to be highly eutrophic; in the non-navigable, but permanently flowing reaches above Lechlade, Ranunculus becomes entangled with Enteromorpha and Cladophora in the summer before being removed by floods in the autumn (John, pers. comm.). The apparent association of Cladophora with nutrient-rich conditions has been noted a large number of times (Pitcairne & Hawkes 1973; Bolas & Lund 1974; Wong & Clark 1976). Between Lechlade and Teddington the Thames is navigable and dredged to a depth of at least 2 metres. Dr. John quotes the bed of these reaches as composed of unconsolidated mud and sand with "algae rarely found growing at depths greater than 0.5m" Berrie (1972a) found attached algae present over the whole bed of the river at Reading but substantial growths were only found in shallow areas (Rhizoclonium). In this shallow water gross primary production exceeded 6g O_2 m^{-2} day^{-1} with a maximum in July of over 12g O_2 m^{-2} day^{-1} . Most of the bed of the stream is deeper, covered in flints and chalk lumps with only small populations of attached algae.

Further downstream, where the Thames is deeper and more canalized, marginal growths cannot occur but horizontal bands of macroalgae occur to a depth of 0.5m on the piles and other permanent fixtures.

There are several factors which probably interact in controlling the growth of attached algae in the lower Thames.

a) The river is very turbid. Calculations provided for the phytoplankton in this report confirm that light is rapidly attenuated and that below 1m irradiance will be the critical factor limiting algal growth.

Cladophora growth is believed to be favoured by high light intensities (reviewed by Whitton 1970). The lower Thames carries a sizeable phytoplankton and falls into that category of river where phytoplankton predominates over periphyton (Vannote et al. 1980). The river continuum concept predicts a switch from periphyton to phytophyton in approximately 7th order streams. However, the lower Thames is probably a 5th order stream, although conservatively estimated as 4th order by Smith & Lyle (1979). There is no inherent contradiction here because river systems in southern England have relatively few tributaries.

b) Boat traffic provides a continuous disturbance to the river bed, hindering the anchoring of algae. Continuous resuspension of silt and sand will decrease light penetration.

Any increase in summer flows should be beneficial. Reductions in summer flows (even if this is not the current intention) could lead to a number of problems. Water velocities are already low during dry periods (< 10 cm s^{-1}). Further reductions could lead to increased light penetration due to sedimentation of suspended solids and hence to an increase in filamentous algae in the river, unless there is an equivalent increase in phytoplankton. Whitehead & Hornberger (1982) refer to Microcystis in the Thames during the drought of 1976, encouraged perhaps by the prolonged low flows. John (pers.comm.) refers to floating growths of Enteromorpha in the backwaters. Reductions in flow during long, dry summers will encourage the growth of Enteromorpha, possibly leading to the invasion of the main river.

In the tidal part of the Thames the change in statutory control will have little effect on the salinity. There are unlikely to be more than minor changes to the distribution of freshwater and saline species reported by Tittley & Price (1977).

5.5. PHYTOPLANKTON (C.S. Reynolds)

5.5.1. Introduction

This section seeks (a) to establish largely by reference to the available literature, the nature and seasonal distribution of phytoplankton populations in the lower Thames, (b) to offer some account of the impact of environmental factors in generating these fluctuations and (c) to make some prognoses about the responses of the phytoplankton to reduced spring-time fluvial flows, based upon the preceding assessment. It is necessary to emphasize at the outset that these prognoses assume that there is no upstream alteration of natural discharges into the lower Thames, as such alteration could well affect the application of the assessment.

5.5.2. The present position

5.5.2.1. General features of river phytoplankton populations

The phytoplankton of rivers has apparently attracted relatively less interest of researchers than has that of lakes and reservoirs or of the seas. Nevertheless, the many qualitative descriptions available (e.g. Allen 1913; Williams & Scott 1962; Greenberg 1964; Swale 1969; Hynes 1970; Lack 1971; Houghton 1972; Whitton 1975) enable some general conclusions to be drawn :

(i) That river-borne algae comprise three elements :

a) Benthic algae, derived from epilithic or epiphytic populations;
b) lake phytoplankton, derived from inflows draining standing waters or from impoundments along the water course itself;
c) a true river phytoplankton (or potamoplankton), not always conspicuously different in composition from (b), that develops within the flowing water itself.

For some time, the existence of (c) was disputed but quantitative studies (see later) attest to prolific *in situ* growth. The relative proportions of these elements depends upon the length and flow rate. Mean velocities of less than 0.5 m s-1 appear to be critical to the successful development of (c), whether or not the inocula are supplied principally as (b) (Gessner 1955).

(ii) That true plankton (c) typically increases absolutely and relative to (a) downstream.

(iii) That this true plankton is characteristically dominated by diatoms, especially by centric species of *Stephanodiscus*, *Cyclotella*, *Melosira*, less commonly by pennate species of *Asterionella*, *Nitzschia*, Chlorococcales are also well represented (*Chlorella*, *Ankistrodesmus*, *Scenedesmum*, a.o.), as are Volvocales *Chlamydomonas*. *Gonium*, *Pandorina*); Cryptophyceae and Chrysophyceae are less common; cyanobacteria, such as *Anabaena*, *Aphanizomenon* and *Microcystis*, rarely grow well in temperate rivers, except in very sluggish situations, though they form an important component in long, tropical or sub-tropical rivers. According to the treatments of Reynolds (1984 a,b), the plankton flora of rivers is characterised by invasive ("reselected") species. suggesting an unstable immature biotype, in a permanent state of colonization.

(iv) That true phytoplankton populations in temperate latitudes fluctuate in time: generally, biomasses are low in winter, increasing through summer, often in two main pulses (viz. April-May, August-September, in the northern hemisphere). The earlier of these is typically overwhelmingly dominated by diatoms. They are also often prominent in the later pulse, though other groups (especially Chlorophyceae) are relatively more abundant and even dominant at that time.

5.5.2.2. Phytoplankton in the Thames

Documentation of the phytoplankton of the Thames, which is probably as extensive as for any other U.K. river, both historically (Hassall 1850; Fritsch 1902, 1903; Rice 1938) and in the number of more recent contributions (which include: McGill 1969; Lack 1969, 1971; Lack & Berrie 1975; Lack, Youngman & Collingwood 1978; Evans & McGill 1970; Evans 1971; Bowles & Quennell 1971; Kowalczewski & Lack 1971; Berrie 1972 a,b) may be reviewed against this general background. The following points may be emphasized.

(i) That, to judge from the studies of McGill (1969), Bowles & Quennell (1971), Lack (1971) and Lack *et al*. (1978), the Thames plankton is dominated through much of the year and along much of its length (Oxford to Walton-on-Thames) by the centric diatom, *Stephanodiscus hantzschii* Grunow. Together with other centric diatom species (*Cyclotella*, *Melosira*), peak populations in spring and late summer may account for 90% of the algal cells present at Reading (Lack 1971) and Walton (McGill 1969). Pennate diatoms (*Nitzschia*, *Synedra* spp.) are, at the present time, only rarely dominant, prior to or immediately following the *Stephanodiscus* episodes, though they share a similar seasonality of abundance. There has been an apparent change over the last 80 years or so, in that Fritsch (1902, 1903) recorded greater relative frequencies of *Asterionella*, *Fragilaria* and *Synedra*, though *Melosira varians* dominated in May; differences in methodology could account for the differences in the position of the collections, though Rice (1938) records alternation between *S. hantzschii* and *M. varians* dominance. The change may well be real, therefore, and could be explained in terms of increasing nutrient loadings on the river.

Between the spring and summer biomass peaks, Chlorophyceae increase to constitute a large proportion of the midsummer biomass minimum: *Ankistrodesmus* and *Scenedesmus* are the most common genera represented, with lesser concentration of *Chlamydomonas*, *Gonium*, *Pandorina*, *Pediastrum* and *Dictyosphaerium* spp.

Other algal groups are frequently represented in back ground numbers that, apart from the Cryptophyceae, show little sustained increase (Lack 1971). However, Whitehead & Hornberger (in press) make reference to a Microcystis 'bloom', confined to the lower Thames (Staines-Teddington), that developed during the low-flow, high-temperature summer period of 1976.

(ii) Longitudinal differences in the means and maxima of the phytoplankton standing crop, presented variously by Bowles & Quennell (1971) and Lack et al. (1978) as total particulate volume (TPV) chlorophyll concentration and cell counts, show the expected increases in magnitude downstream in the classic manner (cf. Greenberg (1964). Peaks at Medmenham cf. Staines or Walton in Spring, 1969, for instance, in TPV (8 cf 14 μl^{-1}), chlorophyll-a (74 cf 80 $\mu g\ l^{-1}$) and S. hantzschii cells (19.9 x 10 cf 29.7 x 10^6 cells l^{-1}) were recorded by Bowles & Quennell (1971). Lack et al. (1978) showed simultaneous chlorophyll-a maxima for 6 stations between Reading and Medmenham from 70 to nearly 100 $\mu g\ l^{-1}$. Incidentally these data superficially suggest that the chlorophyll content of S. hantzschii is equivalent to some 3-4 $\mu g\ (10^6\ \text{cells})^{-1}$, though not all the chlorophyll is necessarily in the diatoms, nor in algae at all; a fraction may be present in the suspended organic matter which, if not separately analyzed, elevates the apparent content of healthy cells. Lack et al. (1978) comment that in some of their estimates, the chlorophyll content was equivalent to only 1.1 $\mu g\ (10^6\ \text{cells})^{-1}$, that is, within the range quoted by Reynolds (1984a) for lake populations. Youngman (in Bowles & Quennell 1971) referred to WRA-data that at high discharges, recorded chlorophyll concentrations could be as high as 60 $\mu g\ l^{-1}$ without any significant phytoplankton being present. Regressing the chlorophyll content against the corresponding diatom populations for the period April-May 1969, presented in Bowles & Quennell, gives intercept values (≡ 'background' chlorophyll) of 17 $\mu g\ l^{-1}$ at Medmenham and 29 at Walton, with slopes of 2.9 and 4.6 $\mu g\ (10^6\ \text{cells})\ l^{-1}$ respectively. These data are relevant to the consideration of the factors determining maximal diatom populations, in the river (see 5.5.2.3.).

(iii) In spite of the expected downstream increases in biomass, none of the data presented by Lack et al. (1974) suggest that maxima pass downstream. Several peaks in the April-May period occurred simultaneously at all six stations (spanning 21 km of river); their collapses did so too. The authors anticipated that the 2-3 days required for river water to traverse this distance was sufficient for further division (direct increase) to perceptibly lag the peaks. These data parallel those of Greenberg (1964) in suggesting that one of the principal tenets about the effect of flow on the ecology of plankton in rivers - that populations are moved downstream - is a misconception.

5.5.2.3. The effect of Flow

Net increase in phytoplankton is dependent upon the simultaneous satisfaction of their energy and nutrient requirements to sustain growth to the extent that it exceeds all sources of loss (respiration, exploitation, death and outwash). In rivers, the dynamic balance is closely linked to the often overriding effects of the flow.

Where reasonable data sets exist, it is apparent that there is a critical level of discharge above which phytoplankton concentration fails to increase or is actively reduced (Butcher 1924; Butcher, Longwell & Pentelow 1937; Swale 1964, 1969). In the Thames, approximate critical discharges are 40 $m^3 s^{-1}$ at Reading (Lack 1971), ~ 50 $m^3 s^{-1}$ at Medmenham (Lack et al. 1978) and nearer 70 $m^3 s^{-1}$ at Walton-on-Thames (from data presented by Bowles & Quennell 1971). The generalized implication is that when critical discharge is exceeded, then plankton maxima are swept downstream. Yet the simultaneity of bloom and collapse events along vast lengths of river, referred to in 5.5.2.2.(iii) suggest that the relationship is more complex. That the relationship is central to the prognoses about effects of altered discharges in the lower Thames, demands further analysis of the complexities.

Three interacting components can be readily identified: the hydraulic effects of fluctuating discharge *per se*; the impact on velocities; and the impact upon turbidity.

(a) Discharge effects. Increased discharge is brought about by the (usually) abrupt augmentation of flow by additional, largely plankton-free sources of water, which dilute the existing plankton suspension. In reality, the total number of organisms present need not be altered. Lack (1971) has provided a graphic illustration of this effect. In May, 1967 he observed a peak of 26.5×10^9 cells m^{-3} of *S. hantzschii* at a station near Reading, when the discharge was 12.7 $m^3 s^{-1}$. One week later, discharge had increased to 85 $m^3 s^{-1}$ and the population had fallen to 3.6×10^9 cells m^{-3}. In both instances, the discharge of diatoms (337×10^9, 306×10^9 s^{-1}) is similar. Put another way, the factors for dilution and increased discharge are almost identical. Moreover, the effect would be contemporaneously similar along much of the downstream reaches.

(b) Velocity is a function of discharge on mean cross-sectional area. It is well known that velocity varies, independently of discharge, with channel section to give the familiar pool-riffle alternation. Apart from detailed cross-sections illustrated in Berrie (1972a) and to measurements referred to by Lack (1971) and by Kowalczewki & Lack (1971), I have found little information on which to formulate any independent velocity/discharge relation for specific reaches of the Thames, though several authors have given mean velocity figures. These latter take the form of river distance travelled per day or more, which is a helpful measure in assessing population 'recovery' downstream, in the sense of distance traversed per cell doubling. Velocity does not increase in direct proportion to discharge in rivers - increased discharge is part compensated by an increase in level and, in extremes, in width (flooding), to give an increased cross-sectional area. Data of the former Thames Conservancy presented by Bowles & Quennell (1971) show how traverse times from a selection of upstream points are altered to sustain given discharges at Teddington. Thus, 36h is theoretically required to travel 128 km from Oxford to Walton to maintain a discharge of 263 $m^3 s^{-1}$ (mean velocity 0.99 $m s^{-1}$); at 52.5 $m^3 s^{-1}$, 96h is required (0.37m s^{-1}); at 13 $m^3 s^{-1}$, the requirement is 168h, (0.21 $m s^{-1}$). The equivalent travel-times in the lower Thames (Windsor-Walton, 32 km) are 9, 24 and 42h respectively. Thus, a 20-fold variation in discharge is sustained by a 5-fold variation in mean velocity. Equally, for 2-fold variations in discharge between Reading and Medmenham (21 km) Lack et al. (1978) estimated only a 1.5-fold variation in traverse time (equivalent to 2-3 days, or 0.08-0.12 $m s^{-1}$). Incidentally, their data imply a corresponding variation in mean cross-sectional area, $\sim$375 to $\sim$500 m^2, which agrees tolerably with the area represented in the section of Berrie (1972a).

These traverse times must be set against the net doubling-times of river algae (see later). Were the doubling time 24h, for instance, the algal population could be expected to increase between Oxford and Walton by a factor of 1.6 at the highest discharge quoted, but by 127 times at the lowest. Between Windsor and Walton, the factors would be 1.3 and 5.8, respectively. So long as growth is not limited by any other factor there is no theoretical bar to population increase along the river length. However, the distance traversed per population doubling will determine how much growth can take place in a river for a given discharge; given uniform flow rates within the river, its absolute length should critically determine the velocity wherein downstream growth might become significant. The data of Bowles & Quennell (1971) suggest that the spring maximum was not initiated before mean velocity fell below 0.3 $m s^{-1}$, whereas high algal abundance coincides with velocities of <0.1 $m s^{-1}$ (Youngman, in the same paper).

Direct estimates of transit times, however, often underestimate the true passage times. There is therefore an additional effect of mean velocity on phytoplankton that can be derived on theoretical grounds, but for which few concrete data exist. This effect is due to the nature of river channels and to frictional resistance to acceleration. Across any given section of river there exists a wide spectrum of instantaneous velocities. Away from the banks and bottom, the water flows relatively rapidly but is slowed down in contact with solid surfaces, including weed growth. In an unpublished report, D.F. Westlake showed how a combination of flow distributions might affect the exponential growth of algae in a section of a river. He considered a theoretical cross section in which half the water traversed at 20 km d^{-1} (0.23 $m s^{-1}$) and half at 10 km d^{-1} (0.12 $m s^{-1}$). Over a distance of 300 km with a doubling time of 0.5 d^{-1}

algae in the fast water would undergo 7.5 divisions (181-fold increase) but 15 (32,800-fold increase) in the slower water. If the waters were fully integrated after 300 km, the population increase in the whole would be equivalent to 16 470 times. Yet if the same discharge were treated as a single mean (15 km d^{-1}) only 10 divisions (1020-fold increase) could be accommodated. In the Thames, the presence of locks and weirs might further enhance this effect. It is possible then, to conceptualize a river as having a dominant flow pattern but, superimposed upon it, a "reservoir" of pools, eddies, bays, backwaters from which the main flow is 'reseeded' continuously (cf. Warwick 1964). The 'reservoir' would alter in size and capacity with discharge: high discharges would mean proportionately more of the river operates as a rapid-flow "piston"; low discharges would permit regrowth in the revitalized reservoirs. Applied to the Thames, this model cannot be evaluated but must serve to enhance the effects of discharge upon instantaneous velocity distribution and, thus, upon phytoplankton growth and depletion.

(c) Turbidity. Variations in discharge are generally accompanied by increases in the suspended load, which includes catchment-derived silts, clay particles and organic matter, autochthonously derived vegetal matter, non-planktonic algae washed off stones and plants and benthic growths, suspended by the current, besides the potamoplankton load. These effects may be attributed to the increases in erosional scour and in carrying capacity ('competence') associated with accelerating velocity and to the additional material-loads washed into the river by the surface- and through-flows that feed the higher discharges. The presence of these materials potentially increases the vertical attenuation of light through river waters, rendering proportionately less of the volume capable of supporting net planktonic photosynthesis and net growth. That some river waters are distinctly coloured may contribute measurably to the same effect. Since it is, superficially, a velocity-related function, it is difficult to separate the effect of increased turbidity *per se* from other depressive effects of increased discharge. Yet they may well be significant, as Williams (1964) has shown: he attributed winter growth of planktonic diatoms in North American rivers, despite near-freezing temperatures and sustained flows, to the fact that little silt was washed in when the ground was frozen.

I am unaware of any published measurements of light penetration in the River Thames that would permit evaluation of the supposed discharge - turbidity relationship, but the primary production data presented in Kowalczewski & Lack (1971; see also Lack & Berrie 1976) derived from photosynthetic measurements in bottles suspended at different depths in the Thames near Reading are instructive in this respect. The original study included photosynthetic measurements, chlorophyll content and cell counts (the latter are to be found in Lack 1971) and there are measurements of discharge available throughout the study period (also in Lack 1971). It should be possible to derive the euphotic depth ($\underline{z}_{eu}$, above which net photosynthesis is sustainable), and approximation of the vertical extinction coefficient for each occasion, and the proportion of that attributable to algae. The remainder (due to inert particles) could then be tested for correlation against discharge. Without reference to the original data, realistic calculation has not been attempted here. A very rough approximation was derived by regressing a derivation of the apparent euphotic depth (viz.,$3.7/\underline{z}_{eu}$, equivalent to the minimum vertical extinction coefficient, ε_{min}) against chlorophyll content for occasions where the latter was high, and dominated by S. hantzschii, and discharges were generally low. This gave:

$$\varepsilon_{min} = 1.04 + 0.013\ [chl];$$

that is, 1 mg chlorophyll $m^{-3} \equiv 0.013 m^{-1}$. Subtracting ($0.013 \times$ chlorophyll concentration) from the calculated ε_{min} values for other dates spanning the Stephanodiscus bloom period left a series of residual values, (ε_R) which, when regressed against corresponding discharges (q, in $m^3\ s^{-1}$) derived from Lack's (1971) histograms, yielded the equation

$$\varepsilon_R = 0.11\ \underline{q} - 1.10.$$

This equation, then, gives a 'rule of thumb' relation between turbidity and discharge for this section of the Thames, during the Stephanodiscus bloom. Roundly, ε_R solves at $0.01\ m^{-1}$ at $10 m^3 s^{-1}$, $1.1\ m^{-1}$ at $20\ m^3\ s^{-1}$ and $3.4\ m^{-1}$ at $40\ m^3\ s^{-1}$. The capacity to support productive biomass declines with increasing discharge. It must be borne in mind that even these approximations make no allowance for variations in incident light energy, respiration rate, or the 'quality' of the chlorophyll measured (see 5.5.2.2.(iii) above).

5.5.2.4. The effect of light

This consideration of turbidity in Thames water becomes relevant in evaluating the role of light in the ecology of phytoplankton. Growth should take place when photosynthetic production in the daylight period exceeds respirational losses over 24h; the former is influenced by the quality and duration of irradiance income as well as by the proportion of the water wherein there is insufficient energy to support photosynthesis. The latter, as shown above, is dependent in turn on the attenuation of light brought about by suspended matter, to which the plankton itself contributes. The central question to be answered is not how much growth can be sustained but at what level of turbidity will further net growth be prevented. Various related formulations of this quantity are available, owing to Talling, to Vollenweider and to Steel, which have been compared in Reynolds (1984a). A simplified application of Talling's (1957) equation, following Reynolds (1984a), can be used to generate approximations of maximum chlorophyll contents of river water at different times of the year, varying river depths and for different turbidities owing to discharges. The calculations in Table 1 apply to a river depth of $\sim$4m (cf Berrie 1972a) assuming maximum photosynthetic rate to exceed dark respiration by a factor of 15, that ε_R is as predicted by the regression equation in 5.5.2.3.(i)(c) above and that chlorphyll concentration is given by $(\varepsilon_{min} - \varepsilon_R/0.013)$. The tabulated values then state the maximum active chlorophyll concentrations that could be supported. Compared to the observations of Kowalczewski & Lack (1971), the poor net photosynthetic production at winter discharges is apparently well predicted and the equinoxial biomasses are of the correct order, but the summer crops cannot be said to have been light limited in the classical sense. These data are not directly applicable to the lower Thames, where different water depth, turbidity and critical discharge factors may apply. Nevertheless the observed chlorophyll maxima near the 1975 summer solstice and minimum discharge ($\sim$270 mg m^{-3}) could be assumed to have been light-limited if either depth, ε_R or the extinction of increment due to chlorophyll, was increased by 50%.

TABLE 5.1

		Chlorophyll, mg m^{-3}		
Time	ε_{min} m^{-1}	at 10m^{3}s	at 20 m^{3}s^{-1}	at 40m^{3}s^{-1}
WINTER SOLSTICE	2.27	174	90	0
EQUINOX	3.88	298	213	37
SUMMER SOLSTICE	5.50	422	338	161

5.5.2.5. The effect of nutrients

Of the three key nutrients that are thought to occupy critical roles in the ecology of phytoplankton in freshwaters, two (phosphorus and nitrogen) are usually present in Thames water in the order of milligrams per liter. Neither Kowalczewski & Lack (1971) nor Lack (1971) found strong evidence that either was severely limiting to phytoplankton production and this view is supported by the year-round bioassays of Thames water, using Monoraphidium as test organism, presented by Collie & Lund (1980). There is some possibility of occasional rate limitation of algal growth and, owing to the interplay with dilution rate, this may place an apparent absolute limit on algal concentration. There is no clear evidence to refute or confirm this statement.

The position with regard to the third element, silicon, is a little more complex. Lack (1971) reported that significant decreases in silicon concentration (as SiO_2) accompanied sustained diatom growth in the river. At Reading, winter levels varied in the range equivalent to 13-17 mg SiO_2 l^{-1}, which, according to Swale's (1963) data, is theoretically sufficient to sustain the production of some 340 to 440 x 10^6 Stephanodiscus cells l^{-1}, or rather more than 1000 µg l^{-1} of chlorophyll. Of course, planktonic diatoms will compete with benthic forms for available silicon, but superficially, light might be expected to become limiting long before silicon was depleted to a limiting concentration. Nevertheless, Lack's (1971) data showed that spring Stephanodiscus blooms could be accompanied by falls in the silica concentration to about 5 mg l^{-1}. In April 1968, the concentration fell from 5.2 to 2.4 mg SiO_2 l^{-1}, after the Stephanodiscus population had reached its maximum of 70 x 10^6 l^{-1}. Lack (1971) pointed out that a further division of the diatoms would require nearly 2.7 mg SiO_2 l^{-1}, almost exactly accounting for the observed SiO_2 depletion, but the diatom population dropped catastrophically, to $\sim$ 10 x 10^6 l^{-1}. Moreover, even the concentration remaining should not be expected, by itself, to limit further diatom growth, for the same alga can maintain rapid growth in lakes at below 2 mg SiO_2 l^{-1}. Downstream removal of cells constituting the maximum does not seem to apply either, judging from the data of Lack et al. (1978). The end of the spring diatom bloom in the Thames has not been satisfactorily explained.

5.5.2.6. The effect of water temperature

Temperature of Thames water generally fluctuates within the range 4-20°C and may be considered unlikely ever to prevent algae from growing. The physiological activity of all algae is temperature-sensitive and this will be reflected in the growth rate. Insufficient data are yet available in order to make precise predictions of the temperature-dependent growth in the River Thames but an approximation can be made for Stephanodiscus hantzschii. Assuming a light- and nutrient-saturated growth rate of 1.2 d^{-1} at 20° (Hoogenhout & Amesz 1965), a Q_{10} of growth rate in the order of 2.2 (cf Reynolds 1984b) and direct dependence upon the light period, maximal growth rates in clear, standing river water would not be expected to exceed 0.80 d^{-1} in summer (20°, 16th day) or 0.12 d^{-1} in winter (5°, 8h day). Dilution, turbidity and loss processes would further restrict the possible rates of apparent increase, perhaps dictating the low or declining biomasses observed in winter.

5.5.2.7. The effect of other loss processes.

The impact of these on these phytoplankton has been scarcely quantified in rivers. Direct grazing by zooplankton can never be discounted, although limnetic studies (reviewed in Reynolds 1984a) suggest that certain thresholds of temperature and food concentration are essential to the development of a significant zooplankton community. Qualitatively, zooplankton in the Thames is similar in composition to that of many eutrophic lakes, though population densities rarely achieve the densities that would suggest grazing was a significant brake on phytoplankton development (Bottrell 1975a,b), yet grazing may contribute to other effects. The significance of filter feeding by benthic Unionid mussels on phytoplankton has not been evaluated. Sedimentation, especially of post-maximum diatoms, may be more important than realized hitherto, bearing in mind the truncated vertical water columns of rivers and accelerated sinking rates achieved by diatoms at the end of population growth. Observations on epilithic algae in circulating channels presented by Marker & Casey (1982) certainly attest that such sedimentation can occur. Diatoms settling onto fluvial deposits may be resuspended when boundary layers are compressed or disrupted by strong turbulence associated with localized currents but it will not occur everywhere nor necessarily continuously. Given sinking rates in the order 0.1 - 1.0 m d^{-1} losses equivalent to -0.03 to -0.29 d^{-1} might be sustained from a 4 m column. Though these may be occasionally effective in clearing non-growing diatoms from suspension - the decline observed by Lack (1971) referred to above (5.5.2.5.) is equivalent to -0.13 d^{-1} - it is probable that sedimentation, too, merely contributes to several simultaneous dynamic attrition processes. Nevertheless, the hypothesized sequence of declining flow, increasing insolation, clarity and accelerated sinking rate, perhaps abetted by benthic cropping, as a possible explanation for the end of spring diatom maxima might be worthy of future investigation.

5.5.2.8. A tentative interpretation of phytoplankton behaviour in the Thames.

Assembling the various pieces of evidence concerning the possible ecological roles of the environmental factors operating in the Thames above Teddington Weir the following tentative account is offered.

In the early part of the calendar year, day-length and water temperature determine that algal growth will be slow and will barely offset the respirational losses, compounded by high turbidity and mean velocity associated with winter discharges. Initiation of growth occurs partly in response to increasing day length and enhanced light income, coupled with declining discharges that permit greater light penetration into the river water and a proliferation of the 'reservoir' effect (5.5.2.3.(b) above). The acknowledged productive frequency fluctuations in perceived irradiance presumably affords them a competitive edge over other algal groups. Nutrient levels remain high and permit net growth to be maintained until self-imposed light limitations become operative. Potentially, populations approach or, in the lower Thames, actually reach, absolute light-limited proportions at which the standing crop more or less stabilizes. These events, moreover, occur almost simultaneously in whole reaches of the river.

Whether or not silicon-limitation prevents the indefinite maintenance of high standing crops of diatoms is debatable. There seems little doubt that silicon-exhaustion remains a possibility in productive years and against reduced background concentrations. It may occur more frequently than suggested (5.5.2.5. above) although rapid subsequent recovery in dissolved silicon levels may interfere with the detection of low concentrations at conventional sampling intervals. What does seem clear, however, is that late-spring population collapses are not always mediated by increased discharges (in fact, they often decline further) and that there is little evidence of downstream passage of the maximum; other responses to increasing temperature and declining discharges may apply (5.5.2.7. above).

The diatoms eventually recover their numbers during summer, though with the accompaniment of many other species of Chlorococcales, Cryptomonads and, perhaps, Cyanobacteria. Most of these may be considered to have colonist, 'r'-selected growth strategies but the effects of 'seeding' in from tributaries and standing water sources in the catchment cannot be discounted. This is especially true of the relatively slow-growing Cyanobacteria like Microcystis, which though often noted in the river, rarely experiences the warm-water, high-light sluggish conditions necessary for its growth. Summer species are also more sensitive to deteriorating light levels towards the autumn, that are enhanced by seasonal increases in discharge and turbidity. Presumably, these factors restore a marginal selective advantage in favour of diatoms, which eventually stage an autumnal bloom. Cell populations seem generally lower than those of the spring maximum but chlorophyll concentrations may be as high or higher; this may be attributable partly to a relatively enhanced adaptive response to low light (more chlorophyll per cell) and partly to non-planktonic sources of chlorophyll brought into suspension. Restoration of winter discharges, turbidity coupled with declining irradiance income and falling temperatures reduce the phytoplankton standing crops to winter levels.

5.5.2.9. Freshwater phytoplankton below Teddington Weir

It is relevant to consider briefly the behaviour of freshwater algal populations carried into the tidal Thames. Depending upon the fluvial discharge rate and the tidal cycle, Thames water remains recognizably 'fresh' for the first 10-50 km below Teddington Weir but halinity increases downstream as the river flow becomes progressively 'diluted' by sea water. So far as is known, the cells of freshwater algae are ultimately incapable of adapting their osmoregulatory and surface charge capacities to the physiological stresses imposed. Flocculation, loss of physiological vigour and, eventually, death combine with volumn-dilution to bring about a catastrophic decline in the suspended populations of freshwater algae, some 30-60 km downstream of Teddington.

Even in the freshwater reaches immediately downstream of Teddington, tides modify the environmental conditions for phytoplankton development with respect to those obtaining upstream, principally by reducing mean velocity (increasing travel time, reducing non-algal turbidity) which effect is likely enhanced by the half-tide weir at Richmond, that retains river water at low tide. Thus, there remains scope for further cell division of populations introduced into the tidal section at Teddington. The extent to which this might occur will be a function of freshwater flow and the concentration of the phytoplankton in the water entering the tidal river. These factors might be expected to work against each other to mitigate potentially severe water-quality problems, for at high fluvial discharges, phytoplankton will be introduced at low concentrations, while at low discharges, sea water penetrates further upstream. The combination of intermediate flows and substantial sub-maximal algal populations appears to be the least desirable in this respect. Such conditions might obtain shortly before and shortly after population maxima in the non-tidal reaches.

There are apparently few biological data available that can be cited in support of the above hypotheses, with the exception of weekly chlorophyll concentrations made available by Thames Water Authority for a number of stations downstream of Teddington Weir, during 1975 and 1976. These have been compared with corresponding data (presented by Whitehead & Hornberger - in press) for the non-tidal reach above Teddington over the same time span. During the low flows of 1976 ($\sim 10 m^3 s^{-1}$) the fluctuations of chlorophyll concentrations below the weir closely tracked those upstream, but the levels consistently reduced in magnitude between Teddington (maxima, 100-260 mg m^{-3}, similar to those obtaining simultaneously above Teddington) and London Bridge (corresponding maxima, 20-60 mg m^{-3}), 31 km downstream; at intermediate stations (15-22 km downstream), the corresponding maxima were 50-210 mg m^{-3}) During 1975, when the drought was less severe (discharges 20-30 $m^3 s^{-1}$) chlorophyll concentrations in the non-tidal Thames exceeded 250 mg m^{-3} in May-June, simultaneous peaks, of similar order 15-22 km downstream and of 50-100 mg m^{-3} at London Bridge, were experienced. Moreover, there were numerous occasions through the summer period on which the concentrations at the intermediate tidal stations significantly exceeded the submaximal concentrations (50-150 mg m^{-3}) entering the tidal basin at Teddington.

5.5.3. Predicted impact of regulated reductions in vernal discharges

In essence, the TWA proposals would enable greater abstraction (thereby reducing discharge) from the Thames through spring, in order to avoid more severe flow reductions in the summers of very dry years. It is recognized that during wet springs and summers, the impact on the environment of the phytoplankton would be barely perceptible. Concern therefore centres upon likely events in a dry spring and through a continuing summer drought and in the longer term, the consequences of future increases in abstraction and sewage disposal requirements.

If the above interpretation is substantially correct, the general implication is a raising of the upper limit on the maximum standing crop with a greater proportion of the chlorophyll being attributable to phytoplankton. However, the calculations upon which Table 1 is based do not permit any further increase in active chlorophyll content at discharges of $< 10 m^3 s^{-1}$, or $864\ 000 m^3 d^{-1}$. This figure approximates closely to the present statutory minimum flow over Teddington Weir; in other words, reduction of this minimum, by itself should NOT significantly increase the chlorophyll capacity of the lower Thames. Significant increases in the instantaneous carrying capacity would be evident only in response to flow reductions when natural discharges were already well in excess of 10 $m^3 s^{-1}$. Assuming that increased abstraction were to occur before the discharge at Teddington fell to this level, then it may also be predicted that the opportunity to reach the carrying capacity would be provided earlier in the year. Thus, the principal effect of the proposed revisions would be to bring forward the point at which absolute light-limitation could occur and, hence, to extend the period through which the condition might obtain. Equally, some foreshortening of the period of diatom dominance could be anticipated, allowing an earlier inception and possible lengthening of the period in which other species could be abundant: there might then be relatively larger crops of Chlorophyceae and an opportunity for Cyanobacteria to dominate. Although the theoretical upper limit placed on population size would remain unaffected, temporary floral changes, especially those involving bloom-forming cyanobacteria, may present a different category of environmental side-effects. Because the relationship is largely independent of major adjustments in nutrient loading, projected increases in the relative contribution of recycled water to river flow (17% by the year 2006) will not alter this general condition. Locally, however, direct abstration from the lower non-tidal Thames enhances the instantaneous phytoplankton carrying capacity (most strikingly so when the discharge is in the range 20-50 $m^3 s^{-1}$).

These statements apply equally to the tidal sections of the lower Thames wherein the water remains 'fresh'. Lowered flows in spring may provide additional capacity for further phytoplankton growth, within the limits posed by self shading and by upstream penetration of saline water.

Events in the lower Thames during the drought years of 1975 and 1976 (recorded by Whitehead & Hornberger, in press) attest to the validity of these deductions. In 1975, discharges at Teddington Weir fell from a winter high of around 300 $m^3 s^{-1}$ to about 40 $m^3 s^{-1}$ by the end of May, and to 20-30 $m^3 s^{-1}$ through the summer. There was a modest increase towards the end of the year, but discharges remained generally low (30-40 $m^3 s^{-1}$) through winter and declined further during the spring (< 30 $m^3 s^{-1}$) and summer of 1976, reaching a late July low of < 10 $m^3 s^{-1}$; from September 1976, discharges were progressively restored until by December, they once again exceeded 160 $m^3 s^{-1}$. Actual chlorophyll concentrations at Teddington through this period showed the usual midwinter minimum (< 14 mg chla m^{-3}) and a sharp rise between the spring equinox and the summer solstice, from $\sim$20 to $\sim$270 mg m^{-3}, broadly in line with the maximal crops suggested in Table 1. The 'usual' summer minimum was evident ($\sim$ 30 mg m^{-3}) but levels 'recovered' to around 180 mg m^{-3} shortly after the 1975 autumnal equinox, thereafter declining to $\sim$5mg m^{-3}

at the winter solstice, again in line with the tabulated predictions. Increase in chlorophyll concentration recommenced during February 1976, to reach a maximum of < 200 mg m^{-3} a month or so before the spring solstice. The midseason decline (to ∿30 mg m^{-3}) had been passed before the summer solstice, peaking in August at around 240 mg m^{-3}.

If the events of 1974 are considered to provide a more typical comparison, then it is of interest that the vernal increase was not apparent until just before the spring equinox and after a seasonal fall in discharge from > 150 $m^3 s^{-1}$ to between 40 and 50 $m^3 s^{-1}$, with which the maximum (∿ 180 mg m^{-3}) coincided. Summer discharges scarcely fell below 20 $m^3 s^{-1}$ and, by the autumn equinox, had recovered to > 100 $m^3 s^{-1}$. That chlorophyll concentrations should have remained < 15 mg m^{-3} through the late summer is, therefore, also in line with the Table 1 predictions.

These considerations imply that the maximal populations in the lower Thames do, indeed, become light limited but that the limit on chlorophyll content moves higher at the lowest flows (minimal turbidity). Thus, the downward regulation of discharges, such that they fall below 30 $m^3 s^{-1}$, may allow the light-limited carrying capacity to increase but not to the point where it will exceed the highest levels already experienced. Indeed, a good case could be made for allowing the artificial reduction in discharge at the earliest possible point in the year and for subsequent maintenance of flows of 30-40 $m^3 s^{-1}$ through the summer: although this would allow marginally higher vernal standing crops to be achieved, subsequent maxima much exceeding 180 mg m^{-3} might be avoided. Against this, similar or slightly increased standing populations than heretofore might be maintained further downstream into the tidal reaches, but again, not exceeding the highest levels already experienced.

Finally, it might be added that the forecasting model developed by Whitehead & Hornberger (1984), to describe the same series of environmental and chlorophyll data for the period 1974-1976 generated a remarkably good approximation of the observed fluctuations in chlorophyll concentration through 1974 and 1975 while only marginally underestimating those of summer, 1976. The view that low flows advance the onset and, perhaps, extend the duration of maximal populations, but without significant increase of the proportions of the light-limited maximum, is corroborated by the accuracy of the Whitehead-Hornberger predictions.

5.5.4. Conclusions

From a consideration of the limited literature describing the production ecology and population dynamics of the phytoplankton of the Thames, it has been deduced that the sizes of maximal populations in the lower Thames are determined principally by the light requirements of the algae in relation to the light income and variable turbidity levels associated with variable discharge. Although reduced flows may raise the thresholds of light-limitation, and the proportion of the extinction that can be attributed to the algal chlorophyll concentration, the principal effect concerns the phasing and duration of biomass maxima, subject to unconfirmed discharge-independent processes that bring about mid-seasonal declines. Floral changes in summer, perhaps favouring greater standing crops of bloom forming Cyanobacteria cannot be discounted, however.

5.6. HIGHER PLANTS (D.F. Westlake)

There is not much information available on the higher plants of the reaches of the river Thames affected by the proposals. Between Teddington Weir and London Bridge (tidal freshwater reaches) the combination of frequent changes of level, boat traffic, and urban development make it highly probable that there is no significant development of higher plants within the river or on its banks. There are some adjacent areas, e.g. Syon Park, St. Margaret's Grounds Lake, which have plants that could be affected by changes in levels or salinity.

5.6.1. The regional staffs of the Nature Conservancy have made surveys of the floras of most of the major tributaries of the Thames within the affected reaches, but the main river has not been included. Baker (1939) described the flora of various reaches near Oxford, Berrie (1972a) has described a reach at Reading including the depth distribution of higher plants and Furse (1978) has reported on the ecology of the river between Lechlade and Benson Lock near Wallingford. Most of these results deal with navigable waters which are unlikely to differ greatly from the reaches between Romney Weir and Teddington Weir. John (pers. comm.) has made some observations on macrophytes throughout the navigable Thames, during research on benthic algae. Haslam & Wolseley (1981) have described the vegetation of English rivers in general, and predictions using their data agree with the direct observations available and confirm that important differences downstream of Windsor are unlikely. The probable vegetation between Romney and Teddington is summarised in Table 5.2. No evidence has been found of any sites with rare or sensitive species.

Table 5.2.

Expected aquatic higher plants between Windsor and Teddington (based on Berrie 1972a; Furse 1978; Haslam & Wolseley 1981 and Dr D. John pers. comm.)

<u>Submerged and Floating-Leaved Vegetation</u>

(Absent from main channel, found in margins and backwaters. Unlikely to be significant in water greater then 1m deep.)

Frequent	*Nuphar lutea*[1,3]	Common	*Sparganium emersum*
			Polygonum amphibium[3]
			Potamogeton pectinatus
			Fontinalis sp.
			Cinclidotus fontinaloides)
			Rhynchostegium)
			ripariodes)Mosses
	Plus 4 or 5 other species		*Octodiceras fontanum*)

<u>Marginal emergent vegetation</u>

(Replaced by grasses where grazed or trampled)

Fairly frequent	*Sparganium emersum* *Schoenoplectus lacustris*[2]	Common	*Phragmites australis*[2,3] *Acorus calamus*[1,3]
Occasional	*Iris pseudacorus*[1]		Plus 8-10 other species
	Carex spp.		
	Sagittaria sagittifolia		

1. Particular amenity value
2. Particular bank protection value
3. Most likely to decrease if pollution increases or flow decreases.

5.6.2. Two possible consequences of the proposed changes in the rules governing abstraction are (a) that the current demand is satisfied by abstraction according to the proposed rules, (b) that increased demand leads to long periods of low or no flow over Teddington Weir unrestricted by statutory minimum flows. These situations are compared with the results of operating the current rules.

The variable factors affecting the distribution and growth of higher plants in the navigable non-tidal Thames will be depth, turbidity (algal and non-living), human disturbance by boats and trampling, and grazing or mowing (Westlake 1975 Haslam 1978). Water velocity and sewage pollution may have some local effects. Depth and human disturbances are not expected to alter as a result of the proposals, and these will continue to be important influences.

For the example years of 1934 to 1935 predicted mainstream velocities under MRB range from 1 to 40 cm s^{-1}, when they differ from the prediction under current rules, which are between 1 to 10 cm s^{-1} over these periods. Major direct effects of velocity on the higher plants, which are near the bank or the bottom, are not expected for changes of this magnitude.

Turbidity is increased by upstream and tributary inputs (not affected by the proposed changes), increased velocities (increased suspension), decreased discharges (less dilution of turbid inputs) and growth of phytoplankton. Should turbidity increase it would decrease light penetration, decreasing submerged plant growth and ultimately restricting submerged plants to shallower water (where the underwater irradiance is greater than about 5% of the incident). Since velocity increases with discharge, velocity and discharge dilution effects will tend to oppose each other. Furthermore the phytoplankton section points out that phytoplankton often grow until stopped by the combination of algal and non-living turbidity, so that low non-living turbidity leads to high algal turbidity and little change in the total. Overall, turbidity effects consequent on decreasing discharges early in the year would tend to favour plant growth in the spring and increasing discharges in the summer would tend to restrict plant growth in the summer. However given the magnitude and frequency of the predicted changes, such detectable effects on the higher plants are unlikely.

5.6.3. In some reaches the proposed changes in abstraction are expected to lead to occasional deterioration of chemical conditions downstream of effluent discharges (e.g. lower dissolved oxygen; higher nitrate, phosphate and ammonia). These may slightly change the occurrence, distribution and relative abundance of some species (see Table 5.2 footnote 3) and this could become significant if there were long periods of little or no flow above Teddington Weir.

If such periods of stagnation occurred in this reach, the development of dense cover with duckweeds, (Lemna spp, Azolla filiculoides) might be expected, because these thrive in fertile static water. Such growths are unsightly and restrict gaseous exchanges and light transmission at the surface, leading to low oxygen and high carbon dioxide concentrations in the water. However, this was not observed in 1976, and the exposure to wind and the frequent disturbances by boats make this unlikely.

5.6.4. Overall the proposed changes are unlikely to have a detectable effect on the aquatic higher plants, but if increased demand or untimely abstraction allow near-static conditions to prevail for two or three months in spring or summer there may be changes in abundance of some species in favour of the species most tolerant of human interference and eutrophication (see Haslam & Wolseley 1981 and Table 5.2. footnote 3). It would help to prevent such changes if abstraction was maximized as early as possible in the year to avoid the low discharges which occur during the spring in some predictions under MRB.

5.7. MACROINVERTEBRATES OF THE RIVER (M.T. Furse)

5.7.1. Introduction

In common with most deep, lowland reaches of British rivers, the macroinvertebrate fauna of the Thames between Romney and Teddington Weirs is poorly known. No published information has been found for this reach and the few data available have been obtained by TWA during routine biological monitoring. These samples were collected by pond-net and take no cognizance of communities in the mid-stream zone.

The potential permutations of implementation of the MRB system appear to be as numerous as the range of climatic and natural discharge patterns likely to occur. Variation in demand may also alter the way in which the system is operated.

Against this background interpretation of macroinvertebrate response to the MRB strategy must inevitably be restricted to generalized observations concerning the tolerances of those species known or believed to be present. These observations are set against the operating conditions assumed to apply in drought and non-drought years.

5.7.2. The macroinvertebrate community

A composite picture of those macroinvertebrates known to be, or thought likely to be, present has been derived from three sources. There are those collected in routine monitoring, those found in other sections of the river with comparable environmental conditions and those predicted by a modelling system developed by the FBA River Communities Project (Wright *et al* 1984).

5.7.2.1. Known information .- Data made available by TWA consist of four pond-net samples collected between Egham and Teddington Weir in autumn 1980 as part of the DoE River Water Quality Survey. They reflect a community dominated numerically and in species diversity by Oligochaeta (worms), Gastropoda (snails), Bivalvia (mussels) and Trichoptera (caddis). Also abundant at most sites were Tricladida (flatworms), *Asellus aquaticus* (water hog-louse), Amphipoda (freshwater shrimps), Hydracarina (water-mites) and *Caenis moesta* (small mayfly). A diverse Hirudinea (leech) fauna was recorded but, in keeping with their predatory habits, no species were consistently abundant. Chironomidae (non-biting midges) were not identified to species and so no information is available as to their diversity.

5.7.2.2. Extrapolated information - The only extensive longitudinal survey of macroinvertebrate communities in the navigable Thames, upstream of Teddington, was carried out by the FBA under contract to TWA (Furse, 1978). This study, which was undertaken from a boat, reported on 18 inter-lock reaches between St. John's and Day's Locks. Biological samples were collected from mid-stream using air-lifts (Mackey 1972) and from vegetation and marginal areas separately using pond-nets.

Comparison of selected physical and chemical variables for the reaches studies by Furse (1978) and those in the lower Thames indicates close similarity between environmental conditions. The principal differences lie in the total BOD levels which are both higher and more variable in the lower Thames. Overall, the similarity in conditions is taken as validating the extrapolation. This view is substantiated, in part, by the consistency in community composition along the 81 km section studied by Furse (1978).

The results of the middle Thames survey confirmed the numerical importance of those taxa found commonly between Egham and Teddington, but also provided information on those taxa particularly associated with habitats not sampled by TWA in the lower Thames.

Common on vegetation, but not taken in the lower river, were many species in the families Naididae (small worms), Baetidae (mayflies) and Corixidae (water boatmen). Their apparent absence between Egham and Teddington may follow from the sparsity of marginal vegetation rather than changes in water quality. On the other hand distinctions between marginal and mid-stream communities were less pronounced (Furse 1978) principally reflecting differences in species composition of the Chironomidae.

Chironomidae, with 41 recorded taxa, and Hydracarina, with 19, were particularly diverse in the middle reaches (Furse 1978) and would undoubtedly have considerably extended the number of taxa collected from Egham downstream had they been further identified. The association between particular chironomid taxa and habitat types shown by Furse (1978) were considered in much greater detail by Mackey (1976a,b; 1977a,b) who worked on a 3.3 km reach of the Thames at Reading. This work formed part of a more extensive project summarised by Mann (1964, 1972), Mann et al. (1972) and Berrie (1972a). Mann (1964) comments that whilst there were marked differences in macroinvertebrate communities in a cross-stream transect there was little longitudinal variation.

5.7.2.3. Predicted information - The technique being developed by the FBA River Communities Project allows the macroinvertebrate taxa likely to be recorded at a site to be predicted from its known environmental characteristics. The probabilities of each taxon occurring, if the site is sampled three times, are also given (Wright, Armitage & Furse unpublished). The procedure utilises multiple discriminant analysis on environmental data to ascribe sites to a series of pre-defined biological groupings.

The requisite environmental data on sites at Egham and upstream of Teddington Weir were acquired from a variety of sources (Table 5.3.). Most data are considered reliable although information on substratum characteristics follow conversations with TWA personnel rather than direct measurement. A primarily gravel-dominated substratum was also a common feature of most reaches of the middle Thames (Furse 1978).

Table 5.3.

Values of environmental variables used to predict the macroinvertebrate species likely to be present at selected sites on the lower Thames. Predictions are based on the Harmonised Monitoring sites at:-

(1) NSWC intake, Egham TQ 023 718
(2) Teddington Weir TQ 171 714

Variable	Site	
	Egham	**Teddington**
Total oxidised nitrogen (mg l^{-1} N)	7.7	7.04
Chloride (mg l^{-1} Cl)	35.51	38.07
Total alkalinity (mg l^{-1} $CaCO_3$)	210.05	188.87
Mean substratum particle size (phi values)	-0.645	-0.225
Distance from source (km)	206	234
Altitude (m)	14	6
Slope (m km^{-1})	0.4	0.3
Mean depth (m)	1.92	2.44
Mean width (m)	104.5	185.9
Mean air temperature (°C)	10.65	10.875
Air temperature range (Jan-July) (°C)	13.6	13.55

Total oxidised nitrogen, chloride and total alkalinity are mean values for 1979-83 supplied by TWA, Pollution Control West Division.

Substratum assumed to be principally gravel, with some smaller particles, on advice of TWA, Pollution Control West Division.

Temperature values derived from Meteorology Office reports.

Altitude and slope derived from O.S. maps.

Depth and width and distance from source values derived from Section 2.

A feature of the predictive technique is that, for a given range of probability of occurrence, the proportion of taxa actually found should be equivalent to the mid-point of the probability range. Thus for predictions in the range 66.5-100% (Table 5.4) about 83% should be observed if the forecast is accurate. However the

TABLE 5.4. Species predicted to occur at either Egham or Teddington with a probability in excess of 66.5%, with an indication of their presence at selected sites between Lechlade and Teddington.

SPECIES	Probability of occurrence		Recorded occurrence									
	Egham	U/S Teddington Weir	Buscot	Northmoor	Swinford	Abingdon	Day's	Egham	U/S Littleton Intake	U/S Raven's Ait	Kingston (Lower Ham Road)	D/S Teddington Weir
Orthocladius/Cricotopus	97.8	99.3	+	+	+	+	+	?	?	?	?	?
Micropsectra/Tanytarsus	95.8	98.7	+	+	+		+	?	?	?	?	?
Asellus aquaticus	95.6	98.0	+	+	+	+	+	+	+	+	+	+
Sphaerium corneum	97.5	97.6	+	+	+	+	+	+	+	+	+	+
Erpobdella octoculata	93.4	96.6	+	+		E.sp	+	+	•	+	+	+
Limnodrilus hoffmeisteri	83.4	95.0	+	+	+	+	+	?	?	+	+	+
Hydracarina	94.4	86.6	+	+	+	+	+	+		+	+	+
Glossiphonia complanata	92.5	85.2	+	+		+		+	+	+	+	+
Rheotanytarsus/Paratanytarsus	92.3	85.7	+	+	+		+	?	?	?	?	?
Ceratopogonidae	90.8	86.4	?	?	?	?	?	+	+	+	+	
Lymnaea peregra	76.1	90.5	+	+		+	+	+	+			+
Piscicola geometra	65.8	87.2				+	+					
Potamothrix hammoniensis	61.6	85.8	+	+	?	+	?	+		+		+
Pisidium subtruncatum	85.4	72.5							?	?	?	
Sigara dorsalis	53.0	82.8		+	+	?	?					
Caenis moesta/macrura	81.4	71.7	+	+	+	+	+	+	+	+	+	+
Bithynia tentaculata	73.8	80.2	+	+	+	+	+	+	+	+	+	+
Stylaria lacustris	73.7	80.1	+	+	+		+					
Oulimnius tuberculatus	79.7	72.2	+	+			+	+	+	+	+	
Psammoryctides barbatus	79.3	71.0	+	+		+	+	+	+	+	+	+
Lumbriculus variegatus	67.9	78.8	?	?	?	?	+	+	+			
Pisidium nitidum	77.5	57.7		+					?	?	+	
Procladius sp.	59.8	77.1	+	+	+	+	+	?	?	?	?	?

Physa fontinalis	68.8	76.8	+	P.sp							+	+
Thienemannimyia group	76.7	60.0	+	+	+		+	?	?	?	?	?
Centroptilum luteolum	63.1	75.5	+	+		+	+					
Gammarus pulex	75.1	38.2	+	+			+		+	+		+
Hydroptila sp.	74.9	46.8	+	+	+		?					
Eukiefferiella sp.	72.5	35.7	+					?	?	?	?	?
Microtendipes sp.	72.5	58.8	+	+	+			?	?	?	?	?
Baetis scambus/fuscatus	72.4	44.3	+	+			+					
Planorbis albus	72.2	67.6	+	+			+				+	
Helobdella stagnalis	71.4	69.7	+	+	+	+	+	+	+	+	+	+
Crangonyx pseudogracilis	46.7	71.3	+	+	+	+	+	+	+	+	+	
Baetis vernus	70.3	35.0	+	+			+					
Stylodrilus heringianus	70.0	33.8	+	?	?	+	+	+	+			
Polypedilum sp.*	69.4	69.4	+	+	+	+	+	?	?	?	?	?
Potamopyrgus jenkinsi	67.3	45.9	+	+	+	+	+				+	+
Polycentropus flavomaculatus	66.9	44.2	P.sp	P.sp			P.sp	+	+	+		
Baetis rhodani	66.6	43.6	+									

Data for Buscot to Day's taken from Furse (1978)

Buscot	=	St John's Lock to Grafton Lock
Northmoor	=	Shifford Lock to Pinkhill Lock
Swinford	=	Pinkhill Lock to Eynsham Lock
Abingdon	=	Sandford Lock to Culham Lock
Day's	=	Clifton Lock to Benson Lock

Samples collected between 26.7.77 and 5.9.77

Egham to Kingston data supplied by TWA, Pollution Control West Division

Egham (U/S NSWC intake) sampled 28.10.80
U/S MWD Littleton intake sampled 16.10.80
U/S Raven's Ait (Surbiton) sampled 26.11.80
Kingston, Lower Ham Road sampled 19.11.80

D/S Teddington Weir data supplied by TWA, Pollution Control South Division.

D/S Teddington Weir data cover the period 1977-1984

*<u>Polypedilum</u> sp. includes <u>Pentapedilum</u> sp. in Furse (1978)

+ = present
? = possibly present but included in a broader taxonomic category

predictive system is based upon three sampling visits and Furse *et al.* (1981) have shown that a single sample collects only 60-70% of three samples, depending upon the level of identification.

5.7.2.4. Combining this information suggests that, if predictions are reliable, then 50-58% of the expected taxa should be observed in a single sample. Considering only those taxa whose level of identification is equivalent to the predictions, then 56% of the taxa forecast to occur at Egham (Table 5.4) were present in the single TWA sample. It is inferred that the predictions give a reasonable estimate of the taxa that could be recorded at Egham if it were sampled three times.

The situation at Teddington is more complex. The predictions are based on the non-tidal reach upstream of the weir whereas the only available macroinvertebrate data, provided by TWA Pollution Control South, are for the tidal section downstream. Furthermore the species list covers the period 1977-1984 but it is not known how many samples this represents. Fifty-seven percent of the taxa predicted for upstream of the weir were definitely present downstream of it.

Eight taxa forecast to occur at either Egham or Teddington were not recorded at any site in that reach in 1980 (Table 5.4). These were *Stylaria lacustris* (a worm in the family Naididae), *Piscicola geometra* (fish leech), *Centroptilum luteolum*, *Baetis scambus/fuscatus*, *Baetis vernus* and *Baetis rhodani* (all mayflies in the family Baetidae), *Sigara dorsalis* (a water boatman in the family Corixidae) and *Hydroptila* sp. (a small caddis). Most of these taxa have been noted above to be particularly associated with aquatic vegetation. Of the 27 taxa predicted only *Pisidium subtruncatum* (a pea-mussel) was not recorded at any of the sites compared (Table 5.4) although it is known to occur near Sandford Lock (Furse 1978).

5.7.3. Water quality assessment

Many biotic indices have been derived to relate the presence of macroinvertebrates to the quality of the water in which they are found. The best known of these in Britain have been the Trent Biotic Index (Woodiwiss 1964) and the Chandler Index (Chandler 1970). More recently the BMWP system has been recommended for use in the National River Water Quality surveys. The performance of this index has been favourably assessed by Chesters (1980) and by Armitage *et al.* (1983). It has been shown (Armitage *et al.* 1983) that a derivation of the BMWP score, namely the Average Score Per Taxon (ASPT) is more reliable since it is less dependent upon sampling effort or the habitats sampled at the site being examined.

At its extremes the ASPT can vary from 0 to 10 but most sites fall in the range 3 to 7. In theory the lower the ASPT the poorer is the water quality of the site. However Armitage et al. (1983) have shown that the standards expected of sites of different environmental conditions will also vary. Their data suggest that sites having the characteristics of the navigable Thames would have ASPTs in the mean range 4.42 - 5.29, with 10th and 90th percentiles of 4.0 and 5.7, if the water was of reasonable quality.

Assessment of all available data between St. John's Lock and London Bridge shows that most sites fall within the expected mean range with only three lying below the 10th percentile value (Table 5.5.). Two of these, Battersea and London Bridge are within the reach of occasional saline intrusion and need not be considered. The remaining site, at Lower Ham Road, Kingston, has an ASPT of 3.94. It is located downstream of the Hogsmill Stream, and tributary known in the past for its pulses of high ammoniacal nitrogen input into the Thames.

The water quality appears to have slightly improved downstream of Teddington (ASPT 4.18) but still lies near the lower extreme suggested by Armitage et al (1983). It must be strongly emphasised at this point that these values are each based on single autumn samples only and may not represent the prevailing conditions for most of the year. Nonetheless they do draw attention to the particular problems that may exist in this reach.

5.7.4. Environmental ranges of predicted taxa

Approximately 400 running-water sites throughout Britain have been sampled for macroinvertebrates as part of the FBA River Communities Project. Water quality information relating to each site has also been collected producing a powerful data-base for comparing the distribution of taxa with individual chemical variables.

Chemical factors which have been given the greatest emphasis in this appraisal of the MRB scheme have been DO, total BOD, total oxidised nitrogen and ammoniacal nitrogen. The 27 taxa with the highest probability of occurrence at Egham or Teddington, have been ranked in apparent order of sensitivity to deteriorating levels of these variables.

Table 5.5. BIOLOGICAL MONITORING WORKING PARTY (MBWP) SCORES FOR SITES ON THE RIVER THAMES, EXPRESSED AS AVERAGE SCORE PER TAXON (ASPT)

SITE/REACH	ASPT	DATE	SOURCE
St. Johns	5.40	3.8.77	Furse 1978
Buscot	5.43	Aug.1977	Furse 1978
Grafton	5.13	3.8.77	Furse 1978
Radcot	4.95	2.8.77	Furse 1978
Rushey	4.75	2.8.77	Furse 1978
Northmoor	5.25	1.8.77	Furse 1978
Shifford	5.51	Aug.1977	Furse 1978
Pinkhill	5.22	1.8.77	Furse 1978
Eynsham	4.80	Aug.1977	Furse 1978
King's	4.91	29.7.77	Furse 1978
Godstow	4.82	28.7.77	Furse 1978
Osney	4.97	28.7.77	Furse 1978
Iffley	5.36	July/Aug.1977	Furse 1978
Sandford	4.43	27.7.77	Furse 1978
Abingdon	5.22	27.7.77	Furse 1978
Culham	4.67	26.7.77	Furse 1978
Clifton	5.13	July/Sept.1977	Furse 1978
Day's	4.56	26.7.77	Furse 1978
Caversham	4.76	31.12.80	TWA - PCW
Reading	4.90	1962?	Mann 1964
Sonning	5.13	28.10.80	TWA - PCW
Boveney	4.71	10.11.80	TWA - PCW
Egham	4.96	28.10.80	TWA - PCW
U/S Littleton Intake	5.29	16.10.80	TWA - PCW
U/S Raven's Ait, Surbiton	4.87	26.11.80	TWA - PCW
Kingston (Lower Ham Road)	3.94	19.11.80	TWA - PCW
D/S Teddington Weir	4.18	1977-1984	TWA - PCS
Petersham	5.00	1977-1984	TWA - PCS
Kew	4.64	1977-1984	TWA - PCS
Battersea	3.80	1977-1984	TWA - PCS
London Bridge	3.25	1977-1984	TWA - PCS

PCW = POLLUTION CONTROL WEST DIVISION

PCS = POLLUTION CONTROL SOUTH DIVISION

A possible order of vulnerability of the 27 taxa to deteriorating water quality is summarised in Table 5.6. Some taxa, including *Lumbriculus variegatus* (a worm), *Oulimnius tuberculatus* (a beetle) and *Gammarus pulex* (a freshwater shrimp rank consistently high for each variable. Others have particular sensitivity to single factors. *Centroptilum luteolum* (a mayfly) appears especially vulnerable to ammoniacal nitrogen whilst *Piscicola geometra* (fish leech) requires relatively high DO concentrations.

Table 5.6. Possible order of vulnerability to deteriorating water quality of taxa with a predicted probability of occurrence at Egham or Teddington of at least 75%

TAXON	RANKING				
	Dissolved Oxygen	Total BOD 5	TON	Ammon. Nit.	TOTAL
Lumbriculus variegatus	4	1	1	2	8
Oulimnius tuberculatus	2	3	2	3	10
Hydracarina	1	4	4	4	13
Thienemannimyia sp.	6	2	3	8	19
Orthocladius/Cricotopus	3	5	5	11	24
Micropsectra/Tanytarsus	7	6	7	10	30
Rheotanytarsus/Paratanytarsus	5	12	8	6	31
Gammarus pulex	9	7	10	5	31
Ceratopogonidae	8	10	9	7	34
Centroptilum luteolum	16	8	11	1	36
Lymnaea peregra	11	13	6	18	48
Glossiphonia complanata	13	11	12	15	51
Pisidium nitidum	18	9	17	9	53
Caenis moesta/macrura	15	21	15	12	63
Erpobdella octoculata	12	17	13	23	65
Sphaerium corneum	17	14	23	13	67
Stylaria lacustris	19	15	16	17	67
Limnodrilus hoffmeisteri	14	18	14	22	68
Pisidium subtruncatum	22	16	19	16	73
Psammoryctides barbatus	21	19	20	14	74
Piscicola geometra	10	23	26	21	80
Procladius sp.	24	20	18	19	81
Asellus aquaticus	23	22	21	24	90
Bithynia tentaculata	26	24	22	20	92
Physa fontinalis	20	25	25	25	95
Sigara dorsalis	25	27	24	26	
Potamothrix hammoniensis	27	26	27	27	107

Tied taxa ordered by highest single ranking or, if still tied, second highest single ranking.

The position of species groups, such as _Hydracarina_ (water-mites), Ceratopogonidae (biting midges) and the Chironomidae groupings, _Thienemannimyia_ group, _Orthocladius/Cricotopus_, _Micropsectra/Tanytarsus_ and _Rheotanytarsus/Paratanytarsus_ (all non-biting midges), is more difficult to assess because of the number of species involved. The general indications are that representatives of these groups are amongst the more vulnerable of the listed taxa.

Six taxa have consistently low apparent vulnerability (Table 5.6.). These are _Potamothrix hammoniensis_ (a worm), _Bithynia tentaculata_ and _Physa fontinalis_ (snails), _Asellus aquaticus_ (water hog-louse), _Sigara dorsalis_ (a water boatman not yet recorded in the reach) and _Procladius_ sp. (a non-biting midge). They can be expected to be most resilient to any fall-off in water quality that might occur for whatever reasons.

5.7.5. Possible effects of the Maximum Resource Benefit Scheme

Prediction of ecological change is difficult because the nature and extent of the response to environmental modification is dependent upon not only the magnitude of the modification but also its rate and pattern (Smith, Brisbin & Weiner 1979). Included in the pattern of change will be the duration, amplitude and frequency of the extremes of the new conditions. The circumstances prevailing prior to the change are also important because the tolerance of aquatic macroinvertebrates can be increased by acclimation.

Acclimation is important if the rate of change is slow. Under these conditions macroinvertebrate communities can be expected to show considerable resilience to change in terms of species composition. Such changes as do occur are more likely to be in relative abundance, rates of growth or even concentration of pollutants in the bodies of organisms. However if changes of the same magnitude are implemented rapidly the effects are likely to be more profound and this should be borne in mind when manipulating the flow regime. Smith _et al._ (1979) write, "Of the three main characteristics of environmental change .... it is undoubtedly those characteristics of rate which are the most important in determining the exact nature and extent of consequential biological response." This is not to say that the magnitude or pattern of change may not ultimately prove decisive. Despite acclimation effects the tolerance of species is not infinite and ultimately the conditions may become so extreme that species will not survive. Long-term gradual deterioration or short-term or periodic extremes of high amplitude may prove equally decisive. The former category might include the increasing effect of ammoniacal input into the Thames from its tributaries as discharge in the main river slows or stops. In the latter category an example might be the alternately high and low diurnal oxygen concentrations following algal blooms.

Section 3 of this report postulates a series of responses to a set of resource utilisation programmes. In drought years little difference is envisaged between the Chart and the MRB regimes. Where differences do occur they are likely to be beneficial. This is because the MRB policy will probably produce slightly lower spring flows but at the gain of maintaining higher summer flows, thus reducing the likelihood of invoking drought orders. In turn this will reduce the amplitude of the changes experienced by the macroinvertebrate communities.

Set against this, any policy which reduces the discharge and velocity of the river is likely to result in more severe environmental conditions. DO concentrations will become more variable as retention time and water temperatures increase and algal growth accelerates. Similarly the concentrations of chemical variables will increase as the dilution effect diminishes.

This observation has special relevance in non-drought years when the proposed policy of removing as much water as possible in spring, combined with the estimated 17% increase in demand by the year 2006, may significantly reduce summer flows. The IH models indicate that under this practice summer BOD levels will be higher and DO concentrations lower than under the present rules. This effect will be felt most strongly in the reach above Molesey Weir where summer flows are generally lower anyway. Furthermore, the limited evidence available suggests that pollution-sensitive families may already be absent downstream of the Hogsmill at Kingston and this reach may also be vulnerable to low summer flows. Under the MRB tactics, summer conditions in these areas should be ameliorated in drought years but may be exacerbated in non-drought years.

Historically the Mole and Hogsmill were also responsible for high ammoniacal nitrogen input but improvements in effluent treatment facilities are thought to have largely overcome these problems. The conditions under which they may become critical are those in which low dilution in the Thames is combined with high algal growth. The taxa most likely to be vulnerable to deteriorating conditions have been outlined in the preceding sections.

5.7.6. Conclusions

The proposals for implementation of MRB tactics rather than the Chart are more likely to be beneficial than detrimental to macroinvertebrate communities in drought years. Nevertheless consideration should be given to avoiding sharp changes in flow conditions where gradual change would meet the same objectives. Macroinvertebrates are notably resilient to slow rates of change.

In non-drought years MRB operation may place more stress on the macroinvertebrate communities than would occur under the Chart, particularly if the predicted 17% extra demand by 2006 is realised. Therefore care should be taken in devising schedules for spring abstraction in non-drought years which mitigate against undue summer reductions in discharge.

The most sensitive areas are likely to be above Molesey Weir and downstream of the Hogsmill, Any problems in this area could be exacerbated by dense algal growths.

The macroinvertebrate taxa most likely to be vulnerable to deteriorating conditions have been indicated in 5.7.4. However, if as suggested, sharp changes in flow are avoided and non-drought year abstraction policy is carefully planned, little detrimental effect upon the macroinvertebrate communities of using MRB rather than the Chart system is envisaged.

5.8 INVERTEBRATE FAUNA OF THE TIDAL THAMES (R.M. Warwick and R. Williams)

The distance of penetration of marine species into estuaries, and of freshwater species into brackish water, has frequently been correlated with salinity tolerances (review by Carriker 1967). It is considered axiomatic that salinity is the 'master factor' governing estuarine distributions provided other conditions, such as oxygen concentration, remain favourable. Previous sections of this report have shown that measurable changes in environmental variables, resulting from proposed changes in the management system, are only found above London Bridge and, furthermore, that the oxygen levels are in no case adversely affected. This section of the report consequently only considers possible salinity effects between London Bridge and Teddington Lock.

Salinities between 5 and 7 parts per thousand constitute a significant ecophysiological boundary, referred to as the 'horohalinicum', at which the numbers of both marine and freshwater species reach a minimum (Remane & Schlieper 1958). Salinities above London Bridge are generally lower than this and the fauna is predominantly a freshwater one (Aston & Andrews 1978). However, marked fluctuations in the fauna occur at the region of the horohalinicum under present operating conditions, both seasonally and between years. Epibenthic species and zooplankton, particularly crustaceans such as copepods, amphipods, mysids, isopods, shrimps and prawns, migrate up and down the estuary with changing salinity and are largely unaffected by it: Andrews (1977) has documented the upstream migration of these organisms during the drought of 1976. Benthic infauna, for example molluscs and annelid worms, are frequently eliminated locally under conditions of increased salinity: Andrews (1977) reported the disappearance of several molluscs, leeches and tubificid worms from a station just above London

Bridge between 1975 and 1976, while Hunter (1981) recorded seasonal fluctuations in tubificids at Greenwich which he attributed to salinity changes between 1970 and 1972. Hunter also tabulated limiting salinities for three freshwater tubificid species in the Thames, based on both field and experimental evidence, which are generally close to those quoted above for the horohalinicum, although for Limnodrilus hoffmeisteri the limiting salinity is slightly higher (10-12 parts per thousand).

In addition to these published data, TWA biologists have monitored the distribution of some 91 invertebrate taxa at five sites in this region of the tideway (Teddington, Petersham, Kew, Battersea and London Bridge) regularly since 1974. This detailed information highlights the natural variability in the fauna from year to year, particularly toward the seaward end of this region.

Differences between the Chart and MRB systems of management are trivial when measured against the present variability in the system, where migration of epifauna, and elimination and replacement of the infauna are natural features of such dynamic regions. To reiterate earlier sections of this report, during the 1976 drought salinities at Teddington and Syon Park would not have been modified by more than 0.3 parts per thousand and would remain below 2.0 parts per thousand at all times, i.e. considerably below the critical level. Variability in salinity has, if anything, reduced. Similar equally trivial differences are evident in the simulations of other drought sequences, and in view of this it is not considered that the proposed changes in the operating system will have any detectable effect on the benthic fauna of the tidal Thames. It follows therefore that any changes in processes which relate to the activities of these benthic organisms (e.g. their consumption of algae and detritus, and predation on the zoobenthos by fish) will similarly be negligible.

5.9. FISH (R.H.K. Mann)

5.9.1. Introduction

This assessment has been drawn up in the absence of quantitative data on fish populations in the lower Thames (i.e. population densities, age-structures, growth rates etc.) and comparatively little qualitative information. The data pertaining to fish populations upstream of Teddington Weir were obtained from TWA biologists; they originate partly from the Authority's electrofishing surveys for young salmon (Salmo salar) and coarse fish (Banks 1979), and partly from their knowledge of the fish communities through, for example, anglers' catches. More data are available in the tidal reaches below Teddington Weir. Some are from the Authority's surveys, some from collections made at power station intakes (Wheeler 1979), and the rest from catches by commercial eel fishermen and from information from anglers (obtained from TWA).

The information from all these sources has been used in conjunction with predictions from the hydrological models to assess the likely changes in the fish populations in the lower Thames if the MRB pattern of water abstraction is adopted. Of necessity the assessments are subjective. Further, they are based on the model predictions that:

a) Water flows in the spring (before 1 May) will be on average, lower than before.

b) Summer flows will be, on average, higher than before.

c) No substantial changes will occur in water temperatures and oxygen content as a result of the change in the pattern of water abstraction.

5.9.2. Assessment

5.9.2.1. Salmon (*Salmo salar*)

Reduced flows during the spring will inhibit the upstream migration of adult salmon returning from the sea, especially in years when there has been little rainfall during the winter and early spring. This situation could be ameliorated by the provision of fish passes (as exists at Molesey Weir) through which the flow could be concentrated. No data exist on the seasonal pattern of adult salmon migration and it is thus impossible to determine the fraction of the run provided by the early spring salmon.

Immature salmon migrate to sea as smolts during March and April. Diversion of part of the river flow into storage reservoirs, during this part of the year, will inevitably lead to movement of smolts into areas from which return to the river is difficult or impossible. The loss of smolts, and hence the numbers of returning adults in later years, is difficult to gauge accurately. However, the loss is certain to be more acute when river levels are generally low and the percentage of the flow that is abstracted is correspondingly high (given an approximately uniform level of abstraction year-by-year).

Some spawning of salmon occurs in the lower Thames as, for example, at Sunbury where the R. Ash enters the main river. Low flows during early spring may cause some silt deposition in areas such as this, and this would increase the mortality rate of eggs and fry through oxygen depletion. Insufficient data are available to assess the importance of these lower spawning areas in relation to the overall spawning regime of the river.

5.9.2.2. Non-salmonid species in non-tidal waters

In most years, the change in pattern of water abstraction should have no serious implications for the non-salmonid species. An exception to this would occur in dry winter/spring periods when the eggs of lithophilous spawners (e.g. dace Leuciscus leuciscus) could be killed through siltation, or even exposure, of the gravel spawning areas. It is not envisaged that the overall increase in summer flows would have any substantial effect on the non-salmonid fish community.

However, predictions of discharge and DO levels from the IH model, including those resulting from a 17% increase in water demand by the year 2006, do not indicate an exacerbation of these problems for either salmonid or non-salmonid fishes, except for a small increase in BOD levels. It should be remembered that the levels of DO and BOD are, to a large extent, dependent upon the quality of effluents from the various sewage treatment works. The IH model assumes that the quality of effluents will be maintained even though water usage will be increased by 17%.

5.9.2.3. Non-salmonid species in tidal waters

Water abstraction above Teddington Weir during dry spring periods will increase the upstream incursion of salt water, especially if the abstractions coincide with high spring tides. The freshwater fishes are confined to the area above London Bridge and any prolonged restriction of their populations to the upper part of this area could affect their gonad development in the critical period before they spawn. Early spawners, such as the dace, may even have their spawning activity affected. However, this did not occur during the 1975-76 drought, and the IMER hydrological model does not predict any worsening of the problem under the MRB procedure.

This latter effect is likely to be most serious for the smelt (Osmerus eperlanus) which spawns in the March to May period in freshwater zones immediately upstream of saltwater estuaries. The smelt once supported a major fishery in the River Thames and is now increasing in number following a long period of decline. In 1983, over 8000 were caught by TWA in the survey below Tower Bridge, a number exceeded only by the eel (34,000) and flounder (13,000).

The IMER hydrological model suggests little or no change in the effects of very low flows as a result of the MRB policy, and it shows that water upstream of the half-tide lock at Richmond rarely becomes saline.

5.10 SUMMARY

5.10.1. The scarcity of good information about the flora and fauna of the lower River Thames means that it is difficult to make accurate predictions about the likely effects of small changes in environmental conditions.

5.10.2. Much of the physical and chemical data is in the form of mean values based on observations made during normal working hours. Living organisms are more likely to be seriously affected by extreme values which are concealed within mean values and which may be more likely to occur outside normal working hours, notably in the case of dissolved oxygen.

5.10.3. All the biological predictions are based on current levels of demand for water or a 17% increase in demand which has been predicted by TWA for the year 2006. They should not be considered applicable if the demand increases beyond this level.

5.10.4. The biochemical oxygen demand and the levels of ammonia have sometimes been sufficiently high and the levels of dissolved oxygen have sometimes been sufficiently low to cause concern about the environmental conditions in the river around Molesey and Teddington. The models indicate that some deterioration would occur, particularly in dissolved oxygen, under the proposed changes in abstraction but that these are unlikely to be large. However, nocturnal levels of dissolved oxygen will be lower than those used in the models. The predicted deterioration may be more than offset by improvements in the quality of effluents discharged into the river in this area.

5.10.5. The overall effect on the bacteriology of the river is likely to be negligible.

5.10.6. Reduced flows in spring and early summer could lead to increased sedimentation of suspended particles which would allow increased light penetration and this could give rise to larger growths of filamentous algae. Enteromorpha forms floating growths in some backwaters and this could invade the main river. However, there are no records of substantial changes during the extremely low flows experience in 1976 and it seems unlikely that any should occur under the proposed changes.

5.10.7. The River Thames develops large growths of planktonic algae especially in spring and early summer. The extent of the growth is determined mainly by light and turbidity. Reduced flows may affect the timing and duration of the peak periods but should not affect the quantity of species composition. Changes in summer, perhaps favouring greater quantities of bloom-forming cyanobacteria cannot be discounted. Some growth of <u>Microcystis</u> was recorded in 1976 but this did not become a serious ecological problem then and it seems unlikely to do so under the proposed changes.

5.10.8. Changes in higher plants could occur if there are long periods with little or no flow. A floating cover of duckweed is probably the most undesirable change that can be envisaged. Although there was apparently some development of duckweed in backwaters during 1976, the plant did not invade the main river and it seems unlikely to do so under the proposed changes. Wind action and disturbance by boats both inhibit the spread of duckweed.

5.10.9. The factors most likely to affect the invertebrate fauna of the non-tidal river are sudden changes in flow conditions and long periods of very low flows in summer. Provided these are avoided it seems likely that the proposed changes will have little effect on invertebrates. During drought years, environmental conditions for invertebrates are more likely to improve than to deteriorate under the proposed changes. The reach most sensitive to sustained low flow is between the Hogsmill and Teddington Weir.

5.10.10. Salinity is the 'master factor' governing the distribution of invertebrates in the estuary and salinities between 5 and 7 parts per thousand constitute a significant ecological boundary. Marked fluctuations in the invertebrate fauna occur in this region under current operating conditions, both seasonally and between years, and very low flows would enhance the magnitude of such fluctuations.

5.10.11. Reduction in flow due to increased abstraction in spring could affect fish in several ways. There would be less flow to stimulate the upstream migration of salmon. Silt deposition would be increased and this could affect the eggs of fish such as salmon and dace which spawn on the bed of the river. If salmon become re-established in the river there is a danger of smolts being diverted into the reservoirs during their downstream migration. The models indicate that the changes in flow will be small and any effects are consequently unlikely to be significant.

5.10.12. The spawning areas of dace and smelt in the upper part of the estuary could be restricted if decreased flows in spring result in salt water coming further up the estuary. The models indicate that such changes are not likely to be great.

5.11 REFERENCES

Allen, W.E. (1913) A quantitative and statistical study of the plankton of the San Joaquin River and its tributaries in and near Stockton, California. Univ. Calif. Publs in Zool. 22, 1-292

Andrews, M. (1977) Observations on the fauna of the metropolitan river Thames during the drought in 1976. Lond. Nat. 56, 44-56

Armitage, P.D., Moss, D., Wright, J.F. & Furse, M.T. (1983) The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running-water sites. Wat. Res. 17, 333 -347

Aston, K.F.A. & Andrews, M.J. (1978) Freshwater macroinvertebrates in London's rivers, 1970-77. Lond. Nat. 57, 34-52

Baker, H. (1939) Personal communication of unpublished memorandum on aquatic vegetation of the Thames near Oxford. In The British Islands and their vegetation (editor A.G. Tansley), 626-628, Cambridge University Press, Cambridge.

Baker, J.H. & Farr, I.S. (1977) Origins, characterization and dynamics of suspended bacteria in two chalk streams. Arch. Hydrobiol. 80, 308-326

Banks, J.W. (1979) River Thames fish surveys. Proc. 1st Br. freshwater Fish Conf. 336-344

Berrie, A.D. (1972a) Productivity of the River Thames at Reading. Symp. zool. Soc. Lond. 29, 69-86

Berrie A.D. (1972b) The occurrence and composition of seston in the River Thames and the role of detritus as an energy source. Mem. Ist. ital. Idrobiol. (Suppl.) 29, 473-483

Bolas, P.M. & Lund, J.W.G. (1974) Some factors affecting the growth of Cladophora glomerata in the Kentish Stour. Wat. Treat. Exam. 23, 25-51

Bottrell H.H. (1975a) The relationship between temperature and duration of egg development in some epiphytic Cladocera and Copepoda from the River Thames, Reading, with a discussion of temperature function. Oecologia (Berl/) 18, 63-84

Bottrell H.H. (1975b) Generation time, length of life, instar duration and frequency of moulting and their relationship to temperature in eight species of Cladocera from the River Thames, Reading.. Oecologia (Berl.) 19, 129-139

Bowles, B. & Quennell, S. (1971) Some quantitative algal studies of the River Thames. Wat. Treat. Exam. 20, 35-51

Butcher, R.W. (1924) The plankton of the River Wharfe. The Naturalist, Hull April-June 1924, 175-214

Butcher, R.W. Longwell, J & Pentelow F.T.K. (1937) Survey of the River Tees. Part 3: The non-tidal reaches - chemical and biological. Wat. Pollut. Res. Tech. Pap. No 6, 1-189

Carriker, M.R. (1967) Ecology of estuarine benthic invertebrates, a perspective. In Estuaries (editor G.H. Lauff), 442-487. American Association for Advancement of Science, Washington DC.

Chandler J.F. (1970) A biological approach to water quality management. Wat. Pollut. Control 69, 415-422

Chesters, R.K. (1980) Biological Monitoring Working Party. The 1978 national testing exercise. Department of the Environment, Water Data Unit Technical Memorandum 19, 1-37

Collie, A. & Lund, J.W.G. (1980) Bioassays of waters from the Severn and Thames river systems. J. Instn Wat. Engrs Sci. 34, 180-188.

Doxat, J. (1977) The Living Thames, 96, London, Hutchinson Benman.

Evans, J.H. (1971) The use of a model A Coulter counter in quantitative determination of the phytoplankton of the River Thames. Proc. Soc. analyt. Chem. 8, 260-264

Evans, J.H. & McGill, S.M. (1970) An investigation of the Coulter Counter in biomass determinations of natural freshwater phytoplankton populations. Hydrobiologia 35, 401-419

Fritsch, F.E. (1902) Algological notes. III. Preliminary report on the phytoplankton of the Thames. Ann. Bot. 16, 576-584

Fritsch, F.E. (1903) Further observations on the phytoplankton of River Thames. Ann. Bot. 17, 631-647

Furse, M.T. (1978) An ecological survey of the middle reaches of the River Thames. Report to Thames Water Authority, FBA River Laboratory, East Stoke, Wareham, Dorset.2 vols.

Furse, M.T. Moss, D., Wright, J.F. and Armitage, P.D. (1981) An appraisal of pond-net samples for biological monitoring of lotic macro-invertebrates Wat. Res. 15, 679-689

Gannon, J.J. Busse, M.K. & Schillinger, J.E. (1983) Faecal coliform disappearance in a river impoundment. Wat. Res. 17, 1595-1601

Gessner, F. (1955) Hydrobotanik. Bd I. Energiehaushalt. Veb. Deutscher Verlag der Wissenschaften, Berlin.

Greenberg, A.E. (1964) Plankton of the Sacramento River. Ecology 45, 40-49

Haslam, S.M. (1978) River Plants, 396 pp, Cambridge Univ. Press, Cambridge.

Haslam, S.M. & Wolseley, P.A. (1981) River Vegetation: Its Identification, Assessment & Management, 154pp, Cambridge Univ. Press. Cambridge.

Hassall, A.H. (1850) A microscopic examination of the water supplied to the inhabitants of London and the surburban districts. Samuel Highley, London.

Houghton, G.U. (1972) Long-term increases in planktonic growth in the Essex Stour. Wat. Treat. Exam. 21, 299-308

Hoogenhout, H. & Amesz, J. (1965) Growth rates of photo synthetic microorganisms in laboratory cultures. Arch. Mikrobiol. 50, 10-25

Hunter, J. (1981) Survival strategies of tubificids in the Thames and other estuaries. In Feeding and survival strategies of estuarine organisms

Hynes, H.B.N. (1970) The ecology of running waters. Univ. of Liverpool Press, Liverpool.

Kowalczewski, A & Lack, T.J. (1971) Primary production and respiration of the phytoplankton of the rivers Thames and Kennet at Reading. Freshwat. Biol. 1, 197-212

Lack, T.J. (1969) Changes in the phytoplankton population of the River Thames at Reading (Berkshire). J. Ecol. 57, 29P-30P

Lack, T.J. (1971) Quantitative studies on the phytoplankton of the Rivers Thames and Kennet at Reading. Freshwat. Biol. 1, 213-224

Lack, T.J. & Berrie, A.D. (1976) Phytoplankton production in the rivers Thames and Kennet at Reading during 1970. In Light as an ecological factor: II (editors G.C. Evans, R. Bainbridge and O. Rackham) 43-62. Blackwell, Oxford.

Lack, T.J., Youngman, R.E. & Collingwood, R.W. (1978) Observations on a spring diatom bloom in the River Thames. Verh. int. Verein. theor. angew. Limnol. 20, 1435-1439

Mackey, A.P. (1972) An air-lift for sampling freshwater benthos. Oikos 23, 413-415

Mackey, A.P. (1976a) Quantitative studies on the Chironomidae (Diptera) of the Rivers Thames and Kennet. I The Acorus zone. Arch. Hydrobiol. 78, 240-267

Mackey, A.P. (1976b) Quantitative studies on the Chironomidae (Diptera) of the Rivers Thames and Kennet. II The Thames flint zone. Arch. Hydrobiol. 78, 310-318

Mackey, A.P. (1977a) Quantitative studies on the Chironomidae (Diptera) of the Rivers Thames and Kennet. III The Nuphar zone. Arch. Hydrobiol. 79, 62-102

Mackey, A.P. (1977b) Species diversity of some larval Chironomidae. Oikos 28, 270-275

Mann, K.H. (1964) The pattern of energy flow in the fish and invertebrate fauna of the River Thames. Verh. int. Verein. theor. angew. Limnol. 15, 485-495

Mann, K.H. (1972) Case history of the River Thames. In River Ecology and Man (editors R.T. Oglesby, C.A. Carlson & J.A. McCann) 215-232. New York, Academic Press.

Mann, K.H., Britton, R.H., Kowalczewski, A., Lack, T.J., Mathews, C.P. & McDonald, I. (1972) Productivity and energy flow at all trophic levels in the River Thames, England. In Productivity Problems of Freshwaters. Warszawa, Polish Scientific Publications.

Maker, A.F.H. & Casey, H. (1982) The population and production dynamics of benthic algae in an artificial recirculating hard-water stream. Phil. Trans. R. Soc. Lond. (B) 298, 265-308

McGill, S.M. (1969) The ecology of the phytoplankton of a new reservoir in the Thames Valley. PhD Thesis, Univ. of London.

Pitcairn, C.E.R. & Hawkes, H.A. (1973) The role of phosphorus in the growth of Cladophora. Wat. Res. 7, 159-171

Remane, A. & Schlieper, C. (1958) The biology of brackish waters, 372pp. Wiley-Interscience, New York.

Reynolds, C.S. (1984a) The ecology of freshwater phytoplankton. Cambridge Univ. Press, Cambridge

Reynolds, C.S. (1984b) Phytoplankton periodicity: the interaction of form, function and environmental variability. Freshwat. Biol. 14, 111-142

Rice, C.H. (1938) Studies in the phytoplankton of the River Thames (1928-1932) 1 and 2 Ann. Bot. (New Series) 2, 539-557, 559-581

Smith, I. & Lyle, A. (1979) Distribution of freshwaters in Britain. Institute of Terrestrial Ecology, Cambridge, England.

Smith, M.H., Brisbin, I.L. & Weiner, J.G. (1979) Some basic principles concerning biological response to environmental change. In Selection, Management Utilization of Biosphere Reserves (editors J.F. Franklin & S.L. Kaufman), 105-122. USDA Forest Service, Pacific North West Forest and Range Environmental Station, Corvallis, Oregon.

Swale, E.M.F. (1963) Notes on Stephanodiscus hantzschii Grun. in culture. Arch. Mikrobiol. 45, 210-216

Swale, E.M.F. (1964) A study of the phytoplankton of a calcareous river. J. Ecol. 52, 443-446

Swale, E.M.F. (1969) Phytoplankton in two English rivers. J. Ecol. 57, 1-23

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

Tittley, I & Price, J.H. (1977) An atlas of the seaweeds of Kent. The Kent Field Club 7, 1-80

Vannote, R.L. Minshall, G.W., Cummins, K.W. Sedell, J.R. & Cushing, G.E. (1980) The river continuum concept. Can. J. Fish. Aq. Sci. 37, 130-137

Wawrik, F. (1964) Zur Frage : Fuhrt der Donaustrom autochthones Plankton Arch. Hydrobiol. Suppl. 27, 28-35.

Westlake, D.F. (1975) Macrophytes. In River Ecology (editor B.A. Whitton), 106-128. Blackwell Sci. Publs, Oxford.

Wheeler, A. (1979) The Tidal Thames. 228pp. Routledge & Kegan Paul Ltd, London.

Whitehead, P.G. & Hornberger, G.M. (1982) Modelling algal behaviour in river systems. In Environment systems analysis and management (editor S. Rinaldi). North-Holland Publ. Co. IFIP

Whitehead, P.G. & Hornberger, G.M. (1984) Modelling algal behaviour in the River Thames. Wat. Res. 18, 945-953

Whitton, B.A. (1970) Biology of Cladophora in freshwaters. Wat. Res. 4, 457-476

Whitton, B.A. (1975) River Ecology. Blackwell, Oxford.

Williams, L.G. (1964) Possible relationships between plankton-diatom species numbers and water-quality estimates. Ecology 45, 809-823

Williams L.G. & Scott, C. (1964) Principal diatoms of major waterways of the United States. Limnol. Oceanogr. 7, 365-379

Wong, S.L. & Clark, B (1976) Field determination of critical nutrient concentrations for Cladophora in streams. J. Fish. Res. Bd Can. 33, 85-92

Woodiwiss, F.S. (1964) The biological system of stream classification used by the Trent River Board. Chem. Ind. 11, 443-447

Wright, J.F., Moss, D., Armitage, P.D. & Furse, M.T. (1984) A preliminary classification of running-water sites in Great Britain based on macro-invertebrate species and the prediction of community type using environmental data. Freshwat. Biol. 14, 221-256

Youngman, R.E. (1971) In Discussion of Bowles & Quennell (1971, q.v.)

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