NATIONAL RIVERS AUTHORITY
THAMES REGION

MAIDENHEAD, WINDSOR and ETON
FLOOD ALLEVIATION SCHEME
WATER QUALITY STUDY

FINAL REPORT
APPENDICES

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Appendix 1

Draft brief

NATIONAL RIVERS AUTHORITY - THAMES REGION

MAIDENHEAD, WINDSOR AND ETON FLOOD ALLEVIATION SCHEME

PREQUALIFICATION OF WATER QUALITY CONSULTANT

DRAFT BRIEF

Objective of Study

The construction of a flood relief channel for Maidenhead, Windsor and Eton poses a potential for changes of water quality in the River Thames. The channel will also contain water at all times and an assessment is required of the likely quality which will arise. A particular concern is the development of algal blooms in the new channel and in the River Thames. Further studies are required as a matter of urgency to determine these effects.

Furthermore the proposals need to be studied in the light of impending statutory water quality objectives. These WQO's will be "use-based" and, for each use, there will be a set of numeric standards. The statutory system is under development and standards will evolve from the existing RQO system and include common EQS values. This will require the NRA to decide what range of uses or options for uses the channel will fulfil and determine how they will be met. These aspects need to be considered now so that appropriate plans can be made to meet them and to avoid an unacceptable quality.

Terms of Reference

1. To predict the effects of the development on the aquatic environment in three distinct river sections:

   (i) the current River Thames channel between inlet and outlet of the flood relief channel, noting that several discharges are made to this section, either directly or indirectly, and that abstractions are taken, either directly or indirectly, from this section. The impact (quantity and quality) on the discharges and abstractions are also required;
   (ii) the flood relief channel (FRC)
   (iii) the River Thames downstream of the FRC

2. The effects are to be predicted assuming the following flow regimes:

   (i) flow is only diverted down the FRC during flood conditions;
   (ii) Thames flow is split to ensure equal velocities down the main channel and the FRC at times of average and low flow.

In order to establish 5 below a number of intermediate strategies should also be considered.

3. The effects are to be predicted assuming that the FRC is:

   (i) unlined
   (ii) lined in the vicinity of Slough STW.

4. The environmental effects to be considered are:

   (i) seasonal and diurnal variations in water quality, particularly in biochemical oxygen demand, ammonia, dissolved oxygen, un-ionised ammonia, phosphate and nitrate
   (ii) macrophytic growth (emergent, submerged, floating)
   (iii) algal blooms (species and numbers)
   (iv) the effect of the above on bird life (eg botulism)
   (v) the effect on fish communities

5. To determine the optimum operating regime so that the impact on existing surface water abstractions and discharges is minimised and that the quality in the channel is maximised. The major abstractions in the area are those of the water companies in the Lower Thames and a proposed abstraction at Bray. Major discharges in the area include a number of sewage treatment works discharging to The Cut and Slough STW discharging to a small north bank tributary.

6. To evaluate the effects of the proposed channel cross-sectional shape on the aquatic environment.

Method

Existing models should be used and extended where possible. Any model must be agreed with the NRA.

A model has been produced by RPS Environmental Sciences Ltd for predicting groundwater levels, quality and flows. This must be used to predict inputs to the new channel from groundwater. A dynamic flow and water quality model exists for the main stem of the River Thames. This could be extended to include the FRC and incorporate the effects of groundwater flows into the channel as predicted by the RPS model.

Data Availability

River flows, water quality, biological and fisheries data as currently exist will be made available by the NRA. Survey data are available for the Thames and specific cross-sections are available for the new channel.

It should be assumed that there is no need for substantial data collection to be undertaken.

Appendix 2 Surface water flow and quality model (QUASAR)

1 INTRODUCTION

1.1 Background

The model QUASAR (Quality Simulation Along Rivers) has been developed at the Institute of Hydrology to assess the environmental impact of pollutants on river water quality. The model has evolved over a number of years during which time there have been many applications to rivers in the UK and overseas. The model was originally developed as part of the Bedford Ouse Study with the primary objective of simulating the dynamic behaviour of flow and water quality along the river system (Whitehead et al, 1979, 1981). Initial applications involved the use of the model within a real time forecasting scheme collating telemetered data and providing forecasts at key abstraction sites along the river (Whitehead, 1984). The model was also used within a stochastic or Monte Carlo framework to provide information on the distribution of water quality within river systems, particularly in rivers subjected to major effluent discharges (Whitehead and Young, 1979). This technique was later adapted by Warn (1982) to assess mass balance problems within river systems. There has also been a range of model applications to other UK rivers such as the River Tawe to assess heavy metal pollution and the River Thames, to assess the movement and distribution of nitrates and algae along this river system (Whitehead and Williams, 1982, Whitehead and Hornberger, 1984).

QUASAR (**QUAlity Simulation Along Rivers**) is a water quality and flow model. The model has been developed to combine upstream inputs due to accidental, man made and natural inputs. Forecasting and planning information is generated for key locations along the river. The water quality parameters modelled are nitrate, dissolved oxygen (DO), biochemical oxygen demand (BOD), ammonia, ammonium ion, temperature, ortho-phosphate, pH, and a "conservative" water quality parameter. To model these parameters the river is divided up into reaches. The reach boundaries are determined by points in the river where there is a change in the water quality or flow due to the confluence with a tributary, the location of a sewage treatment final effluent discharge, abstraction, or location of weirs. Water quality changes due to biological or chemical reactions are also considered by ensuring appropriate reach lengths.

Two sets of equations have been developed to represent flow and the nine water quality parameters. One set consists of the differential equations relating the rate of change of these parameters with time. These equations are solved by a "differential equation solver" subroutine in the program. The other set consists of "analytical solutions" or the integrated differential equations. These equations are solved at discrete time intervals, specified in the program as the model time step. The first decision to be made in using **QUASAR** is whether planning or forecasting information is required.

1.2 Planning Mode

In the stochastic or planning mode a cumulative frequency curve and distribution histogram of a water quality parameter are generated by repeatedly running the model using different input data selected according to probability distributions defined for each input variable. Whitehead and

Young (1979) and Warn (1982) have used this technique, known as Monte Carlo simulation, to provide information which aids in long term planning of water quality management. In this mode statistical data of the water quality and flow in the first reach at the top of the river, and in tributaries, STW discharges, and abstractions at key locations along the river are required. These data include, for each variable input to the model, the mean, standard deviation, and shape that the probability curve takes ie. lognormal, rectangular, or gaussian. Random numbers are generated as water quality and flow values are chosen from these characterized distributions. A mass balance is performed at the top of each reach to include tributaries, discharges, abstractions and any other inputs to the river at that point on the river for each run of the model. The values generated by the model equations represent the water quality or flow at the end of the reach. The model equations are run using the random numbers as the input values either until steady state has been reached or for a maximum of 30 time periods. Steady state is said to have been achieved when the results of successive runs differ by less than 1%. Five hundred and twelve random numbers are generated. The output is stored and used to produce cumulative frequency distributions and distribution histograms.

1.3 Dynamic Mode

In the forecasting or dynamic mode, the water quality and flow are simulated over selected periods. This allows the possible affects of a pollution event on a river to be investigated. In this mode time series data are required for water quality and flow parameters for the first reach of the river and for tributaries, STW discharges and abstractions along the section of the river of interest. The model run time step, ie. the time interval over which the model will dynamically compute river quality and flow, and the run output length, ie. the number of output steps that the model runs for, must also be specified. Once these data have been input the model can be run. A mass balance is performed at the beginning of each reach for inputs such as tributaries entering at that point on the river. The model input then goes to the differential or analytical equations and the output from each reach is stored and used as the input of the next reach. The model is run for 40 time periods before the specified start of the model run using the "default" values to ensure that the system has reached equilibrium. The output values are used in generating profiles of water quality parameters along the river at a given time or in generating time series data at a specified location.

2 Description of the QUASAR Model Equations

Nine water quality parameters and flow are modelled. In the following sub-sections a summary of the differential equations is given listing the major processes occurring. A detailed explanation of the processes and the assumptions made in the equations is then given. Analytical solutions (ie integrated differential equations) are given in Appendix A.

2.1 Flow

The flow in the river is represented by:

$$\frac{dX_t}{dt} = \frac{U_t - X_t}{[1-b] \cdot TC}$$

In this differential equation, X_1 refers to the downstream flow (reach output) and U_1 refers to the upstream flow (reach input). TC is the reach residence time, often referred to as travel time, which varies as a function of flow, and b is a constant defined below.

2.1.1 Development of Equation

As mentioned previously, the river has been divided into reaches. The boundaries of these reaches are located at the confluence of tributaries, weirs, effluents, abstractions, or at other locations where changes in the water quality occur. Each reach is further divided into cells. Flow variation in each cell is analogous to the variation in concentration of a conservative pollutant under the assumption of uniform mixing over the cell. The concentration of a conservative pollutant is described by the lumped parameter equations (Whitehead et. al., 1979, 1981, 1984).

We know that, in all cases:

(i) $V = TC \cdot Q$

(ii) $TC = \dfrac{l}{v \cdot N}$ and

(iii) $\dfrac{dV}{dt} = U_1 - X_1$ (mass balance),

where:
V is the volume of the ~~reach~~, lag
TC is the time taken for water to travel down the ~~river~~, reach lag
Q is the average flow in the ~~reach~~, lag
l is the length of the reach, ✓
v is the average velocity of the water in the reach, ✓
N is the number of lags (divisions within the reach) and
X1 and U1 are ~~as above~~. flows into & out of lag

provided that we are dealing with the continuous case (as we shall be doing). Then, if we assume that the reach is a stirred tank system (which this model does assume) we have:

$X_1 = Q$

Also we have the empirical relationship:

$v = a + b \cdot Q^c$

which is obtained from measuring both v and Q; a, b and c are different constants for each reach; a is almost always zero.

So, we may now derive the equation:

$$\frac{dV}{dt} = d\frac{(TC \cdot Q)}{dt} = Q \cdot \frac{dTC}{dt} + TC \cdot \frac{dQ}{dt}$$

by the chain rule

But

$$\frac{dTC}{dt} = d\frac{\left(\frac{l}{Nv}\right)}{dt} = -\frac{l}{Nv^2} \cdot \frac{dv}{dt}$$

$$= \frac{-TC}{b \cdot Q^c} \cdot b \cdot \frac{dQ^c}{dt} = \frac{-TC}{Q^c} \cdot \frac{dQ^c}{dQ} \cdot \frac{dQ}{dt} = \frac{-TC}{Q^c} \cdot c \cdot Q^{c-1} \frac{dQ}{dt} = \frac{-TC \cdot c}{Q} \cdot \frac{dQ}{dt}$$

So

$$\frac{dV}{dt} = TC \cdot \frac{dQ}{dt} - c \cdot TC \cdot \frac{dQ}{dt} = [1 - c] \cdot TC \cdot \frac{dQ}{dt} = U_1 - X_1$$

If

$c = 1$

then this reduces to

$U_1 = X_1$

which is the case for regulated rivers where the water level is kept constant.

If

$c \neq 1$

then we have:

$$(1-c)\quad \frac{dX_1}{dt} = \frac{U_1 - X_1}{\cancel{(1-c)} \cdot TC} \qquad (1-c)\frac{dx}{dt} = U_1 - x$$

as

$X_1 = Q$

from the stirred tank assumption.

The values of N affect the relative importance of floodwave advection and dispersion in a reach; values of N, a, b and c can be determined by calibration on an observed record of downstream flow or from tracer experiments (see Whitehead et. al.,1984).

This then is the continuous solution which is solved by a numerical differential equation solver.

2.2 Nitrate

Two processes affect the rate at which the nitrate concentration changes in the water column. These are nitrification and denitrification. The differential equation describing the rate of change of nitrate concentration with time is given below:

If $c \neq 1$

then

$$\frac{d(X_2)}{dt} = \frac{U_2 - X_2}{TC \cdot \cancel{(1-c)}}$$

$- K_5 \cdot X_2$ denitrification

$+ K_{15} \cdot X_6$ nitrification

where U_2 and X_2 are the input and output nitrate concentrations and K_5 and K_{15} are the rate coefficients associated with the processes indicated. X_6 is the ammonia concentration.

If $c = 1$

then

$$X_2 = U_2 - K_5 \cdot X_2 \cdot TC + K_{15} \cdot X_6 \cdot TC$$

Note that if the Dissolved Oxygen level goes to zero, then the terms involving K_{15} and K_5 are left out.

$c=1 \Rightarrow U_2 = X_2$ unless $(1-c)\frac{dX_2}{dt} = \frac{U_2 - X_2}{TC} - K_5X_2 + K_{15}X_6$

$N^+_4 + 2O_2 + 2H_2O \rightarrow$

$NO_3^- + H_2O + 2H$

2.2.1 Nitrification

Nitrification is the process resulting in the conversion of ammonium to nitrite and then to nitrate. The two biochemical reactions are shown below.

$NH_4^+ + 2\ O_2$ ---(nitrosomas bacteria)---> $NO_2^- + 2\ H_2O$
$NO_2^- + 2\ H_2O$ ---(nitrobacter bacteria)--> $NO_3^- + H_2O + 2H^+$

Curtis, Durrant, and Harman (1974) studied nitrification in rivers in the Trent Basin and found growth rates for nitrosomas and nitrobacter were virtually the same. Laboratory work by Alexander (1965) showed nitrobacter was five times as efficient as nitrosomas in transforming nitrite and ammonium respectively. This indicates that the ammonia (ammonium ion) concentration is the rate controlling process. Knowles and Wakeford (1978) modelled the change in nitrate concentration to be dependant on the temperature, ammonia and nitrosomas concentration. In **QUASAR** the rate of change of nitrate concentration is dependant on the concentration of ammonia, the temperature °C, and the ammonia nitrification rate, K_{15}, which is usually in the range of 0.01 to 0.5 $days^{-1}$. The value for the ammonia nitrification rate can be edited by the user. The equation is given below where T is the temperature in Celsius and K_{15} is the nitrification rate in $days^{-1}$.

nitrification = $K_{15} \cdot 10^{(T \cdot 0.0293)}$ ($days^{-1}$)

2.2.2 Denitrification

In denitrification, nitrate is reduced to nitrogen gas and oxygen by denitrifying bacteria. The simplified reaction is given below:

$2\ NO_3$ ----> $3\ O_2 + N_2\ (g)$

The oxygen produced is consumed by the bacteria as an oxygen source so does not add to the oxygen concentration in the river. Toms et al., (1975) studied the factors affecting the denitrification process. These researchers found that the process is first order and proportional to the nitrate concentration, and required the presence of mud. They also found that for every 10 °C increase in temperature the rate of denitrification increased by a factor of 1.9 which can be described in the equation as $10^{T \cdot 0.0293}$.
The relationship they developed is:

NO

$$\frac{dNO_3}{dt} = -K \cdot 10^{(0.0293T \cdot 0.0294)} \cdot A \cdot CN$$

where A (m^2) is the surface area of mud in contact with water, CN is the concentration of nitrate in water in mg l^{-1}, T is the temperature in Celsius. K is a value in the range of 0.29 (clean gravel type bed), to 3.0 (soft muddy bed supporting denitrifying bacteria). In **QUASAR** modelling of denitrification is based on this work. The equation is given below:

denitrification = $K_5 \cdot 1.0698 \cdot 10^{(T \cdot 0.0293)}$ no it isn't

Note that 1.0698 is calculated from $10^{0.0294}$. K_5 is in units of day^{-1} and in the range of 0.0 to 0.5. The value for K_5 can be edited by the user.

2.3 Conservative

A conservative water quality parameter has been included in the model to describe any conservative determinand, for example chloride. This can be used to get a worst case estimate when modelling a variable not included explicitly in **QUASAR**. U3 and X3, the input and output conservative water quality parameter concentrations, are related by the equation:

If $c \neq 1$

then

$$\frac{d(X_3)}{dt} = \frac{U_3 - X_3}{TC \cdot (1 - c)}$$

If $c = 1$

then

$$X_3 = U_3$$

2.4 Dissolved Oxygen

The change in dissolved oxygen concentration is modelled as a result of photosynthetic O_2 production, benthic oxygen demand, reaeration (natural or due to the presence of a weir), nitrification, and loss due to BOD. The differential equation is given below:

If $c \neq 1$

then

(1-c)

$$\frac{d(X_4)}{dt} = \frac{U_4 - X_4 + WEIR}{TC \cdot (1-c)}$$

$+K_{11}$	net algae O_2 contribution
$-K_4 . K_6 . X_4$	benthic oxygen demand
$+K_2 . (CS - X_4)$	reaeration
$-4.43 . 10^{(T \cdot 0.0293)} . K_{15} . X_6$	nitrification
4.57 $-K_1 . X_5$	loss due to BOD

If $c = 1$

then

4.57

$$X_4 = U_4 + WEIR + [K_{11} - K_4 \cdot K_6 \cdot X_4 + K_2(CS - X_4) - 4.43 \cdot K_{15} \cdot X_6 - K_1 \cdot X_5] \cdot TC$$

where U_4 and X_4 are the input and output dissolved oxygen concentrations and Ki are the rate coefficients associated with the processes indicated. X_5 and X_6 are the BOD and ammonia concentrations respectively and WEIR is the contribution or loss of oxygen due to the presence of a weir in the reach.

2.4.1 Reaeration at Weirs

The contribution or loss of dissolved oxygen due to the presence of a weir in a river is described by the equation, (DOE, 1973).

$$X_4 = CS - \frac{(CS - XO_4)}{RT}$$

where CS is the oxygen saturation concentration, XO_4 is the dissolved oxygen above the weir and RT is the deficit ratio. The DO deficit ratio takes into account the type of weir using a factor B, the pollution of the water (percent saturation), A, the height from the top of the weir to the downstream water level (m), H, and the temperature, T (° C) of the water as shown in the equation below.

what is η !

$RT = 1 + 0.38ABH(1 - 0.11H)(1 + 0.46T)$

There are 4 types of weirs; free, slope, step, and cascade. A free weir or normal weir takes a value of unity for B. A step weir has a value of 1.3 for B. A cascade weir consists of a large number of steps with a value for B of 0.4 and a sloping weir has a sloping face down with a value for B of 0.2. The equation is given below,

2.4.2 Algae Contribution to Dissolved Oxygen

Algae, aquatic plants and phytoplankton utilize water, carbon dioxide, and sunlight to photosynthesize simple sugar and oxygen which is released to the water column. Respiration, which depletes the dissolved oxygen store in the water, occurs throughout the day. These two processes result in the highest dissolved oxygen concentration at midafternoon and the lowest concentration during the early hours of the morning. The two processes are described below and related in the differential equation by K_{11}=P-R where P represents photosynthetic oxygen production and R represents respiration.

2.4.2.1 Photosynthetic Oxygen Production

Photosynthetic oxygen production in river systems has been described by Owens et. al., (1969) in which oxygen production is related to the light intensity and plant biomass or algal levels. They found that once there is sufficient plant biomass to provide adequate and uniform cover of the river bed the plant biomass has apparently no affect on the rate of photosynthesis due to self-shading. Whitehead et. al. (1981) used a modified version of the Owens model and estimated the relevant parameters for the Bedford Ouse. A similar approach was adopted for **QUASAR** and the following relationship developed:

Chlorophyll-a concentrations less than 50 mg/l

$P = K_8\ (1.08^{(T-20)} I^{0.79}\ 0.317\ Cl_a)$ (mg/l-day)

Chlorophyll-a concentrations greater than 50 mg/l

$P = 1.08^{(T-20)} I^{0.79}\ (K_8\ (0.317 \times 50) + K_9\ 0.317\ Cl_a)$ (mg/l-day)

Here the user specifies the two rates at which photosynthetic oxygen production occurs, one when the chlorophyll-a concentration is greater than 50 mg/l, K_9, and another when the concentration is less than 50 mg/l, K_8. K_8 is usually in the range of 0.0 to 0.03 day^{-1}, and K_9 is in the range of 0.0 to 0.02 day^{-1}. The two rates are to take account of the self shading effect at high algae concentrations. Cl_a is the chlorophyll-a concentration g/m^3, I is the solar radiation level at the earth's surface in watt hours per m^2 day. I is only input during sunlight hours determined from longitude and latitude data and also from the time of year. This assumes no cloud cover.

2.4.2.2 Respiration

The loss of oxygen due to algae respiration is described by an equation developed from Kowalczewski and Lack (1971) based on observed algae concentration measured as chlorophyll-a and respiration rate for the River Thames. Cl_a is the chlorophyll-a concentration measured as gm^{-3} and **T** is the temperature in degrees Celsius.

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

2.4.3 Benthic Oxygen Demand

Oxygen is also lost by benthic oxygen demand (river bed or mud respiration). There has been considerable research into this process (Edwards and Rolley, 1965) and the following equation has been used, where M is the benthic oxygen demand,

$$M = \frac{K_4 \quad X_4^{0.45} \quad 1.08^{(T-20)}}{\cancel{d}} \; d$$

where X_4 in the equation refers to the DO concentration mg l^{-1}, d is the river depth in metres, K_4 is the rate of oxygen uptake by the sediment and T is the temperature in degrees Celsius. The original work of Edward and Rolley was conducted on the highly polluted muds of the River Ivel and later studies by Rolley and Owens (1967) showed that the parameter K_4 varied considerably from river to river. In the Thames a value for K_4 of 0.15 day^{-1} was found to provide the best fit to the observed DO data. In **QUASAR** the equation representing benthic oxygen demand is given below:

$$M = \frac{K_4 \quad X_4 \quad 1.08^{(T-20)}}{\cancel{d}} \; d$$

K_4 is the oxygen uptake rate by sediment, usually in the range of 0.0 to 0.1 day^{-1}. This value can be edited by the user. d is the river depth in metres and is specified in the spatial data for the reach, T is the temperature in degrees Celsius.

2.4.4 Reaeration

Oxygen is added to the system by the natural reaeration of the river at the surface. Several workers have developed empirically and physically based equations. Owens Edwards and Gibbs (1964) combined previous work of Churchill et al.,(1962), and Gameson et al.,(1955) to derive the equation:

reaeration = K_2*(CS - X_4)　　　v ft s^{-1}, v_1 m s^{-1}

where K_2 is the reaeration constant given by,　d ft, d_1 m

$$K_2 = \frac{9.4 * V^{0.67}}{150 \, d^{1.85}} \qquad \left.\begin{matrix} v = a v_1 \\ d = a d_1 \end{matrix}\right\} a = 3.28$$

(days^{-1})　　$k_2 = \frac{9.4}{a^{1.18}} \frac{v_1^{0.67}}{d_1^{1.85}}$　　$a^{1.18} = 4.06$

V is the stream velocity in ft s^{-1} , d is the river depth in ft. This equation is valid within the experimentally observed ranges (velocity 0.1-5.0 ft s^{-1}; depth 0.4-11.0 ft). Elmore and West (1961) determined the temperature coefficient for the reaerration constant, later used by Churchill et al.,(1962) as shown in the equation below. Note that T is the temperature in degrees Celsius.

$$k_{(T.C)} = k_{(20 \cdot C)} \times 1.024^{(T-20)}$$

CS is the saturation concentration for DO defined as:

CS =14.652 - 0.41022T + 0.0079910T^2 -0.000077774T^3

In **QUASAR** this equation has been used with the temperature correction applied;

$$K_2 = \frac{\overset{2.27}{38.19} \times V^{0.67} \times 1.024^{(T-20)}}{d^{1.85}}$$

$d = 1$ ft　　$K_2 = 9.4$
$v = 1$ ft/s

As these variables (river velocity, temperature and depth) are all either input at the beginning of the model or generated during the model run the user does not have direct control of the reaeration coefficient and therefore the amount of oxygen added due to natural reaeration.

2.4.5 Nitrification

If there is ammonia in the water column this will be converted to nitrate. During this reaction oxygen is consumed. Thus there is a term for oxygen depletion as a result of nitrification as discussed in 2.2.1

Nitrification $= 4.57 \,.\, 10^{(T \cdot 0.0203)} . K_{15} . X_6$

where K_{15} is the ammonia nitrification rate coefficient generally ranging from 0.0 to 0.5 day^{-1}. The value for K_{15} can be edited by the user. T is the temperature in degrees Celsius, and X_6 is the ammonia concentration. The 4.57 term arises from the stoichiometry of the reaction.

2.4.6 BOD

The biochemical oxygen demand is caused by the decay of organic material in the stream. As the material decays it consumes oxygen, a process which is included in the model as:

BOD $= K_1 . X_5$ (mg/l-day)

where K_1 is the rate coefficient for the loss of BOD and X_5 is the concentration of BOD in the stream. The value for K_1 can be edited by the user.

2.5 Biochemical Oxygen Demand

The change in the biochemical oxygen demand is due to decay, sedimentation and addition due to dead algae. The differential equation describing the rate of change of BOD concentration with time is given below:

If $c \neq 1$

then

$$\frac{d(X_5)}{dt} = \frac{U_5 - X_5}{TC \cdot (1-c)}$$

$$\frac{\left(-K_1 \cdot X_5 \right.}{} \quad \text{BOD decay}$$

$$-K_{18} \cdot X_5 \quad \text{sedimentation}$$

$$\frac{\left. +K_{10} \right)}{1-c} \quad \text{BOD contribution by algae}$$

If $c = 1$

then

$$X_5 = U_5 - [K_1 \cdot X_5 - K_{10}]TC$$

where U_5 and X_5 are the input and output BOD concentrations and K_1 and K_{18} are the rate coefficients associated with the processes indicated.

Note that if the Dissolved Oxygen level goes to zero, then the term involving K_1 is left out.

2.5.1 BOD Decay

The biochemical oxygen demand is caused by the decay of organic material in the stream. As the material decays it consumes oxygen. Knowles and Wakeford (1978) found the rate of change due to oxidation to be dependant on the temperature. This process has been modelled in the same manner:

BOD $=1.047^{(T-20)} \cdot K_1 \cdot X_5$ (mg/l-day)

where T is the temperature in degrees Celsius, K_1 is the rate coefficient for the loss of BOD and is usually in the range of 0.0 to 2.0 day^{-1} and X_5 is the concentration of BOD in the stream in mg/l. The value for K_1 can be edited by the user.

2.5.2 Loss by Sedimentation

Loss of BOD can also occur by sedimentation. This occurs at a rate proportional to the amount of BOD present. The sedimentation rate is currently set at 0.1 day^{-1}.

2.5.3 BOD Contribution by Algae

As algae die they contribute to the BOD. The rate of contribution is proportional to the product of the concentration of algae and the rate of BOD addition by dead algae, usually in the range of 0.0 to 0.1 day^{-1}. This value can be edited by the user.

2.6 Ammonium Ion

The loss of ammonia is due to oxidation. The differential equation describing the rate of change of ammonia concentration is given below:

$$\frac{d(X_6)}{dt} = \frac{U_6 - X_6}{TC \cdot (1-c)}$$

$$- K_{15} \cdot X_6 \quad \text{loss by nitrification}$$

where U_6 and X_6 are the input and output ammonia concentrations and K_{15} is the nitrification rate. A detailed description of this process is given in section 2.2.1. The ammonia nitrification rate is dependant on the temperature and described by the equation:

Ammonia nitrification rate = $K_{15} \cdot 10^{(T \cdot 0.0293)}$

Note that if the Dissolved Oxygen level goes to zero, then the last (K_{15}) term is left out.

2.7 Ammonia

The concentration of ammonia is not actually produced as an output by the model, but it is computed by the plot program from the ammonia concentration, pH and temperature data. This is determined by considering the following equilibrium:

$NH_4^+ = NH_3(aq) + H^+(aq)$

It is assumed that the modelled Ammonia, NH_3, is the total ammonia present, ie.:

$NH_{3(modelled)} = NH_4^+ + NH_3$

The concentration of the ammonia is then given by the equation:

$$NH_3 = \frac{NH_{3(modelled)}}{(1.0 + 10^{pKA - pH})}$$

where pKA is the dissociation constant, KA adjusted for temperature. The value of pKA is assumed to vary inversely with absolute temperature (this assumption being derived from the equation for Gibbs Free Energy): $pKA = \frac{2754.9}{(T + 273.15)}$ where T is the temperature in degrees Celsius.

2.8 Temperature

The differential equation for temperature is given below where U7 and X7 are the input and output temperatures.

If $c \neq 1$

then

$$\frac{d(X_7)}{dt} = \frac{U_7 - X_7}{TC \cdot (1 - c)}$$

If $c = 1$

then

$X_7 = U_7$

2.9 Ortho-phosphate

Changes in Ortho-phosphate are due to decay.

If $c \neq 1$

then

$$\frac{d(X_8)}{dt} = \frac{U_8 - X_8}{TC \cdot (1 - c)}$$

$$- K_{16} \cdot X_8 \qquad \text{Ortho-phosphate decay}$$

If $c = 1$

then

$X_8 = U_8 - K_{16} \cdot X_8 \cdot TC$

where U_8 and X_8 are the input and output ortho-phosphate concentrations and K_{16} is the rate of ortho-phosphate decay usually in the range of 0.0 to 2.0 days^{-1}.

2.10 pH

The differential equation for pH is given below where U_9 and X_9 are the input and output pH.

$$\frac{d(X_9)}{dt} = \frac{U_9 - X_9}{TC \cdot (1 - c)}$$

3 Data Requirements

Three sets of data are required to operate **QUASAR**; a catchment structure consisting of a river map, boundary conditions which define the water quality and flow of the tributaries and of the water at the top of the river, and reach parameters consisting of data specific to each reach.

3.1 Catchment Structure

The first step in creating a catchment structure is to determine the river network to be modelled. Tributaries entering the river network need to be specified and finally the river must be divided into reaches. Reach boundaries are determined to be points in the river at which there is a change in the water quality due to the confluence of a tributary, the location of a sewage treatment works effluents discharge, abstractions, and locations of weirs. Water quality changes due to biological or physical chemical reactions should also be considered by ensuring the reach length is not too long. Reach boundaries can also be established at points were water quality monitoring stations are located to be used as calibrating points. Below is a summary of the steps required in establishing a catchment structure.

1. Determine the extent of the river to be modelled.
2. Determine if any tributaries enter the river network which are not being modelled.
3. Establish reach boundaries:

- at the location of tributaries, Sewage Treatment Works Effluent Discharges,
  weirs, abstractions, monitoring stations
- roughly determine distances between reach boundaries
- further divide the river up by using a reach length of no greater than 5 km
  as a guide

3.2 Boundary Conditions

The boundary conditions consist of the water quality and flow data of the river network at the points modelling begins and for the tributaries that are not modelled at the point where they enter the river network. Time series data are required in the dynamic mode while statistical data are required in the planning mode.

3.2.1 Planning Mode

In the planning mode each water quality and flow parameter requires a probability distribution and its characteristics to be specified. A choice between three probability distributions is presently available; gaussian, lognormal, or rectangular. The mean and standard deviation are required if lognormal or gaussian distributions are chosen, and lower and upper bounds are required if a rectangular distribution is required. The list below is a summary of the required data:

Flow & Water Quality		
	Distribution	-normal -lognormal -rectangular
	Characteristics	-mean, standard deviation -or lower/upper bounds

3.2.2 Dynamic Mode

In the dynamic or forecasting mode time series data of the water quality and flow are required as well as the run and output time step, and the run output length. Time series data consist of daily mean flow values and monthly mean water quality values. The run time step allows the user to specify the time period over which the model equations will operate. The output time step defines the time period for which output data are generated. The run output length specifies the number of output steps that the model will generate. The list below is a summary of the required data:

Daily mean flow data
Monthly mean water quality values
Run time step
Output time step
Run output length

3.3 Reach parameters

Reach parameters consist of data specific to each reach such as the rate coefficients, velocity-flow relationships, spatial data, weir specifications and monthly algae data. They must be specified for each reach.

3.3.1 Rate Coefficients

Rate coefficients are required to describe the rate at which the chemical processes are occurring in the reach. The ratewhich have to be specified include:

- K_5 Denitrification (0.0 - 0.5 day$_{-1}$)
- K_1 Biochemical Oxygen Demand decay (0.0 - 2.0 day$_{-1}$)
- K_{15} Ammonia nitrification (0.0 - 0.5 day$_{-1}$)
- K_4 Oxygen uptake by sediment (0.0 - 1.0 day$_{-1}$)
- K_{10} Addition of BOD by dead algae (0.0 - 0.1 day$_{-1}$)
- Photosynthetic oxygen production
- K_8 - chlorophyll - a up to 50 mg/l (0.0 - 0.03 day$_{-1}$)
- K_9 - chlorophyll - a above 50 mg/l (0.0 - 0.02 day$_{-1}$)
- K_{16} Decay of ortho-phosphate (0.0 - 2.0 day$_{-1}$)
- K_{19} Sedimentation of BOD (0.0 - 2.0 day$_{-1}$) · I ?
- Algae Respiration (offset) (0.0 - 2.0 day$_{-1}$)
- Algae Respiration (offset) (0.0 - 2.0 day$_{-1}$)

mud surface area

3.3.2 Velocity-Flow Relationship

The reach's velocity - flow relationship has three parameters that relate the velocity of water (m/s) in the reach to its flow in cumecs. The equation is of the form:

velocity (m/s) = A + B . Flow ^{c}.

The A, B, and C coefficients are entered into **QUASAR**.

3.3.3 Spatial Data

Spatial data for the reach consist of the reach length and depth, the number of lags (or cells) in the reach and the latitude, longitude, and time zone that the reach is in. Below is a list of the required data for each reach.

length (m)
depth (m)
number of lags (cells)
latitude
longitude
time zone

3.3.4 Weir

The presence of a weir in a reach can be specified in the reach parameters. Four types of weirs can be chosen from, these include: free, slope, step, or cascade (or none). The height from the top of the weir to the downstream water level must also be specified. Below is a list of the required data for each reach.

Type of weir (free, slope, step, cascade, none)
Height (m) Distance from the top of the weir to the downstream water level

3.3.5 Algae Data

Monthly algae data specifying chlorophyll-a concentrations are required in the calculation of dissolved oxygen concentrations.

4 References

Alexander M. (1965) Nitrification. Agronomy. **10**, 307-343.

Curtis E.J.C., Durrant K., and Harman M.M.I. (1974) Nitrification in Rivers in the Trent Basin. Wat. Res. **9**, 255-268.

Department of Environment. (1973) Aeration at Weirs. Notes on Water Pollution. **61**.

Edwards R.W. and Rolley H.L.J. (1965) Oxygen Consumption of River Muds. J. Ecol. **53**, 1-19.

Kowalczewski H. and Lack T. (1971) Primary Production and Respiration of the Phytoplankton of the River Thames and Kennet at Reading. Freshwater Biology. **1**, 197-212.

Knowles G. and Wakeford A.C. (1978) A Mathematical Deterministic River-Quality Model: Part 1: Formulation and Description. Wat. Res. **12**, 1149 - 1153.

Lack T. (1971) Quantitative Studies on the Phytoplankton of the River Thames and Kennet at Reading. Freshwater Biology, **1**, 213-224.

Owens M., Edwards R.W., and Gibbs J.W. (1964) Some Reaeration Studies in Streams. Int. J. Air Wat. Pollution. **8**, 469-486.

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

Toms I.P., Mindenhall M.J. and Harman M.M.I. (1975) Factors Affecting the removal of Nitrate by Sediments form Rivers, Lagoons, and Lakes. Water Research Technical Report TR14.

Wakeford A.C. and Knowles G. (1978) A Mathematical Deterministic River-Quality Model -Part 1: Formulation and Description. Wat. Res. **12**, 1149-1161.

Warn, A.E. (1982) Calculating Consent Conditions to achieve River Quality Objectives. Effl. and Water Treatment J., 152-155.

Whitehead P.G. and O'Connell P.E. (1982) Water Quality Modelling, Forecasting and Control In Proc. 1st Inter. Workshop. July/August 1980, Institute of Hydrology.

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. Wat. Res. **15**, 1157-1171.

Whitehead, P.G., Caddy D.E., and R.F. Templeman (1984) An On-Line Monitoring, Data Management and Forecasting System for the Bedford Ouse River Basin. Wat. Sci. Tech. **16**, 295-314.

Whitehead, P.G. and Williams, R.J. (1982) A Dynamic Nitrate Balance Model for River Basins. IAHS Exeter Conference Proceedings, IAHS publication No 139.

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

Whitehead P.G., Williams R.J. and Hornberger G. (1986) On the Identification of Pollutant or Tracer Sources Using Dispersion Theory. J. Hydrol. 84, 273-286.

Whitehead P.G. and P. Young (1979) Water Quality in River Systems: Monte Carlo Analysis. Wat. Resour. Res. 15, 451-459.

Whitehead P.G., Young P., and G. Hornberger (1979) A Systems Model of Stream Flow and Water Quality in the Bedford-Ouse River - I. Stream Flow Modelling. Wat. Res. 13, 1155-1169.

Appendix 3 Calibration of water quality model

Calibration and validation of the water quality model

The input data included all abstraction, discharge, and tributary flow and quality data from Cookham (just upstream of Maidenhead and the proposed start of the flood relief channel) down to Teddington (the upstream limit of all but exceptional tides, and also the site of a well-calibrated gauging site).The model was calibrated on 1974 data and then validated on 1975 data. Flow and quality data were available on the Thames itself at Datchet, Egham, Walton and Teddington which allowed validation at these points down the river. Flow was calibrated first, since it is fundamental to the mass balance for the quality determinands. As the model has accurate measurements of velocity-flow relationships, and the quality of the flow data is by far the best of all the variables, it was not surprising that the fit was very good, especially higher up the river at Datchet (see Figure A3.1; the dashed line shows the modelled). The flow fit at Teddington, at the bottom of the reach system, was still good with the exception of day 248 where the peak was missed - see Figure A3.2. This seems to suggest that the input data contained inaccurate measurements rather than that the model is inaccurate. With this kind of model, the only real scope for error is in the velocity-flow relationships, which in this case were known to be accurate. Overall the flow model calibrated well against the 1974 data set at all locations on the River Thames. The validation against the 1975 data also produced good fits. Thus the flow model is a good basis on which to model the water quality variables.

The other variables were not expected to be such good fits because the inputs were measured at most only twice a week. All input data was linearly interpolated to give daily values; dissolved oxygen and temperature were also adjusted to give values at 12:00 whenever the time of measurement had been recorded in order to be consistent. Midday was chosen because the measurements were almost always made between 09:00 and 15:00. The same correction was applied to the calibration and validation data. It was of course realized that both of these variables vary throughout the day, but the quantity of data available was not sufficient to model to that degree using QUASAR and is dealt with in section 5.2.

(ah!)

The other variables to be calibrated were nitrate, dissolved oxygen, biochemical oxygen demand, ammonia, temperature and orthophosphate. Temperature has no calibration coefficients, but is again very simple to fit, and was reasonably accurate, taking the above considerations into account (Fig A3.3). The rate coefficients are all temperature dependent. A fixed input to the model is the levels of algae which affects DO and BOD. Unfortunately comprehensive measurements of algae concentrations were not available except at Egham for 1974-76; data was only available at Walton for 1989 and at Egham, Walton and Teddington for 1983/4 - these values were then assumed to be the levels for the entire length of the Thames under consideration, except for 1983/4 when the levels at Egham were assumed to be the same as the levels above Egham, and then the levels for Walton and Teddington were used for those two sites. The independent variables were calibrated first - ammonia, orthophosphate and BOD. Ammonia is controlled by one parameter (ammonia nitrification); orthophosphate is only controlled by a decay coefficient and BOD is controlled by three terms, two subtracting (BOD decay and Sedimentation) and one adding (addition to BOD by dead algae). Once these were fitted, nitrate and then DO were then fitted; nitrate is controlled by the denitrification rate, the ammonia nitrification rate and the concentration of ammonia while DO is controlled by the concentrations of BOD, nitrate and ammonia as well as reaeration, weir aeration and algal respiration and photosynthesis. The calibration was done by testing against the measured values at the four measuring sites down the river and so four different sets of values were found for the 11 rate coefficients for the four sections of the river (namely Cookham - Datchet, Datchet - Egham, Egham - Walton and Walton - Teddington).

Nitrate: The fits for nitrate were excellent all the way down the river. (See Figs A3.4 - A3.8)

Ammonia: The ammonia levels were good with the exception of a few unexplained missed peaks at Teddington; the ammonia values are particularly spiky at Teddington - a possible indication of inaccurate measurements either of the values at Teddington, or for one of the inputs as the inputs do not show any signs of fluctuations. (See Figs A3.9 - A3.12)

BOD: BOD had an excellent fit at Datchet and Walton, but was not so good at Teddington and Egham (which is situated between Datchet and Walton), demonstrating very well the effect of intermittent measurements as there was one peak in particular (around day 45) at Egham and Teddington evident in the observed data which is completely missed at the other sites (Figs A3.13 - A3.16). The peak is a short burst of high level BOD and was almost certainly missed at Egham and Datchet because the measurements were made either side of the pulse. The initial high value is also missed in all cases, but this is to be expected as the model is settling down at this stage. The model is also at present unable to cope with better than monthly inputs of algae levels - the quality of data really precludes this, but there is evidence to suggest that the algal peak was actually slightly earlier than the beginning of April as was entered into the model (Fig A3.17).

Dissolved Oxygen: Dissolved oxygen was the least impressive of the fits, although after much effort the fits down as far as Egham were reasonable and not much worse at Teddington. (See Figs A3.18 - A3.19).

Orthophosphate: Orthophosphate has fairly constant levels, values of around 1 mg/l were the norm for both observed and modelled. (See Figs A3.20 - A3.22).

Table A3.1 Calibrated parameter values

Process	Cook - Datchet	Datchet - Egham	Egham - Walton	Walton - Teddington
De-nitrification	0.02	0.02	0.02	0.010
BOD Decay	0.20	0.15	0.15	0.07
Ammonia Nitrification	0.27	0.19	0.18	0.10
Sediment 02 uptake	1.0	1.0	1.0	1.0
Dead algae BOD contribution	0.01	0.02	0.025	0.005
Algal photosynthetic 02 production <50mg/l	0.20	0.10	0.20	0.20
Algal ph. 02 prod. >50mg/l algae	0.15	0.05	0.15	0.15
Orthophosphate decay	1.5	0.0	0.2	0.1
Sedimentation of BOD	0.1	0.1	0.1	0.00
Algae respiration offset	2.00	2.00	2.00	2.00
Slope	0.013	0.013	0.013	0.013

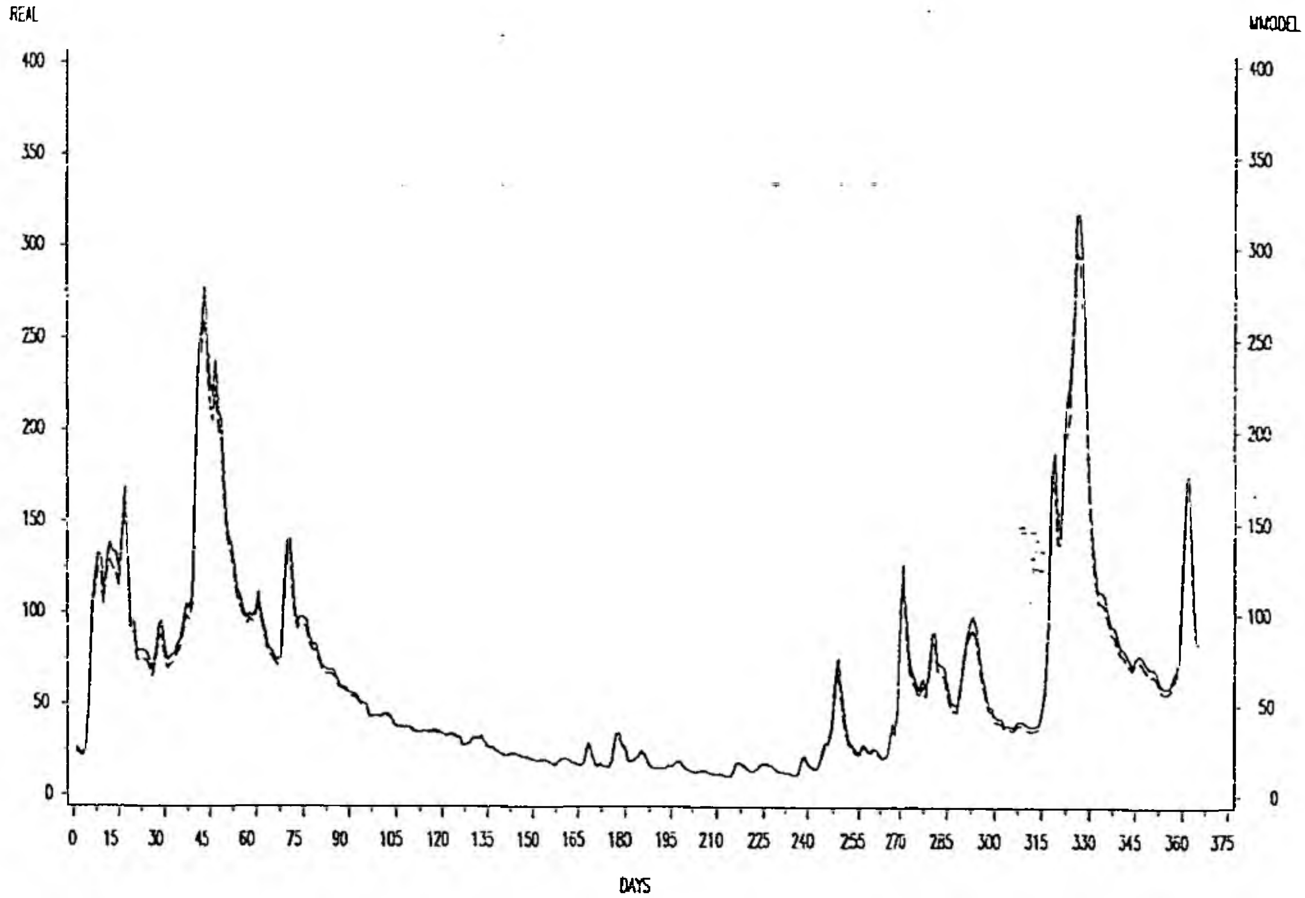
Appendix 3 List of figures

Note: In all these plots
- Observed data are shown as a full line
- Modelled data are shown as a dotted line

Flow at Datchet 1974

Figure A3.1

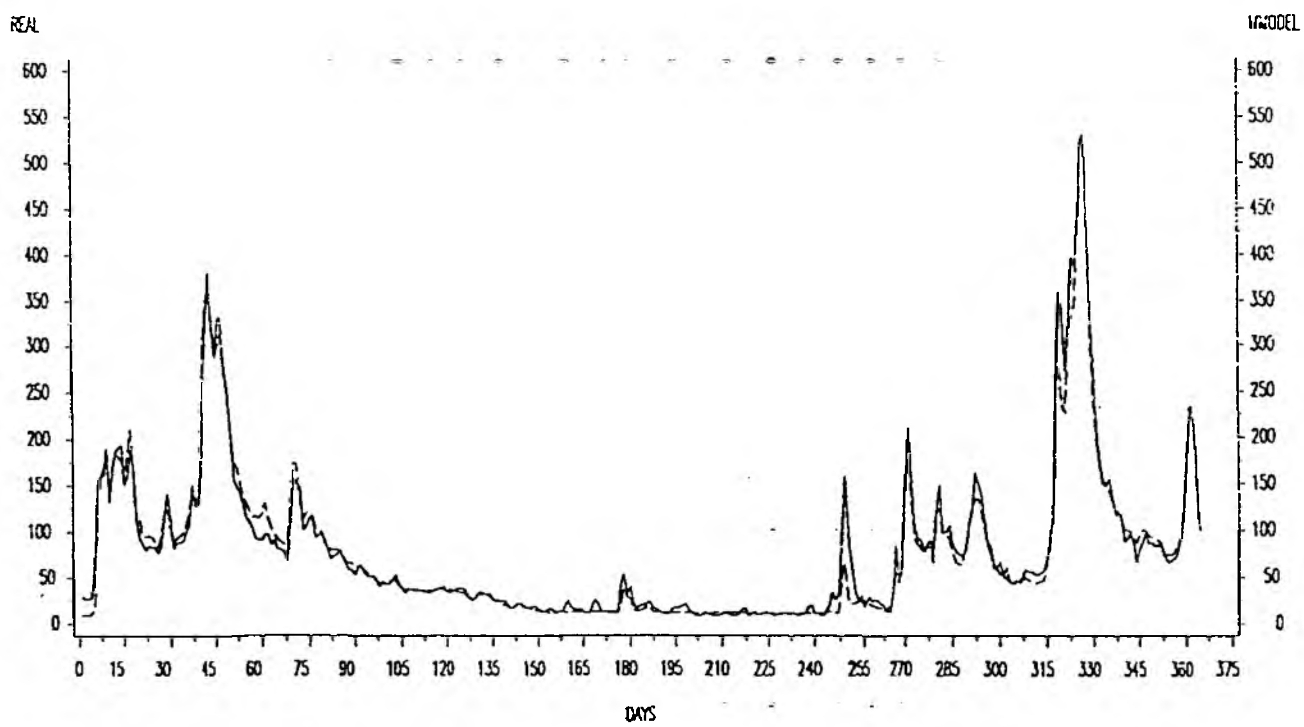
Observed vs Modelled



Flow at Teddington 1974

Figure A3.2

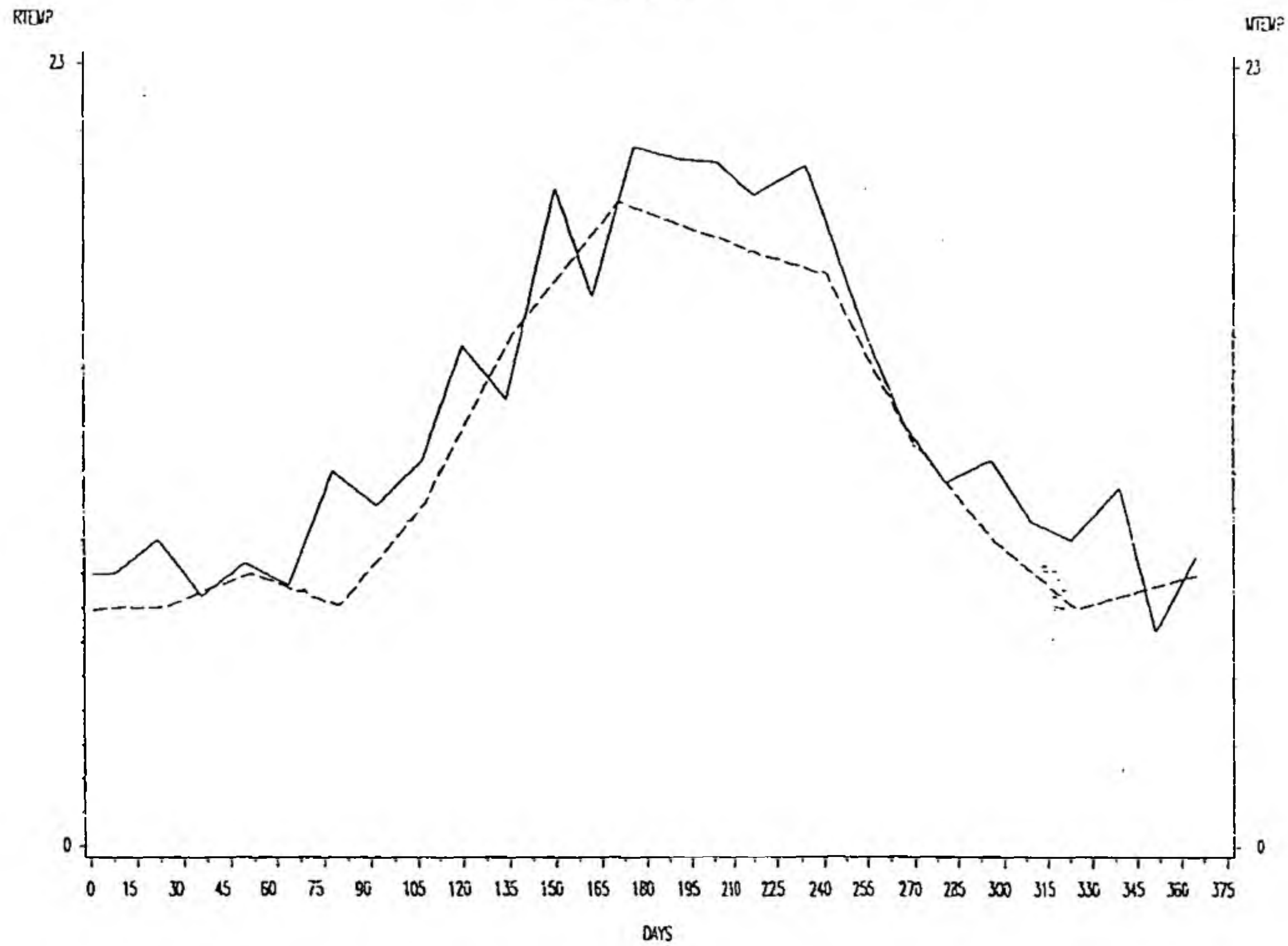
Observed vs Modelled



Temperature at Datchet 1974

Figure A3.3

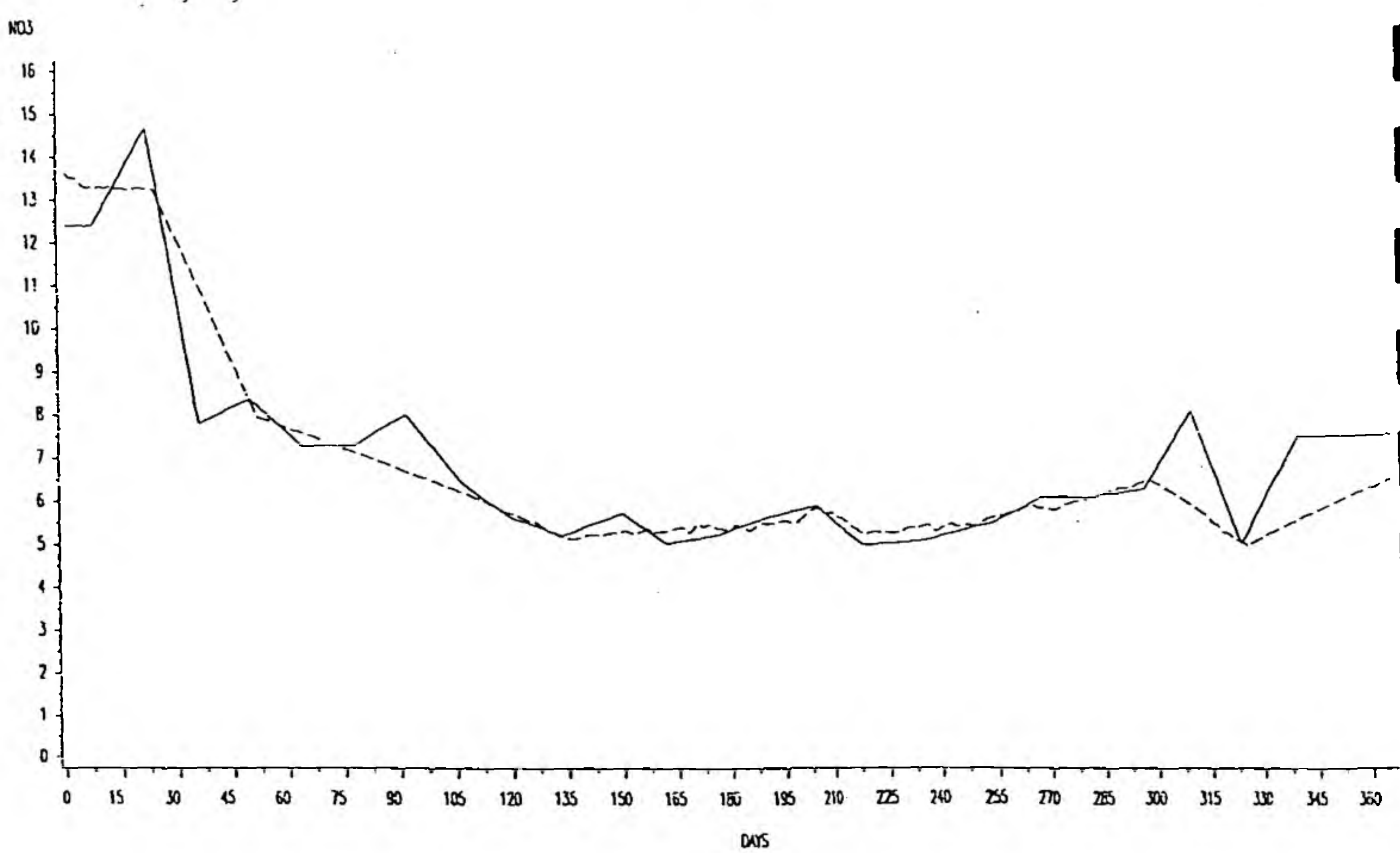
Observed vs Modelled



Nitrate at Datchet 1974

Figure A3.4

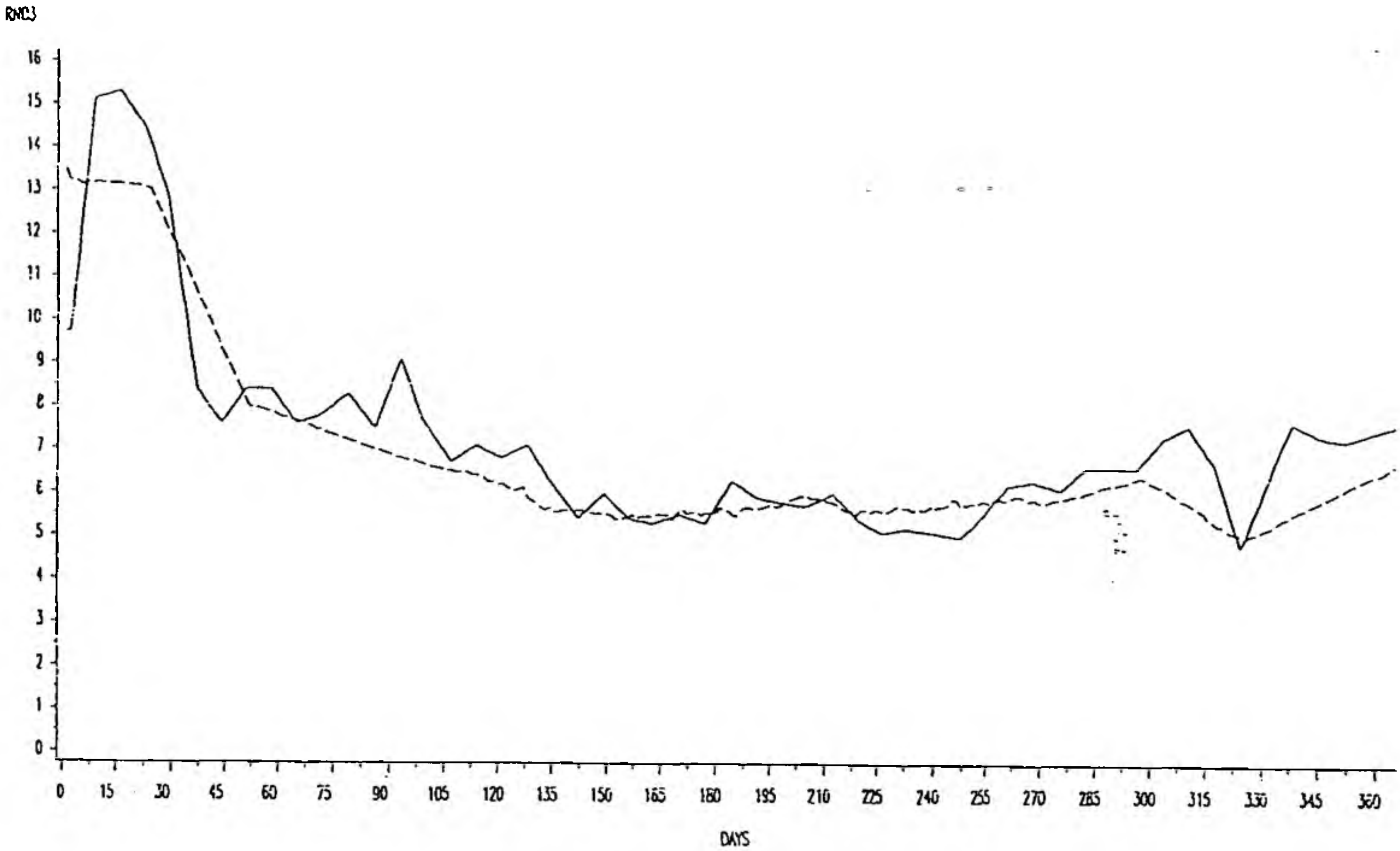
Observed vs Modelled



Nitrate at Egham 1974

Figure A3.5

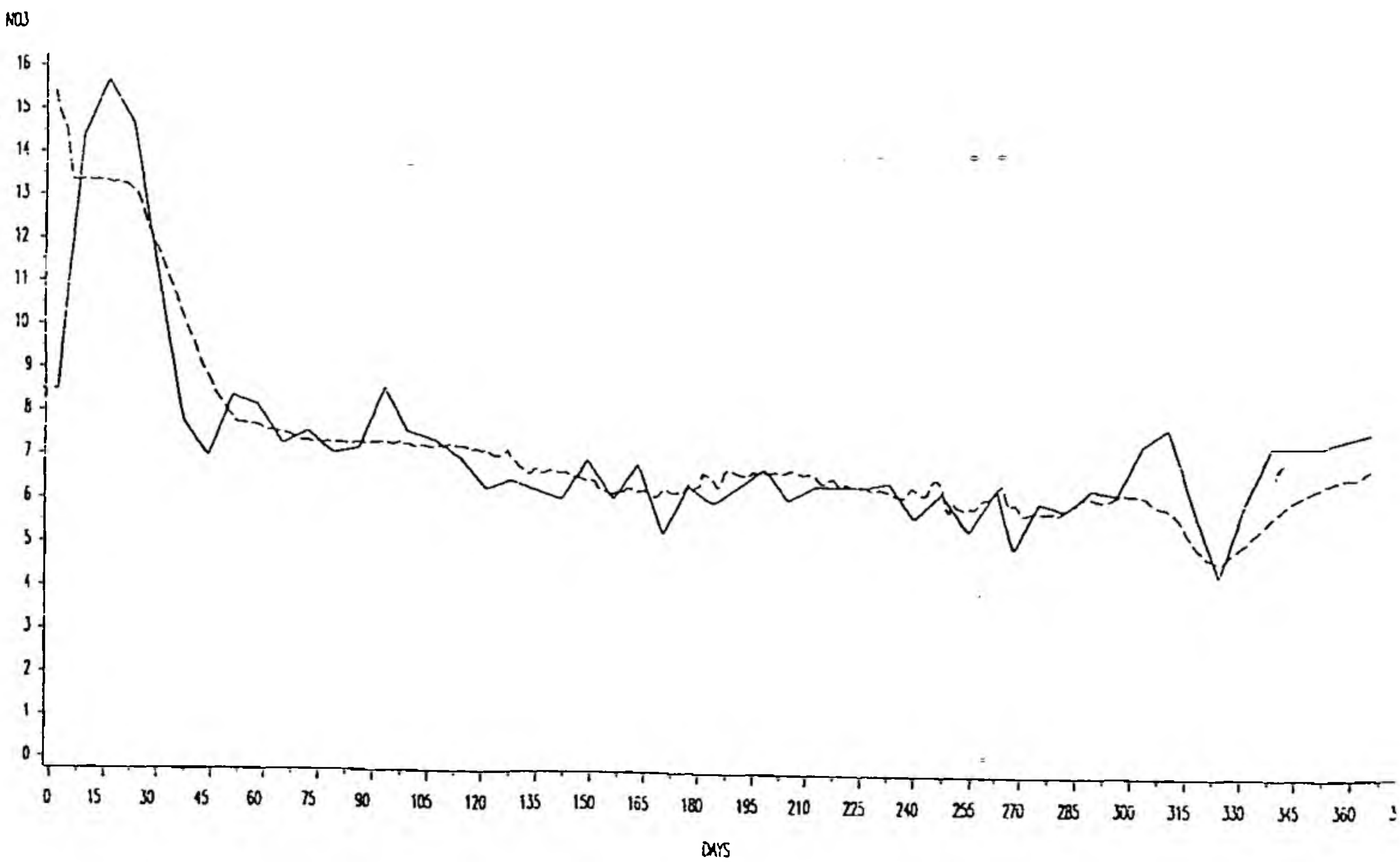
Observed vs Modelled



Nitrate at Teddington 1974

Figure A3.6

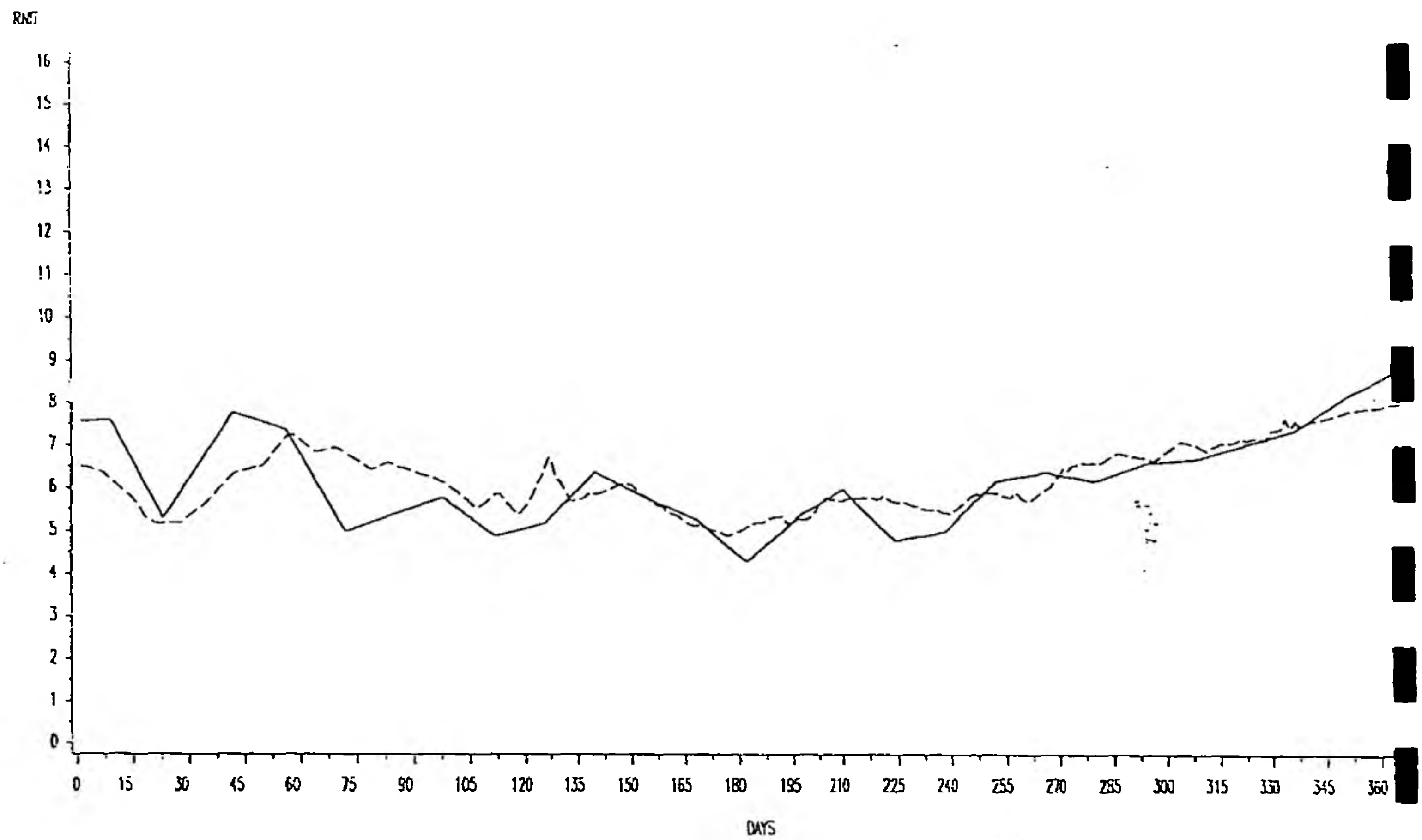
Observed vs Modelled



Nitrate at Datchet 1975

Figure A3.7

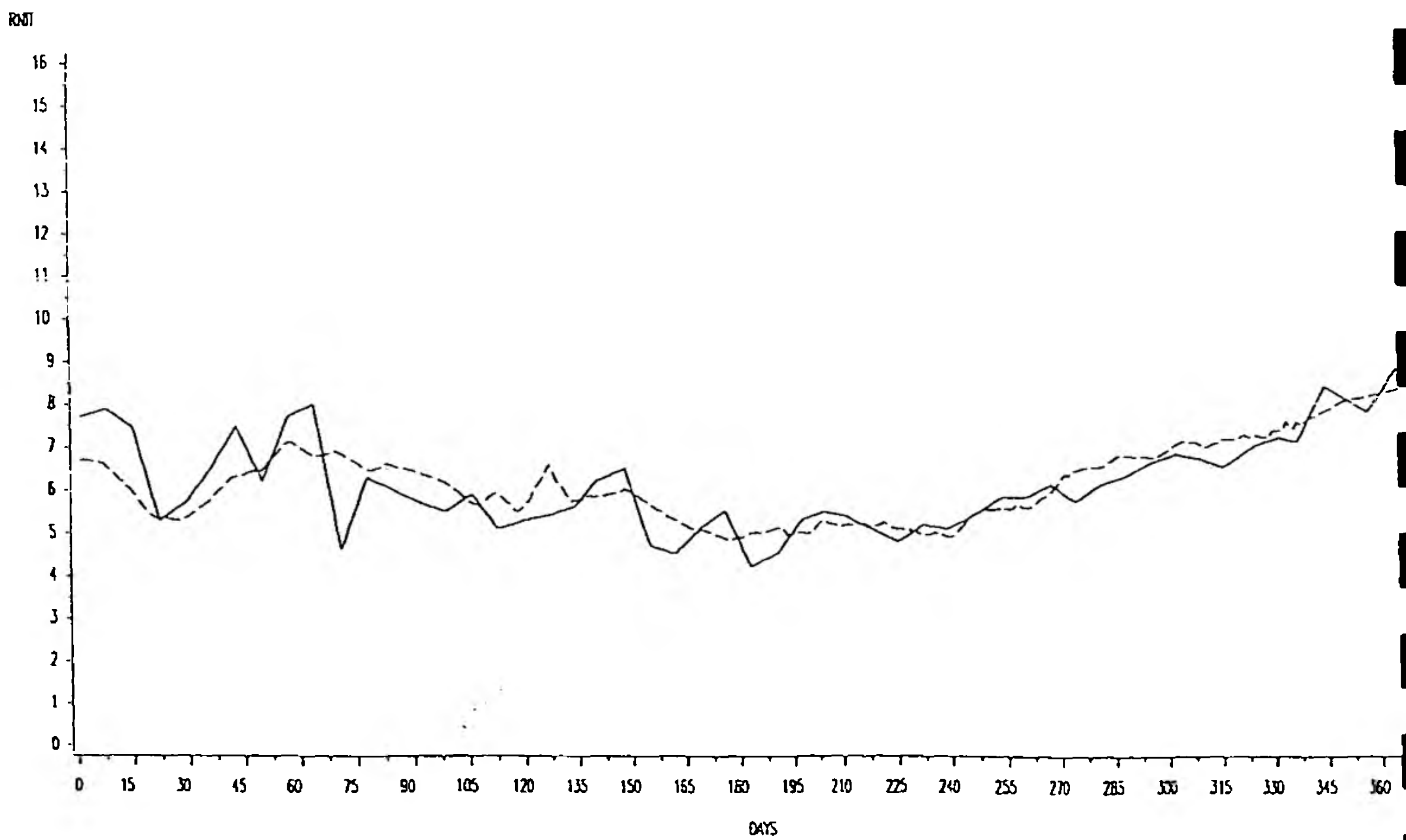
Observed vs Modelled



Nitrate at Egham 1975

Figure A3.8

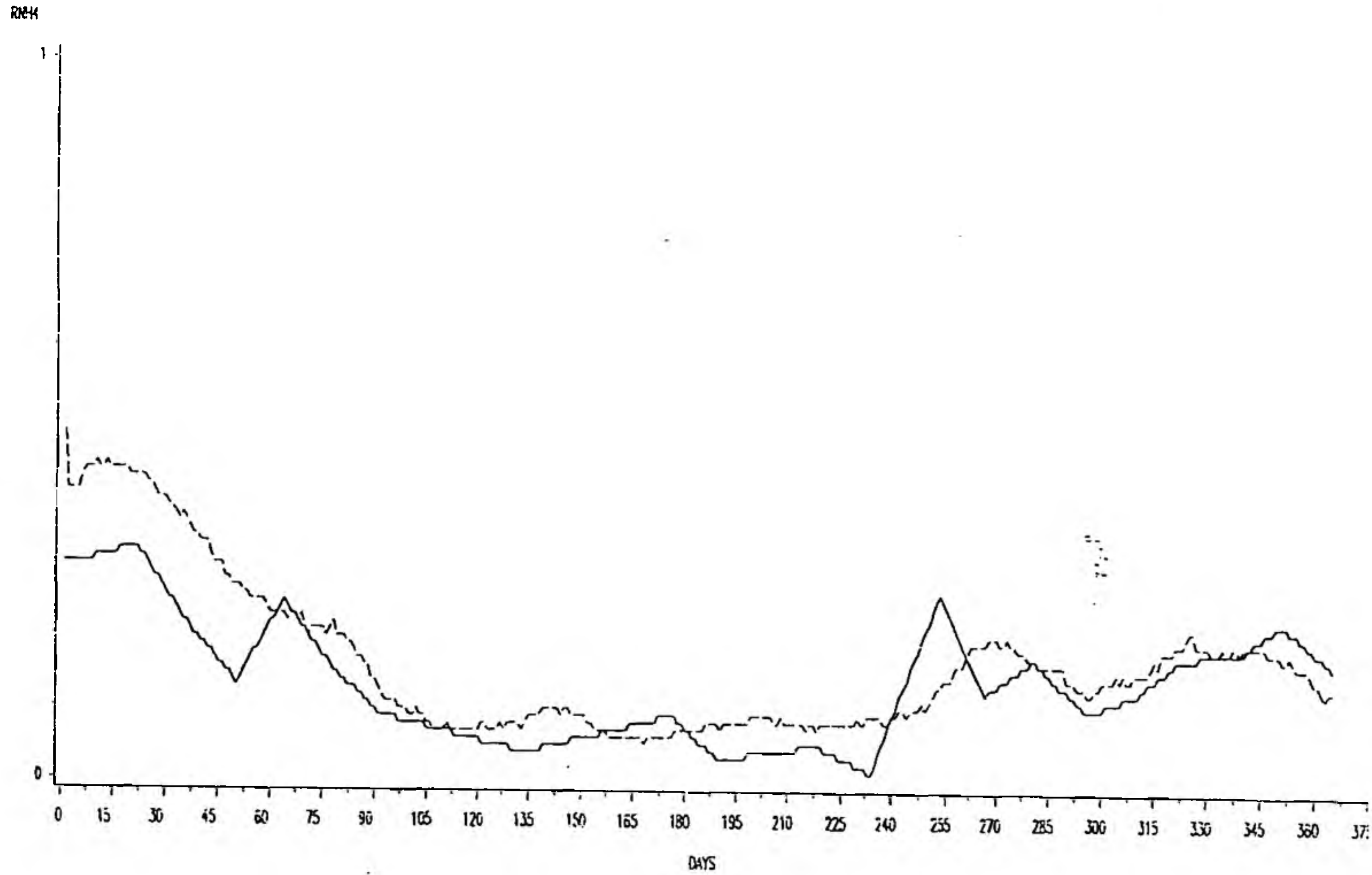
Observed vs Modelled



Ammonia at Datchet 1974

Figure A3.9

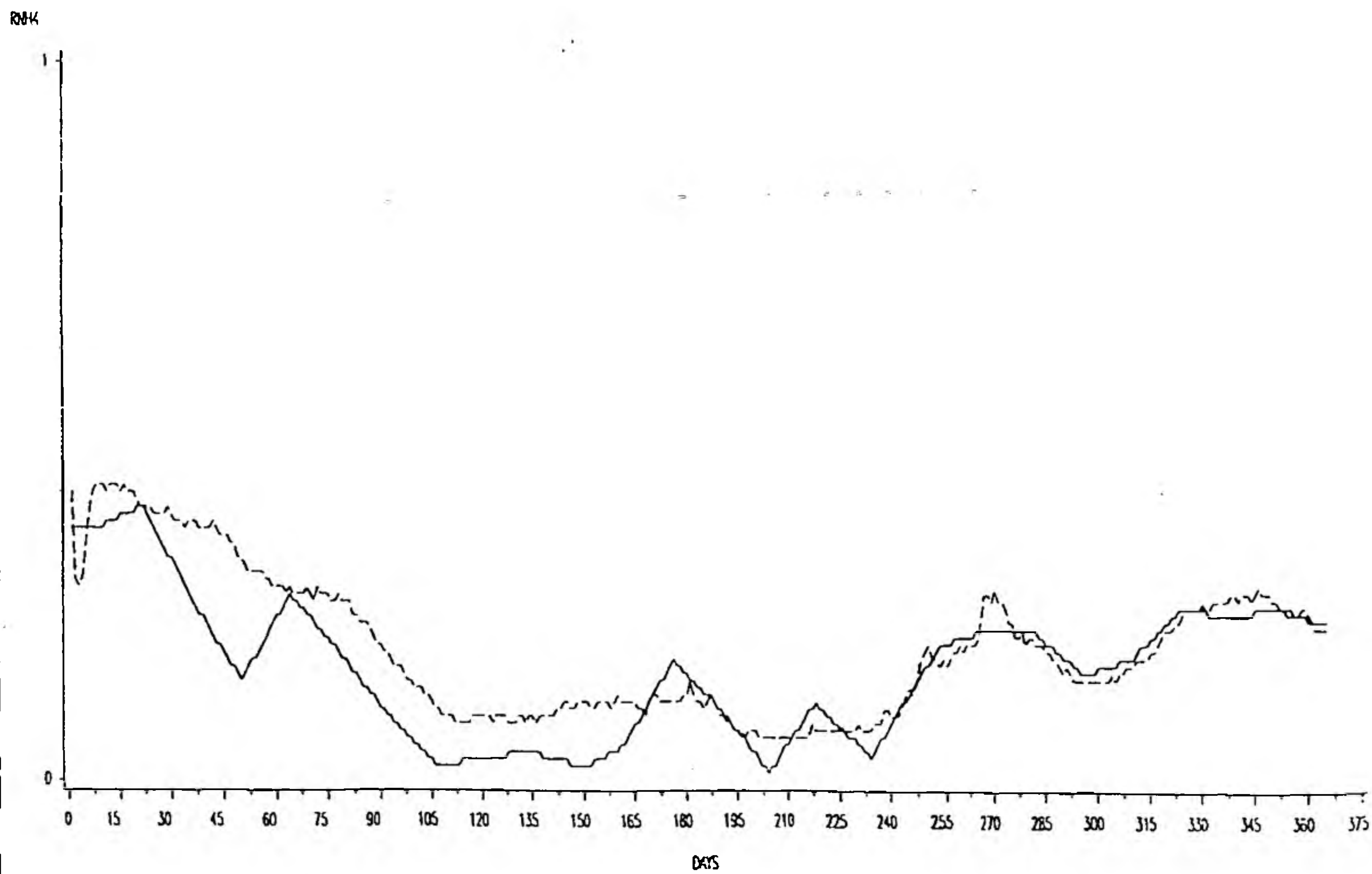
Observed vs Modelled



Ammonia at Walton 1974

Figure A3.10

Observed vs Modelled



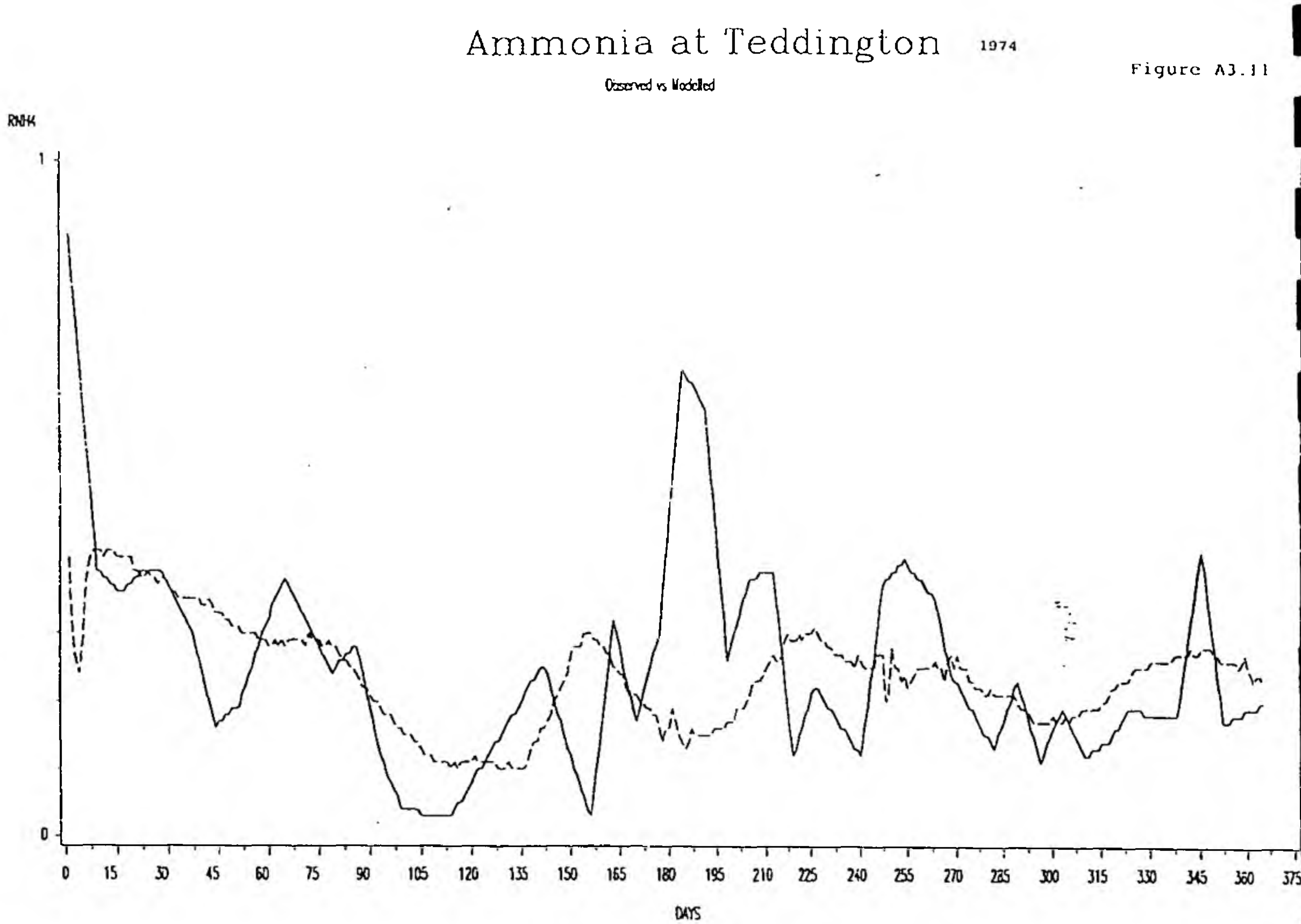


Figure A3.11

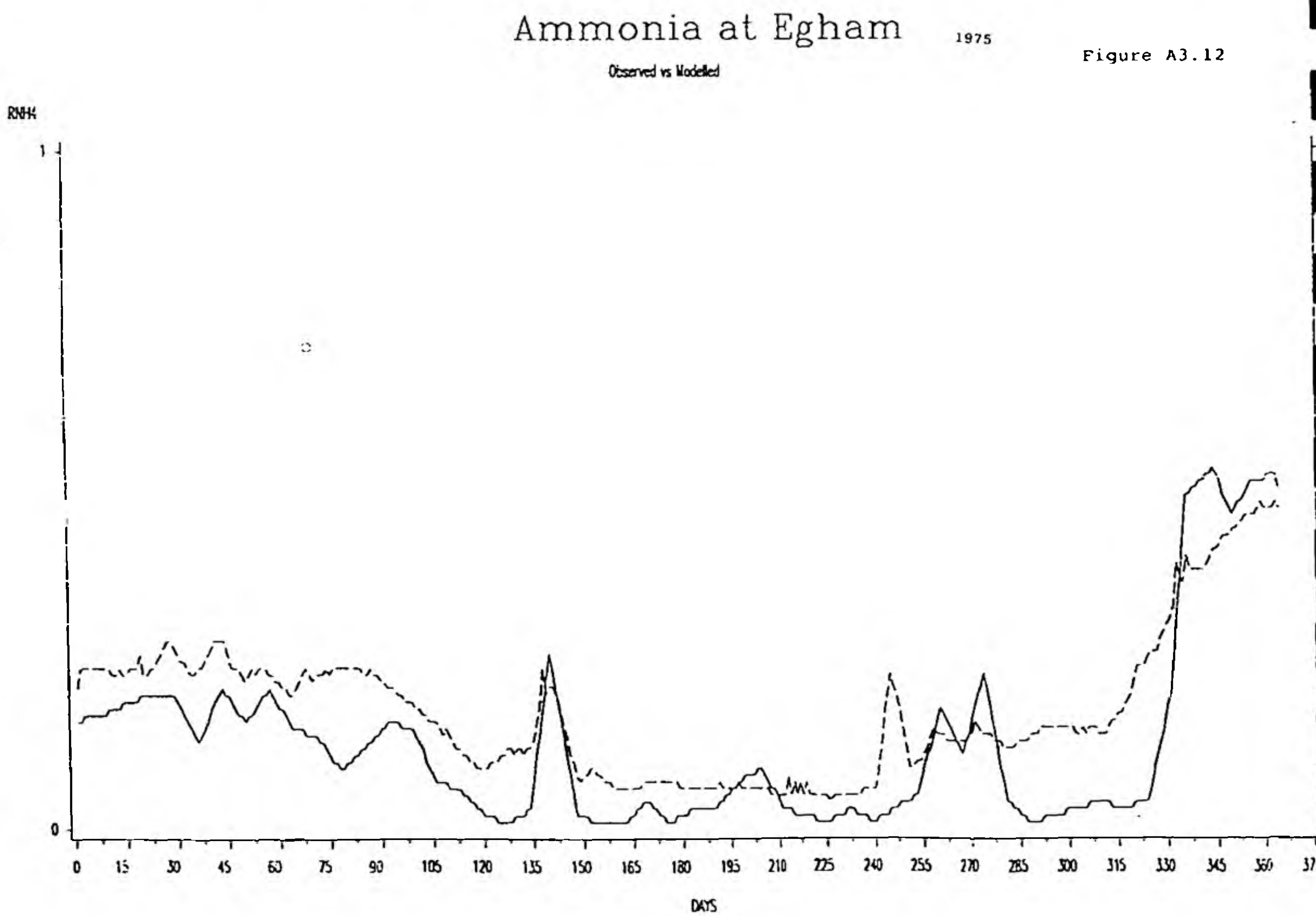
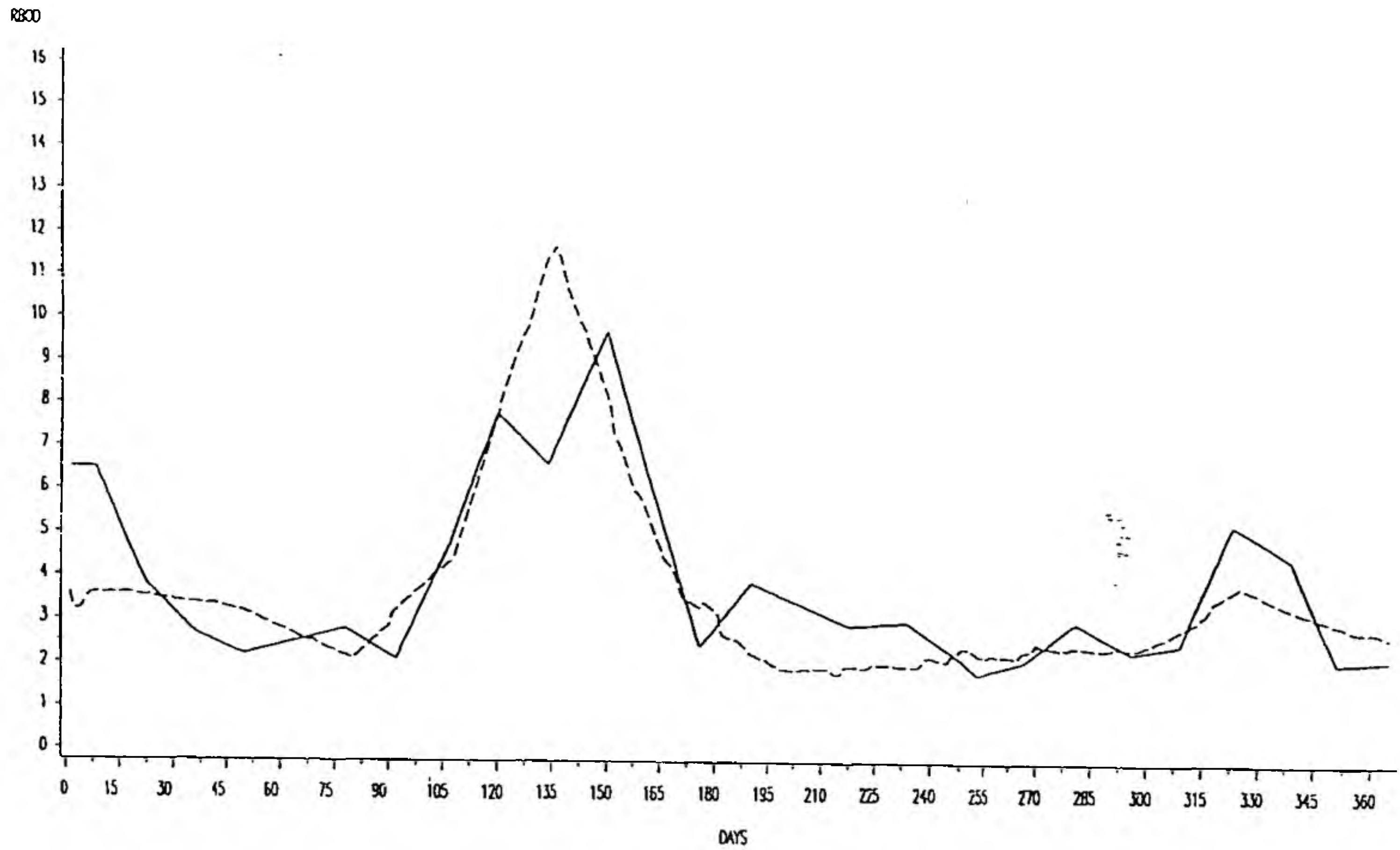


Figure A3.12

B O D at Datchet 1974

Figure A3.13

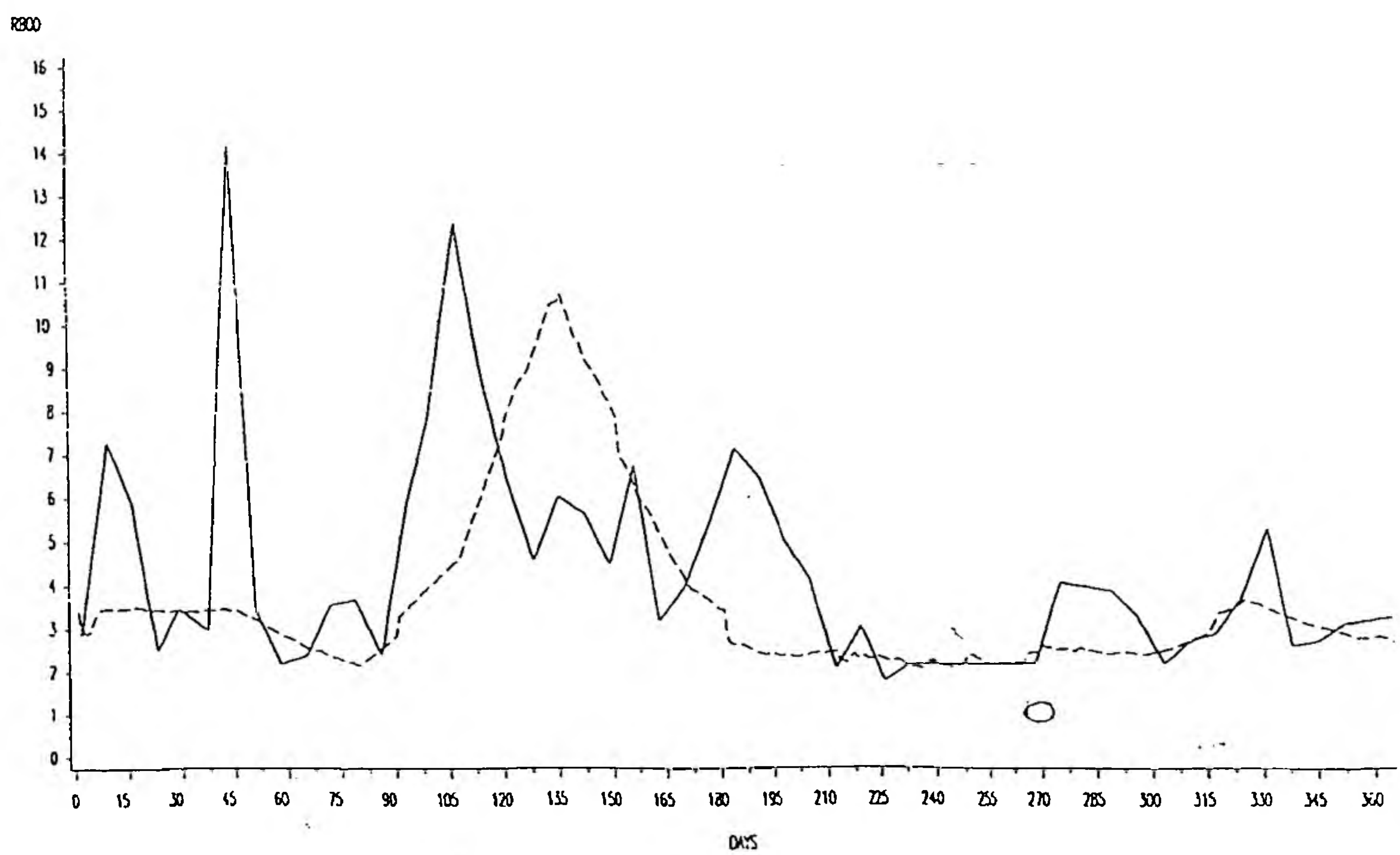
Observed vs Modelled



B O D at Egham 1974

Figure A3.14

Observed vs Modelled

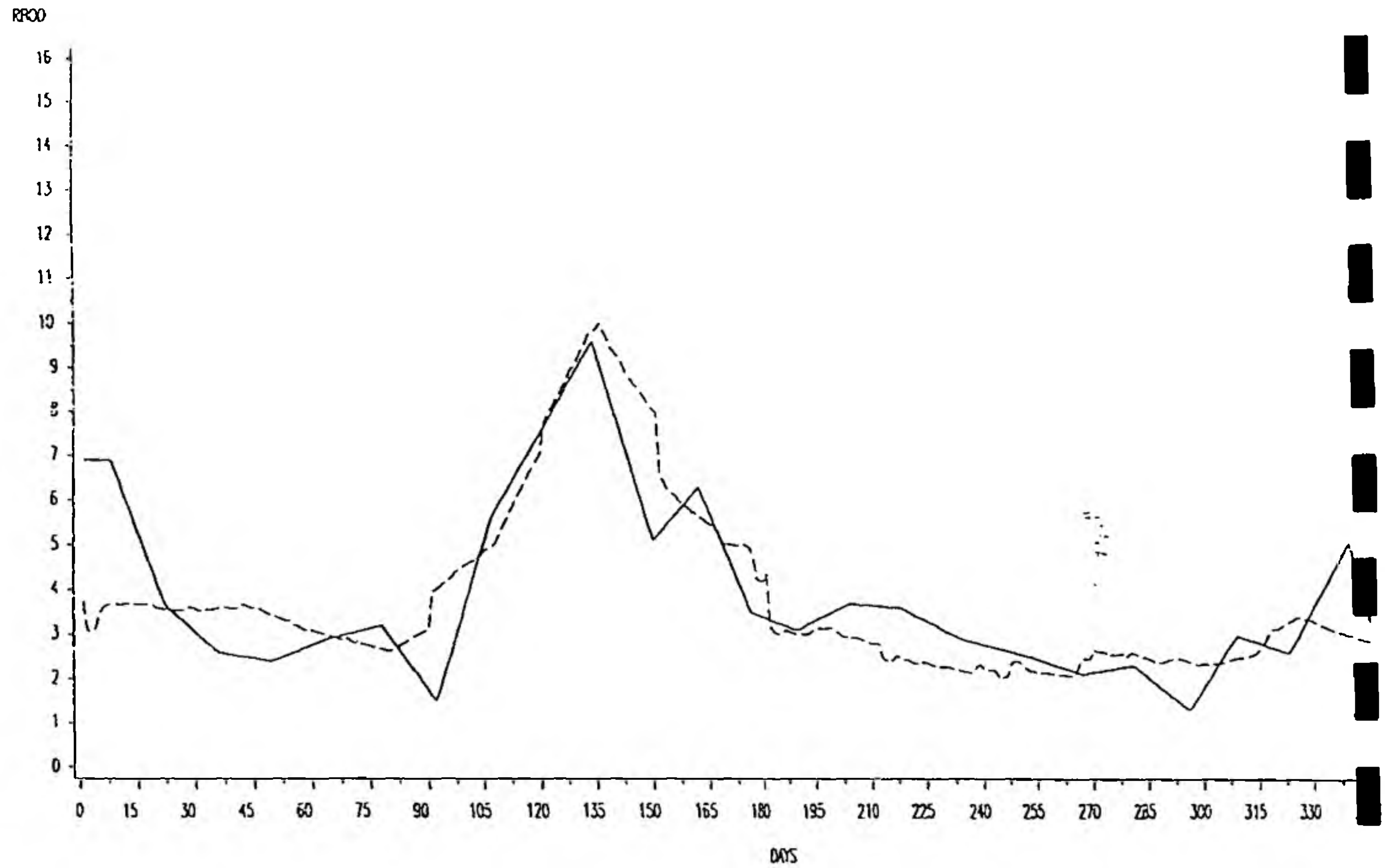


B O D at Walton

1974

Figure A3.1

Observed vs Modelled

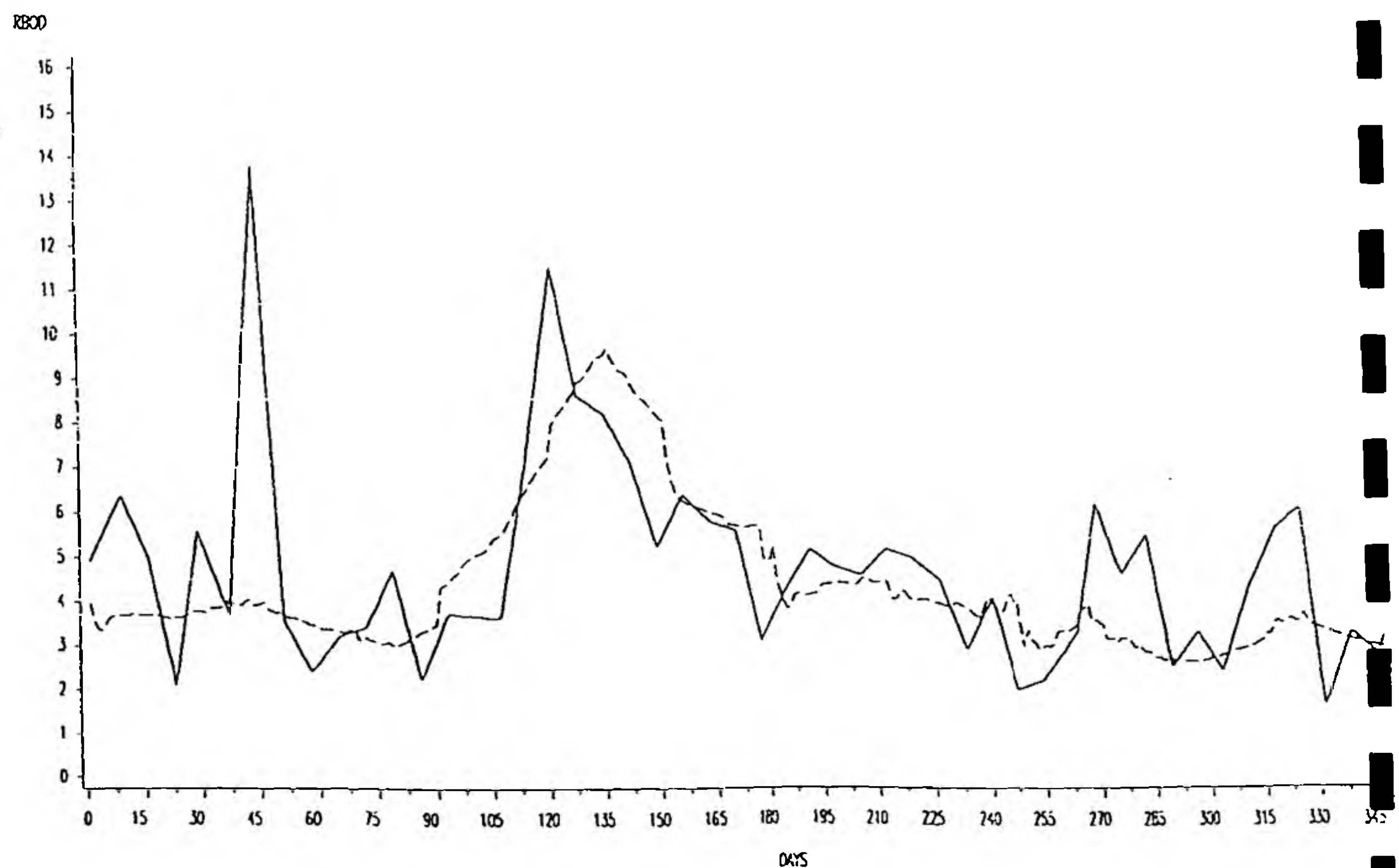


B O D at Teddington

1974

Figure A3.16

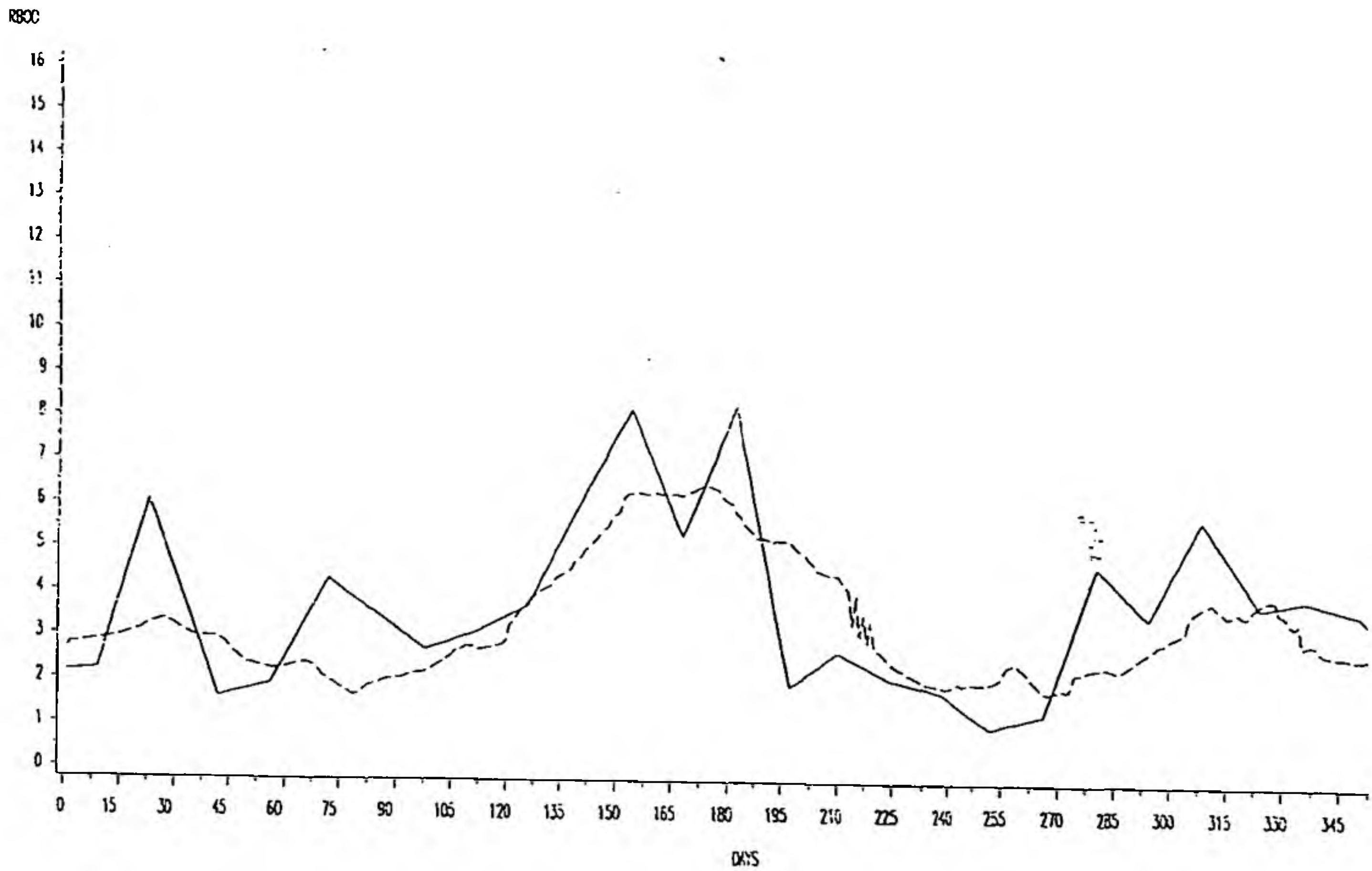
Observed vs Modelled



B O D at Datchet

1975 Figure A3.17

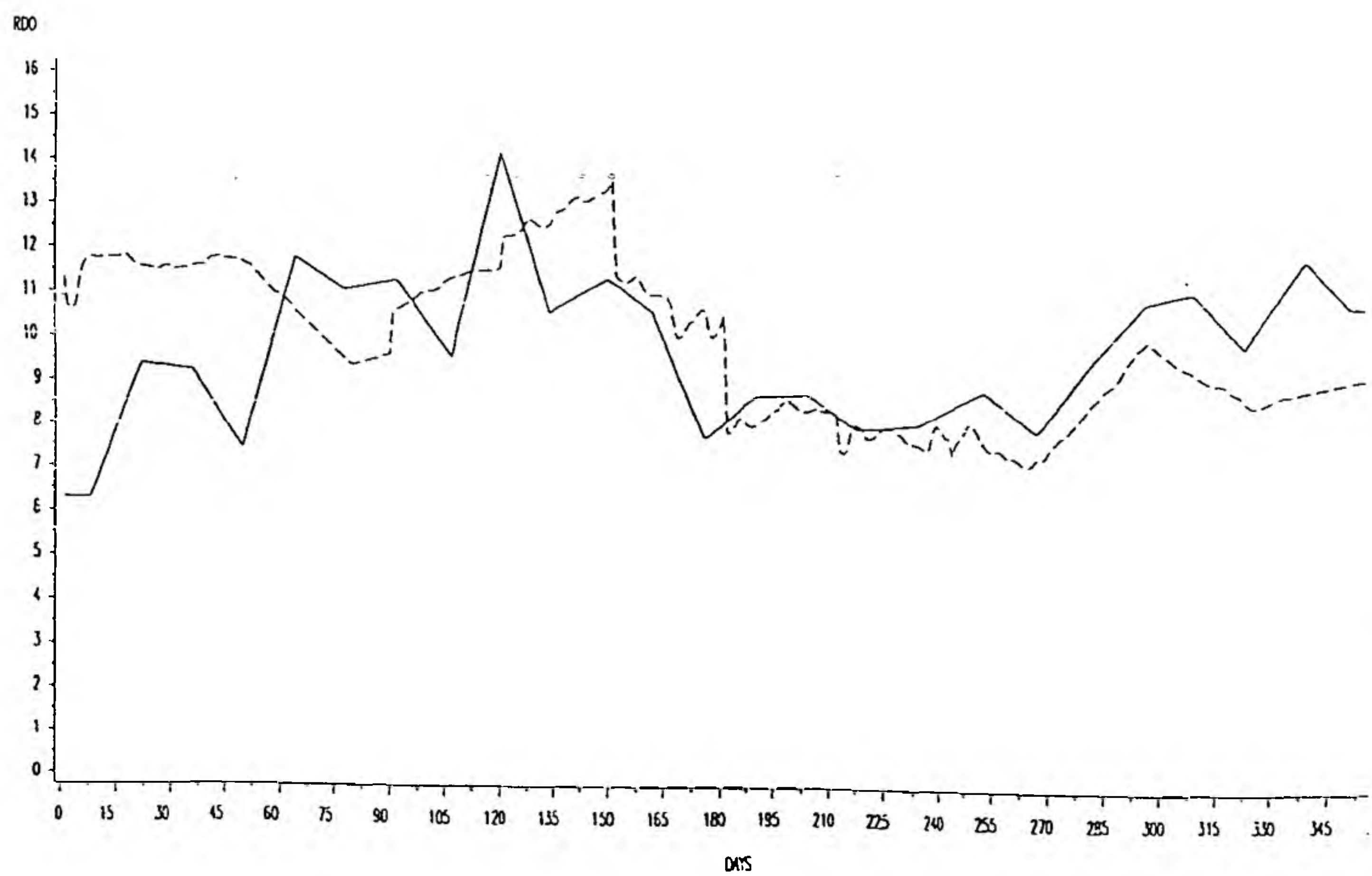
Observed vs Modelled



D O at Datchet

1974 Figure A3.18

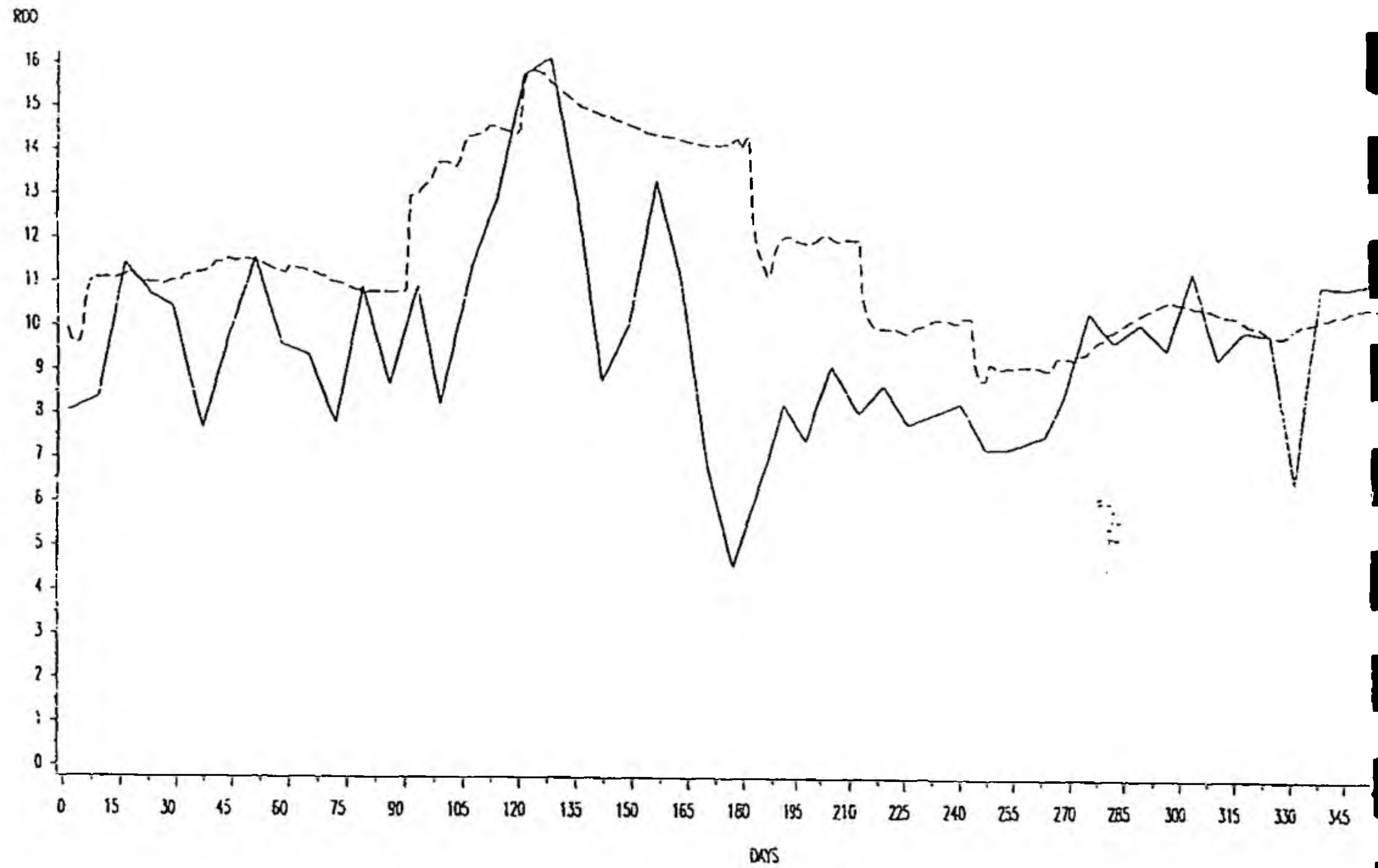
Observed vs Modelled



D O at Teddington

1974 Figure A3.19

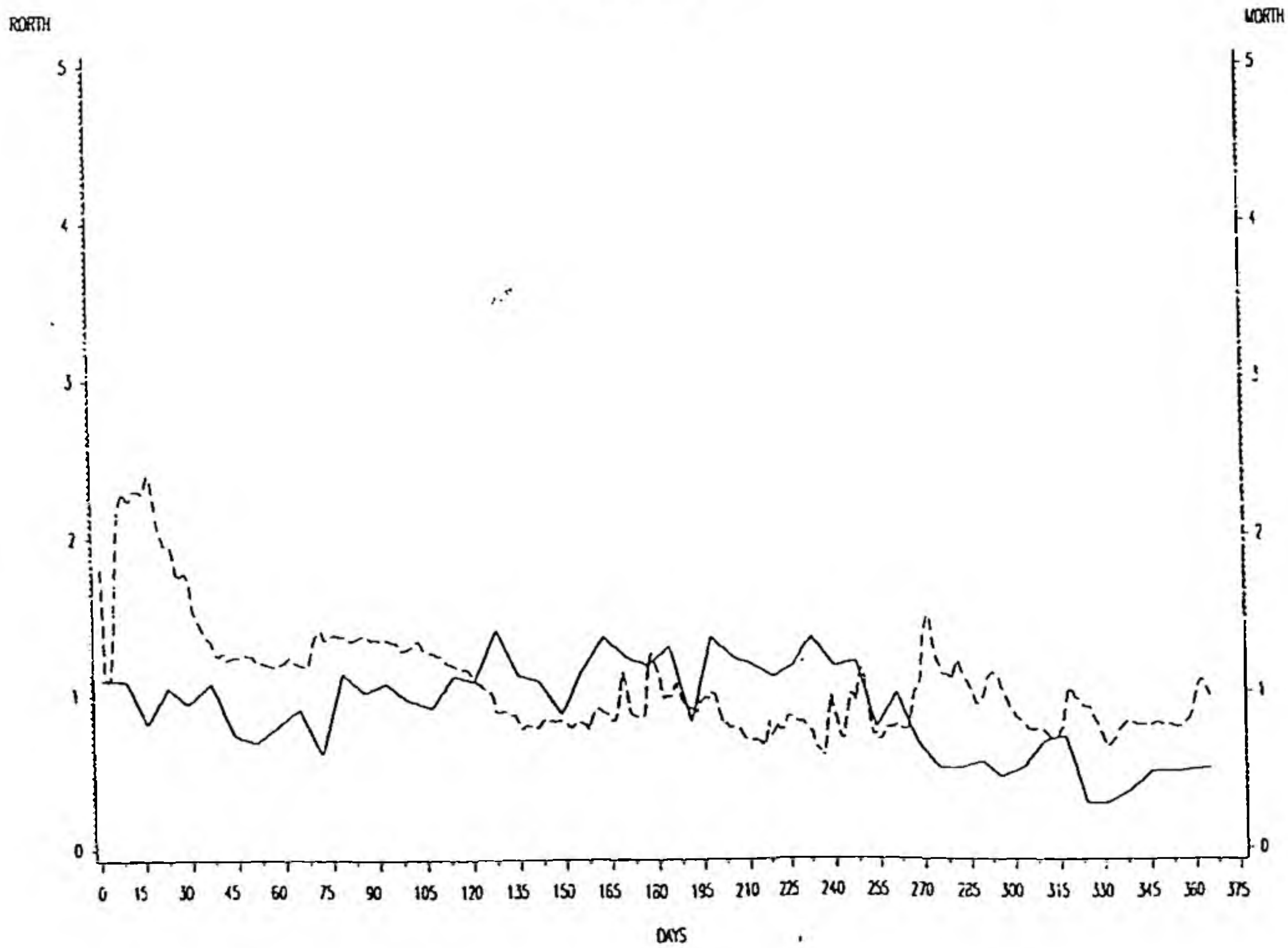
Observed vs Modelled



Ortho-Phosphate at Egham

1974 Figure A3.20

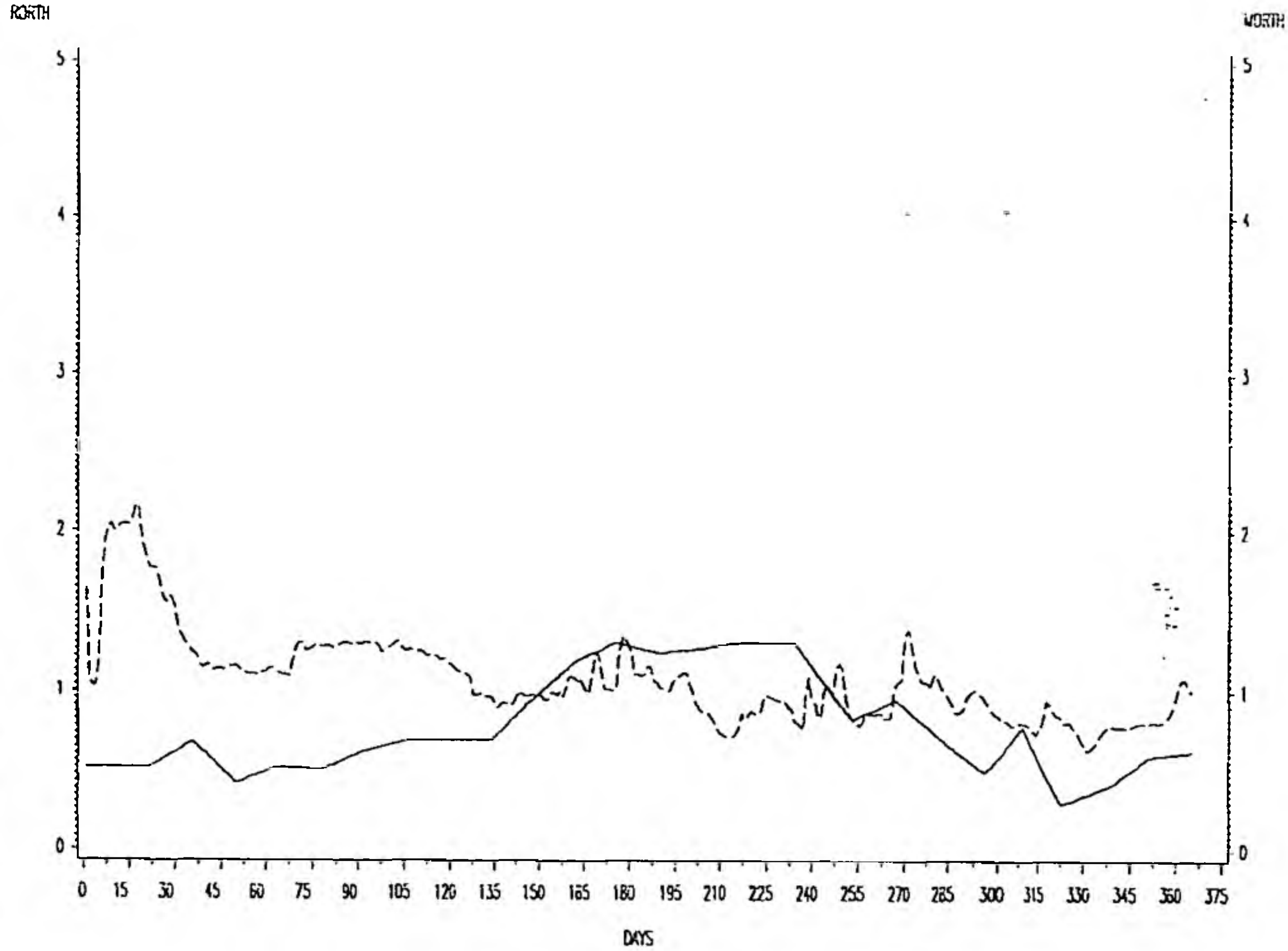
Observed vs Modelled



Ortho-Phosphate at Walton 1974

Figure A3.21

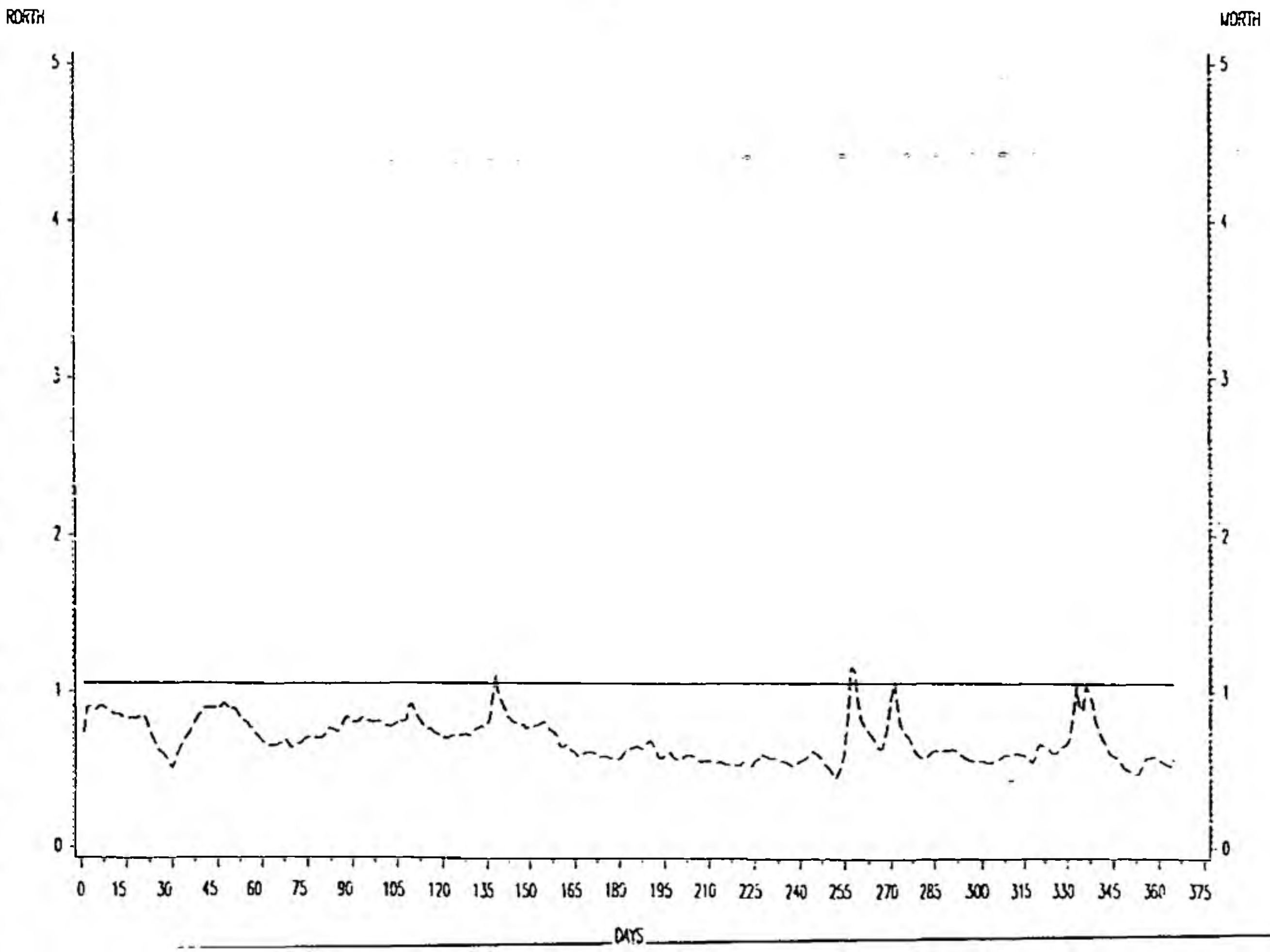
Observed vs Modelled



Ortho-Phosphate at Datchet 1975

Figure A3.22

Observed vs Modelled



Appendix 4

Groundwater flow and quality model

Appendix 4 Groundwater flow and quality model

R P S Clouston

A4.1 INTRODUCTION

A4.1.1

The flood channel will traverse the shallow gravel aquifer enclosed by a loop of the River Thames between Maidenhead and Eton. Water will be retained within the channel by means of retention structures or weirs, and the open water of the channel will be in direct contact with groundwater, with the possible exception of a section of channel adjacent to Manor Farm where the channel may be lined.

A4.1.2

Flows of groundwater from north to south across the line of the channel will constantly introduce water into the north bank of the channel, and recharge of the aquifer through the south bank is also predicted.

A4.1.3

In order to predict the eventual water quality in the flood channel and River Thames downstream it is therefore necessary to predict the flows of groundwater in the aquifer and the chemical nature of the water which may enter the channel. This is acheived by simulating the current physical and chemical characteristics of the aquifer and then assessing the consequences of constructing a channel within the model aquifer. This affects the flows by introducing fixed groundwater heads along the reaches of the channel, as determined by the design levels of the weirs.

A4.2 DESCRIPTION OF THE AQUIFER

A4.2.1

The solid geology of the area is shown in figure A4.2.1. The channel route crosses outcrops of the upper chalk, Reading Beds (varigated clay, locally sandy), and London Clay. The chalk outcrop occupies rising ground to the north and along the edge of the Thames at Taplow. An inlier of chalk and Reading Beds occurs at Windsor, and the downstream end of the channel crosses from the London Clay back onto the Reading Beds for a short section adjacent to the Thames. The chalk outcrops to the southeast here.

A4.2.2

Erosion by the River Thames has created a wide flood plain south of a line between Maidenhead and Slough, and this stretch of nearly flat ground is covered in sand and gravel deposits of recent origin to a depth of up to 7m. The Thames, whose level is regulated throughout the study area by weirs, is in hydraulic connection with the gravel aquifer which crosses beneath the river. The gravel aquifer is very permeable, and the water table to the north is partially controlled by river levels.

A4.2.3

The main inputs of water to the aquifer arise from infiltration, which on the level open ground will account for a high proportion of total precipitation. Relatively little flow appears to enter the aquifer from the rising ground to the north where the gravels thin out on the outcropping chalk and Reading Beds. Urbanisation in the areas of Maidenhead, Slough and Eton will reduce infiltration by intercepting rainfall and diverting this to the drain system.

A4.2.4

The Reading Beds and chalk are saturated beneath the terrace gravels. The Reading beds appear to behave as an aquitard whilst the chalk is an important groundwater resource, with important abstractions located at Taplow and Datchet. Studies of the Taplow abstraction which is adjacent to the River Thames indicate that some recharge is occuring from the river, however it has generally been assumed that there is little movement of water between the chalk and gravel aquifers where they are in direct connection.

A4.2.5

The gravel aquifer is used for public abstraction at Dorney, although a proportion of this water is drawn from the Thames. The channel will pass through Manor Farm which is associated with Slough Sewage Treatment Works and has been receiving sewage sludge for many decades. Effluents and leachates from the sludge beds pass quickly into the ground, and the aquifer is locally polluted by nitrate and ammonia. More widespread contamination by nitrate has resulted from the intensive arable and vegetable farming.

A4.3 THE GROUNDWATER MODEL

A4.3.1

The aquifer is shallow and lies in a fairly homogeneous granular matrix, making it suitable for modelling using Darcy's Law and in two dimensions. The AQUA computer program was used to simulate steady-state groundwater flows and chemical transport within the aquifer, both under the existing situation and with the channel in place.

A4.3.2

The model uses finite-elements, with a maximum number of 1,000 nodes being used to calculate flows and transport between these points. The areas between nodes are referred to as cells and are used in inputting areal values such as infiltration rates and transmissivity values. Nodes are spaced on a 100 m grid with additional nodes at the locations of boreholes and to define the river and channel boundaries.

A4.3.3

The central section of the aquifer, based upon Manor Farm has been previously modelled, and it was not possible to extend the model westwards and eastwards to cover the respective ends of the channel due to the physical limitations of the programme. Separate models were therefore created for the west and east sections, directly abutting the central section. The areas of the three models are shown on figure A4.3.1.

A4.3.4 Flow modelling

For flow calculations the model requires values for transmissivity and boundary flows or heads. Transmissivity figures for each cell were obtained from borehole records showing the depth of saturated gravel at various locations. The saturated aquifer thickness was estimated by subtracting the level of the base of the gravel from the level of the water table.

A4.3.5

Groundwater levels vary seasonally, and it is therefore important to know the date of records. Monitoring over the last three years has provided a good set of data for the western and central models, however monitoring in the east only commenced this year. A review of older records for the eastern area revealed considerable variation in the water table, and extrapolations and assumptions have had to be made for this area.

A4.3.6

Transmissivity is calculated from the product of the aquifer thickness and hydraulic conductivity of the aquifer in that location. Hydraulic conductivity measurements can only be accurately made through pumping tests, and the main source of values was the test conducted at Dorney which extended to most of the central area of aquifer. The Dorney pumping tests, conducted in 1976, found a mean aquifer transmissivity of 9,000 m^2/day, and given an average aquifer thickness of 5 m gives a saturated hydraulic conductivity of 0.02 m/s.

A4.3.7

Variations in the composition of the aquifer matrix, in terms of variable levels of fine sand, silt, and clay in the gravel interstices will influence the transmissivity. A 20 m wide zone of low transmissivity was used to simulate mud in the base of the River Thames. This was extrapolated from a 3 m wide zone with a hydraulic conductivity of 5 X 10^{-6} m/s which was within the range estimated by the Dorney test. The transmissivities adopted for modelling are shown in Figure A4.3.2.

A4.3.8

Boundary conditions are divided into those which apply to the River Thames (used as one boundary of all models) and those which apply to the inland edges of the models. The Thames boundary is fixed at the level of weirs, as determined by the requirements for navigation; minimum summer levels can thus be used to define this boundary. Along mutual boundaries, the outflows from one model have been used to define inflows on the adjacent model. Elsewhere, flows have been calculated from the observed aquifer gradient and estimated transmissivity.

A4.3.9

Abstractions at Dorney and Taplow have been modelled. The Dorney abstraction takes water directly from the gravels, with about 36% percent being derived from the River. An abstraction rate of under 0.2 cumecs was used. The model indicates that between 31% and 43% will be derived from the River depending upon the rate of pumping used. The abstractions at Taplow and Datchet draw water largely from the chalk and have not been simulated. The of effect on the gravel aquifer is minimal due to the very limited thickness of the deposit in this area and proximity to the River.

A4.3.10

The channel is modelled as a 50 m wide high transmissivity zone whose south bank has a head level fixed to that of the weir downstream. Retention levels are given in table A4.3.1. The AQUA model cannot simulate open water flows and therefore it is assumed that there is no flow into the channel from the river, and there is virtually no flow over weirs. The change in head across the weir structures may be up to 2 m and this will induce substantial groundwater gradients in the surrounding aquifer, introducing a component of flow to the east which cuts across the more general north-south groundwater movement. The influence of weirs will be localised, although the retained water levels of the channel will have a general influence upon the water table at times of the year when groundwater levels would naturally be at their highest.

A4.3.11 Transport modelling

Chemical transport modelling has placed emphasis on nitrate, which is assumed to behave conservatively i.e. is not subject to chemical or biological removal. Main nitrate sources are infiltration water, particularly beneath intensively farmed areas, the River Thames, and Manor Farm. Figure A4.3.3 shows the nitrate inputs to the aquifer which have been assumed. These are based upon known groundwater and river concentrations, and estimates of the infiltration concentrations likely to be found.

A4.3.12

Phosphate was not modelled since other studies have indicated that it is strongly bound in aquifers, probably in calcium compounds. Groundwater concentrations are generally below 0.1 mg/l, and average between 0.01 and 0.001 mg/l.

A4.3.13

Ammonium was modelled in the vicinity of Manor Farm, although some uncertainty exists due to the strong adsorption of ammonium to the aquifer and retention which occurs, and the biologically-mediated interchange with nitrate. Ammonia concentrations are in a pH-mediated equilibrium with ammonium, and are estimated to be less than 1mg/l in the alkaline groundwater.

A4.3.14

Lateral and longitudinal dispersion values of 10 m and 100 m have been used respectively. These resemble the observed values from a shallow gravel aquifer injected with sewage effluent. Denitrification is likely to reduce nitrate levels, particularly under Manor Farm where the necessary carbon is found in association with nitrate contamination. This process has not been modelled to remove nitrate from the groundwater, and therefore the estimates of nitrate concentrations are a worst case. In areas of low groundwater flow the accumulation of nitrate is predicted in a way which would not be realistic if denitrification occurs.

A4.3.15 Special conditions

A number of particular features have been looked at in previous studies and are referred to here.

In some areas, the channel will descend to the base of the aquifer, and will thus intercept all groundwater flowing in to the open water body. If silting occurs within the channel then the resulting loss of permeability of the channel bed may restrict the extent of the recharge to the aquifer. The result would be to divert the general pattern of north-south groundwater flow into the channel flow from west to east. This phenomenon has been modelled by applying a reduced transmissivity layer to the south bank of the central channel model. The transmissivity reduction was in the order of 0.02 times that of the adjacent aquifer block in a layer modelled as being 25m wide.

In other areas the groundwater flow will remain able to pass beneath the channel in undisturbed alluvium. Lining of the channel to prevent the ingress of polluted groundwater has been examined and this could be adopted in the areas of Manor Farm and the agricultural land to the west. It is necessary to leave a sufficient depth of high permeable material beneath the channel to transmit the water and prevent damming and possible surface flooding upstream.

The main body of polluted groundwater lies directly to the south of the channel at Manor Farm, and this has been shown to be displaced southwards with the general groundwater flow over a matter of months. There is some field data from Manor Farm which may support this prediction.

A4.4 MODEL VALIDATION

A4.4.1

An estimate of the accuracy of the aquifer simulation is given through the level of agreement between observed and predicted groundwater levels and concentrations (see Table A4.4.1). Good information is generally available for the former, and the western and central models predict groundwater levels at locations along the channel and at monitoring boreholes which are nearly all within 1 standard deviation of the observed levels, giving an acceptable level of fit. The worst fit was found at borehole W31 where the difference was 0.12 m or 1.3 standard deviations.

A4.4.2

Values for the eastern model area depart further from the observed levels, with a maximum difference of 1.22 m found at borehole C33. The lack of reliable monitoring data and geological complexity of this area mean that less confidence can be placed in the predictions for this area than for the central and western sections. Early modelling of this stretch suggested a tendency for groundwater to be retained at much higher than observed levels north of the river. Gradients were later reduced by increasing the permeability of the mud zone along the river bank. Fortunately this section is also likely to have the least significance in terms of channel flows and water quality, due to its short length and proximity to the River Thames.

A4.4.3

The model was calibrated against nitrate as a mobile nutrient. It is assumed that a steady state exists between nitrate inputs and the concentrations measured in groundwater. Comparisons between the predicted and observed nitrate concentrations are given in Table A4.4.2. The predicted concentration for the Dorney boreholes is 15 mg/l which compares well with the figure of 13 mg/l taken from an unpublished hydrogeological map of the area.

A4.5 PREDICTIONS OF GROUNDWATER EFFECTS

A4.5.1

Figures A4.5.1 to A4.5.3 show the groundwater flows in each aquifer section under normal steady state conditions. The degree of matching of levels between the west and central sections is good, but between the central and east sections is poor, reflecting the difficulty experienced in calibrating the east section with the limited data available.

A4.5.2 Channel effects on groundwater

The changes to the groundwater flow patterns caused by placing a fixed head channel are illustrated in Figures A4.5.4 to A4.5.6. The groundwater movements are locally influenced by the weirs. Calculations of the estimated flows across the channel banks have been made, and for this purpose the channel has been divided into reaches between weirs based upon whether the flows across each model node were into or out of the channel. The net flows are out of the channel immediately upstream of weirs, and into the channel downstream. Flows across each reach are given in Table A4.5.4. Within each reach, where flows in exceed those out of the aquifer then the channel is behaving as a source of water recharging the aquifer. Where the reverse occurs then the channel acts as a sink for water discharged from the aquifer. Over the channel as a whole, the net inflow is 1.319 m^3s^{-1} whilst the net loss of water from the channel is 1.02 m^3s^{-1}, giving an increase in channel flow along its length of about 0.3 m^3s^{-1}.

A4.5.3 Nitrate fluxes

Modelled nitrate concentrations in groundwater are shown in figs A4.5.7 to A4.5.9. The mass loadings to each section of the channel can be calculated as the product of the concentration and the groundwater flow. This will be a worst case estimate since a proportion of groundwater will flow beneath the channel without entering the open water.

It is also considered that biological activity within the channel water and silt zones will serve to reduce nitrate concentrations in recharge water to the aquifer where the groundwater nitrate is naturally at a higher concentration than in the Thames intake. Thus, if sweetening flows are maintained from the Thames, nitrate levels could be raised in parts of the aquifer being recharged towards the level of about 7 mg/l found in river water.

A4.5.4 Effect of silting

The consequences of silting are difficult to predict accurately due to the problems of modelling the interaction between open water and groundwater systems and the limitations of a groundwater model in this province. The simple consequence of reducing transmissivity of the southern bank of the channel is to divert water flowing in from the north to the east. As a result groundwater levels and gradients will drop to the south and flows reduce.

In practice, silting might be most likely towards the upper end of the channel and in the areas, particularly just upstream of weirs, where recharge to the aquifer occurs. The most noticeable effect will be where the groundwater flow is perpendicular to the channel flow, the central section being most important in this respect.

The compensating effect of drainage through the aquifer beneath the channel also means that the impact of silting should be small in the aquifer south of the channel.

A4.5.5 Effect on abstractions

The major abstractions taking water from the shallow aquifer are located at Dorney, at least 1 km from the channel route. No physical effect of the channel on groundwater at this abstraction is predicted, with local water table maintained at its current level. The model predicts that there would be insignificant changes in nitrate levels at the abstraction. Other major abstractions in the chalk should be unaffected by the channel.

A4.5.6 Impact of channel lining

The possibility of lining the flood channel to prevent the entry of nitrate contaminated groundwater has been raised. This would require the construction of sub-channel drains, or retention of alluvium beneath the channel base. Model predictions are that groundwater levels upstream would not be significantly raised if a minimum 1 m thick deposit of sand/gravel were retained.

The minimum transmissivity would need to be 0.04 ms^{-1}, and while this is typical of clean granular alluvium such as that found in the study area, silting could reduce this over time leading to rises in groundwater levels to the north of the central section of the channel. Rises of 0.5 m in winter could lead to flooding.

In terms of chemical improvements to the channel water, the dilution effects if substantial sweetening flows occur would be so large as to make the groundwater contribution negligible.

A4.6 CONCLUSIONS

A4.6.1

The groundwater flows into the channel are generally of a small order and will only be significant if there is little or no flow directed into the channel from the River Thames. The largest flows are found in sections where the channel traverses the direction of groundwater movement at a large angle. This is seen primarily in the central section of the channel which runs west-east across a north-south groundwater flow.

A4.6.2

The model predicts an approximate balance between groundwater inflows and outflows to the channel, with highest movements being seen around the weir structures where steep gradients are imposed upon the water table. Although surface water movements cannot be adequately simulated using this model, it is predicted that silting of the outflow bank of the channel will reduce recharge to the aquifer and increase the quantity of flow moving eastwards along the channel. This will have little effect upon the water quality of the channel especially if flows are introduced from the Thames.

A4.6.3

Nitrate is the primary contaminant of concern, and is present in a concentrated plume across the western side of the central section, north of Dorney. A further nitrate plume is seen immediately south of the channel at Manor Farm. This latter is of greater concentration but of smaller areal extent and apparently more transient nature than the agricultural nitrate plume to the west. Maximum concentrations in the order of 20 mg/l nitrate are predicted to enter the channel from groundwater. The effects on water quality within the channel are principally a function of the nitrate concentration of the groundwater and the net inflow, and these areas of greatest nitrate accumulation will also have low flows into the channel.

A4.6.4

Phosphate is not a major contaminant and is found at low concentrations within the aquifer due to its immobility. Maximum flows of 0.1 mg/l phosphate are predicted to enter the channel from groundwater.

A4.6.5

Ammonia is only of significance in the area of Manor Farm where concentrations in the order of 100 mg/l nitrate or 100 mg/l ammonium may be found depending upon the redox state of the goundwater. Under the reduced ammonium regime, the concentration of unionised ammonia is predicted to remain below 1 mg/l. Field data suggest that these concentrations are subsiding due to biological activity and dispersion of the pollution plume following the cessation of sewage spreading operations on Manor Farm. Transient modelling of the observed plume suggests that this will move southwards and away from the channel at over 1,000 m per year.

A4.6.6

Lining of the channel in the areas of polluted groundwater would be technically feasible providing that adequate sub-channel drainage were provided. A risk of increased groundwater levels on the north bank and possible flooding would arise if reduction of the permeability of the drains were to occur through silting. There is no overriding argument for lining the channel if groundwater flows are substantially exceeded by inflows from the Thames.

A4.6.7

The groundwater model predicts that there will be no significant impacts from the channel upon the quality or quantity of water at public abstractions.

Table A4.3.1 Channel retention levels

Reach		Level (m O.D.)
1	Thames - Taplow Mill	23.45
2	Taplow Mill - Marsh Lane	21.00
3	Marsh Lane - Dorney	20.30
4	Dorney - Manor Farm	19.50
5	Manor Farm - Slough Road	18.50
6	Slough Road - Black Potts Viaduct	17.00
7	Black Potts Viaduct - Thames	16.12

Table A4.4.1 Validation data (i) groundwater levels

Monitoring Borehole	Observed Levels Mean	Std. Dev.	Predicted	Difference
West Section				
C9	20.94	0.28	21.10	-0.16
C11	21.01	0.24		
C12	21.13	0.21		
C13	21.19	0.26	21.16	0.03
C14	21.18	0.16	21.07	0.11
W15	21.13	0.20	21.06	0.07
C16	20.93	0.18	20.74	0.19
W17	20.82	0.23	20.76	0.06
C18	20.58	0.39	20.56	0.02
Centre section				
C24	20.34	0.38	20.16	0.18
C25	19.99	0.29	19.86	0.13
W27	19.26	0.19	19.42	-0.16
W28	19.18	0.36	19.21	-0.03
W29	19.17	0.11	19.29	-0.12
W30	18.60	0.17	18.77	-0.18
W31	18.36	0.09	18.24	0.12
W32	18.48	0.21	18.51	-0.03
East Section (one datum)				
C31	18.38		17.92	0.46
C33	18.82		17.60	1.22
C35	16.47		16.95	-0.48
C43	16.38		16.27	0.11

Table A.4.4.2 Model validation data (ii) nitrate

Borehole	Measured	Predicted	Difference (mgN/l)
19	17.4	20	2.6
20	17.7	20	2.3
21	28.3	19	-9.3
22	20.8	20	-0.8
23	18.6	18	-0.6
24	2.4	15	12.6
25	1.6	11	9.4
26	6.6	16	9.6
27	3.7	18	14.3
28	8.9	17	8.1
29	9.3	6	-3.3
30	6.4	5	-1.4
31	7.5	7	-0.5
32	17.1	7	-10.1

Table A.4.5.4 Calculated flows across boundaries

	Reach (grid refs)	Inflow (m^3s^{-1})	Outflow
West Section (m^3s^{-1}) North Bank			
	90657,83324 to 90448,82039	-	0.095
	90445,82020 to 91060,80790	0.088	-
	91108,80767 to ,80378	-	0.084
	91373,80378 to 91800,80200	0.100	-
	91855,80050		0.004
		0.188	0.182
West Section South Bank			
	90495,82946	>0.001	-
	90489,82903 to 90388,82020	-	0.075
	90400,82000 to 90670,81100	0.057	-
	90680,80999 to 91297,80400	-	0.075
	91301,80385 to 91450,80300	0.065	-
	91500,80300 to 91800,80050	-	0.005
		0.122	0.156
Central Section North Bank			
	(reach N1)	0.068	-
	(reach N2)	-	0.033
	(reach N3)	0.110	-
	(reach N4)	-	0.060
	(reach N5)	0.211	-
	(reach N6)	-	0.019
		0.389	0.112
Central Section South Bank			
	(reach S1)	-	0.184
	(reach S2)	0.055	-
	(reach S3)	-	0.082
	(reach S4)	0.091	-
	(reach S5)	-	0.019
		0.146	0.285
East Section North Bank			
	97450,78125 to 97610,78860	0.156	-
	97600,78870 to 97457,78961	-	0.105
	97399,78979 to 96000,79070	0.124	-
		0.280	0.105

Table A.4.5.4 continued

East Section			
South Bank			
	96000,79020 to 96500,78900	0.026	-
	96600,78895 to 97550,78825	-	0.180
	97566,78817 to 97350,78148	0.168	-
		0.194	0.180

Geological map

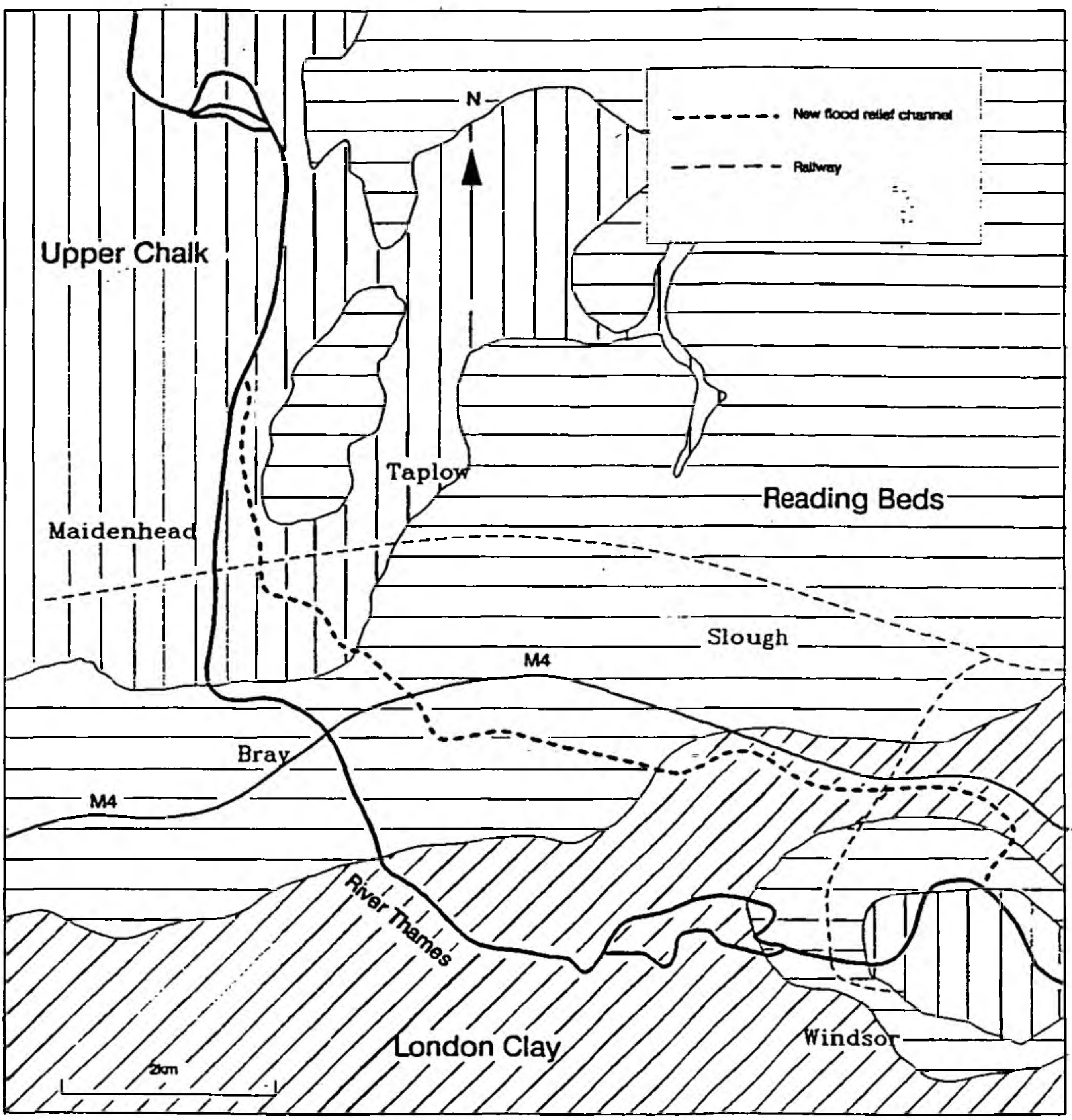


Figure.A4.2.1

Areas of the Groundwater Model

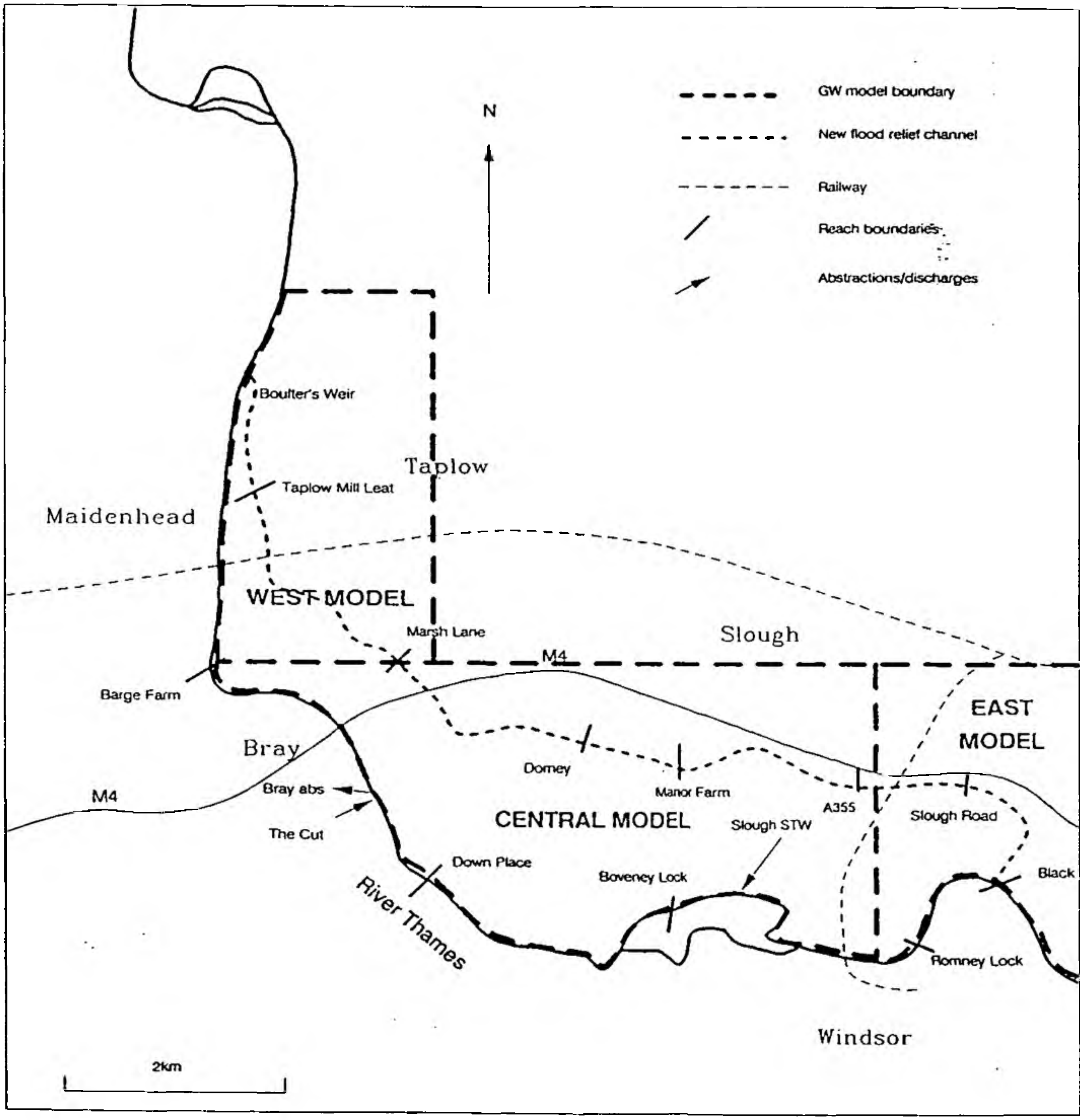
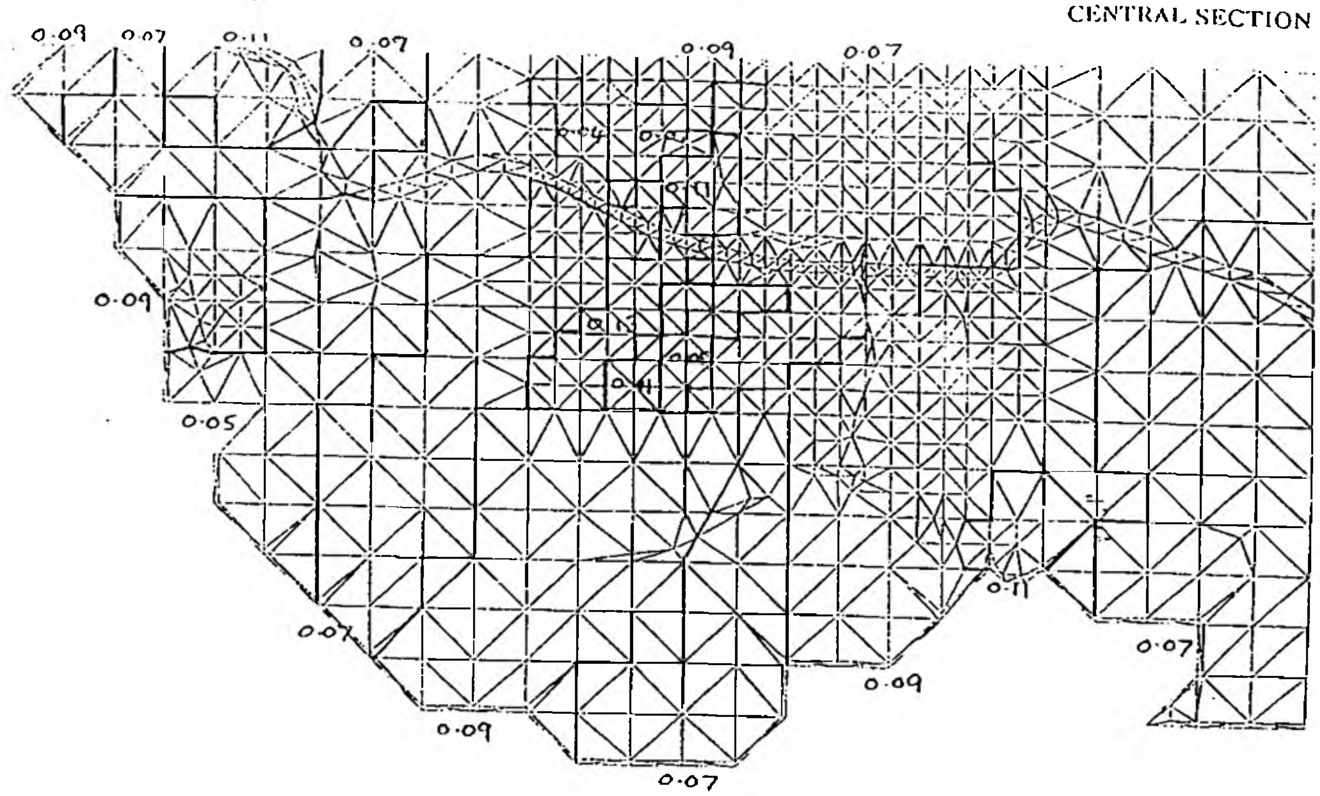


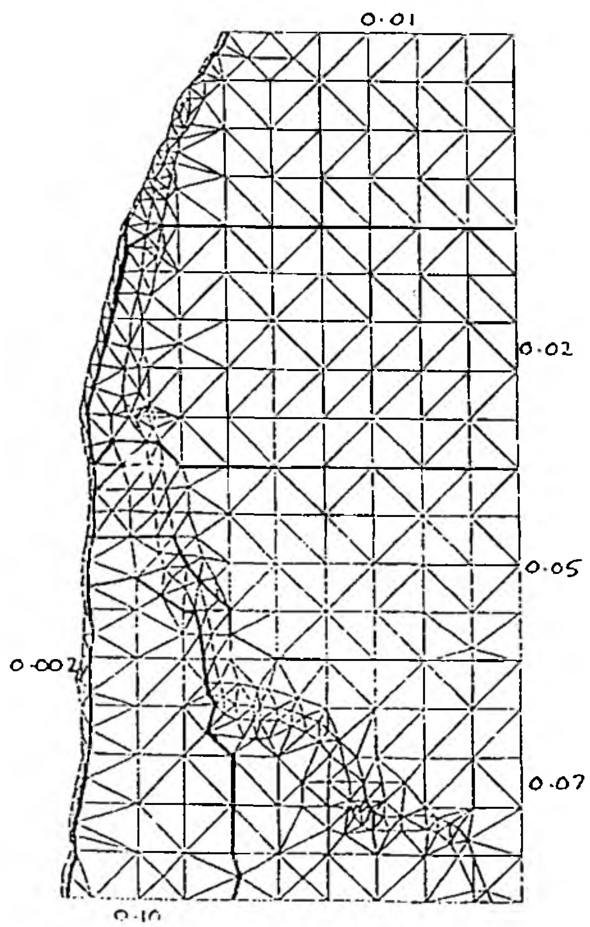
Figure A4.3.1

A 4.3.2 Transmissivity Data for Aquifer Model

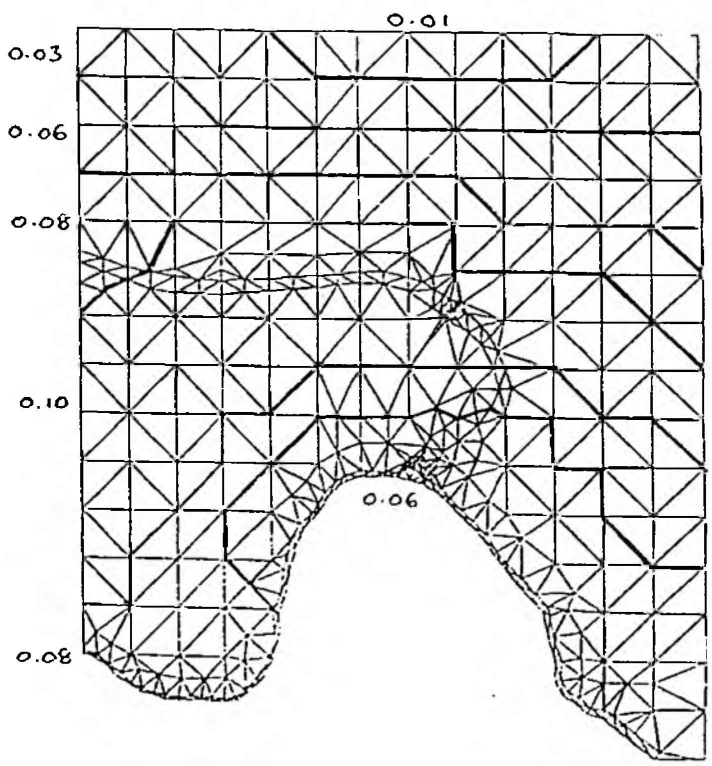
CENTRAL SECTION



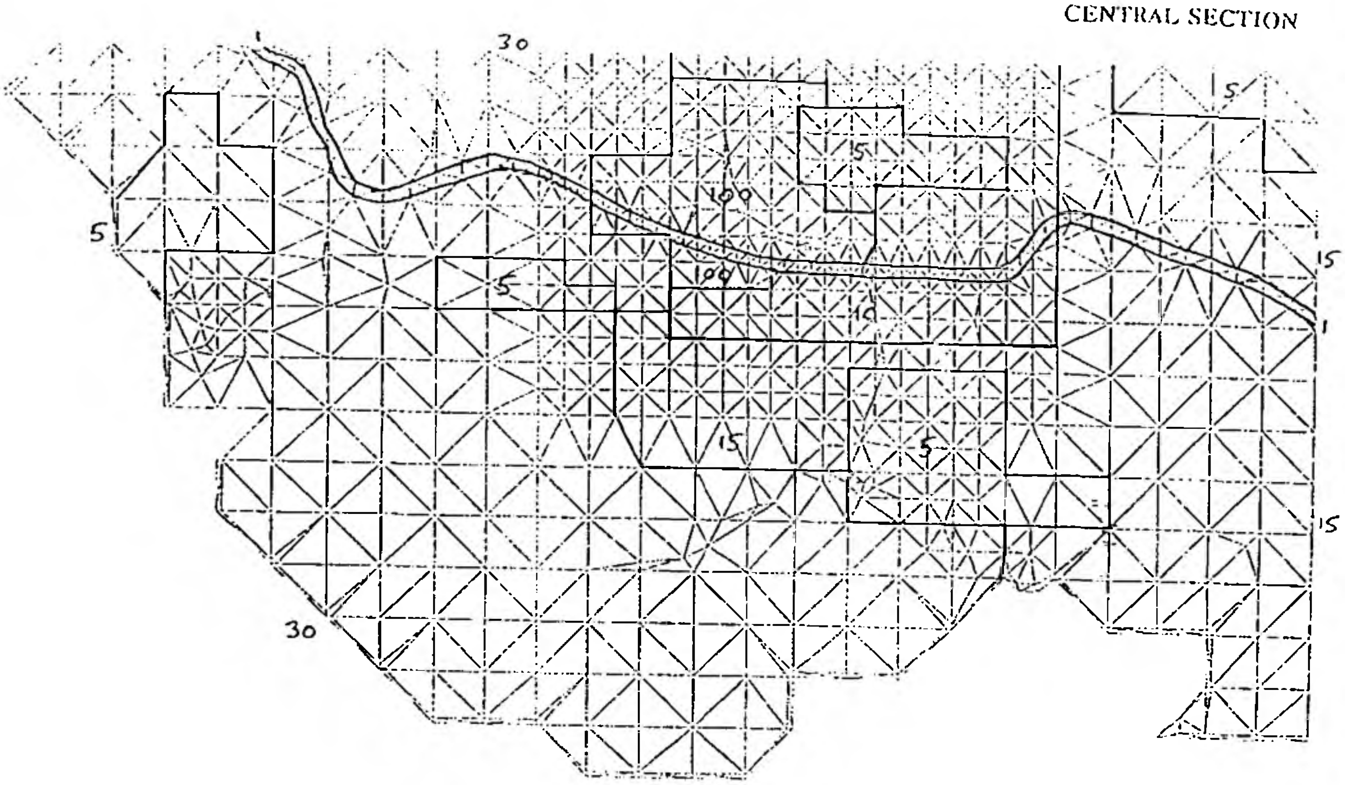
WEST SECTION

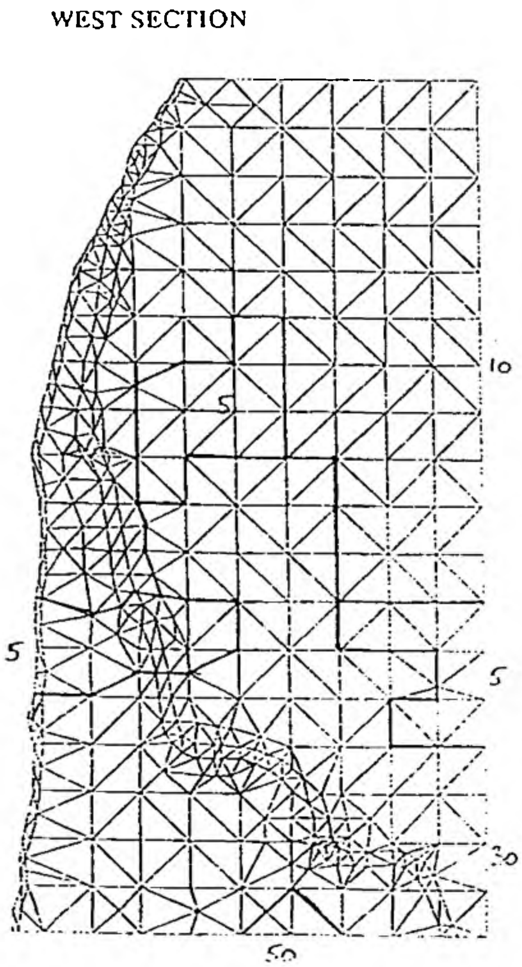


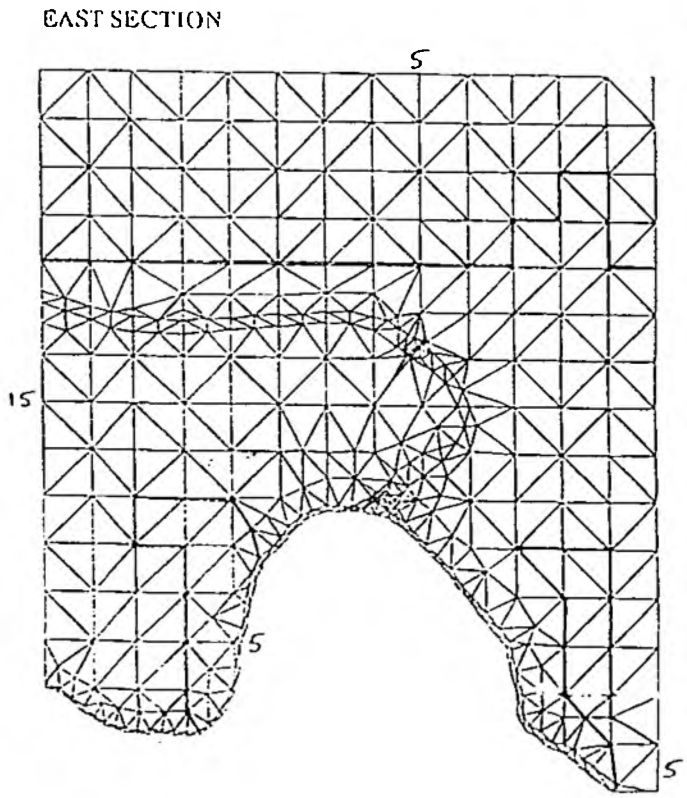
EAST SECTION

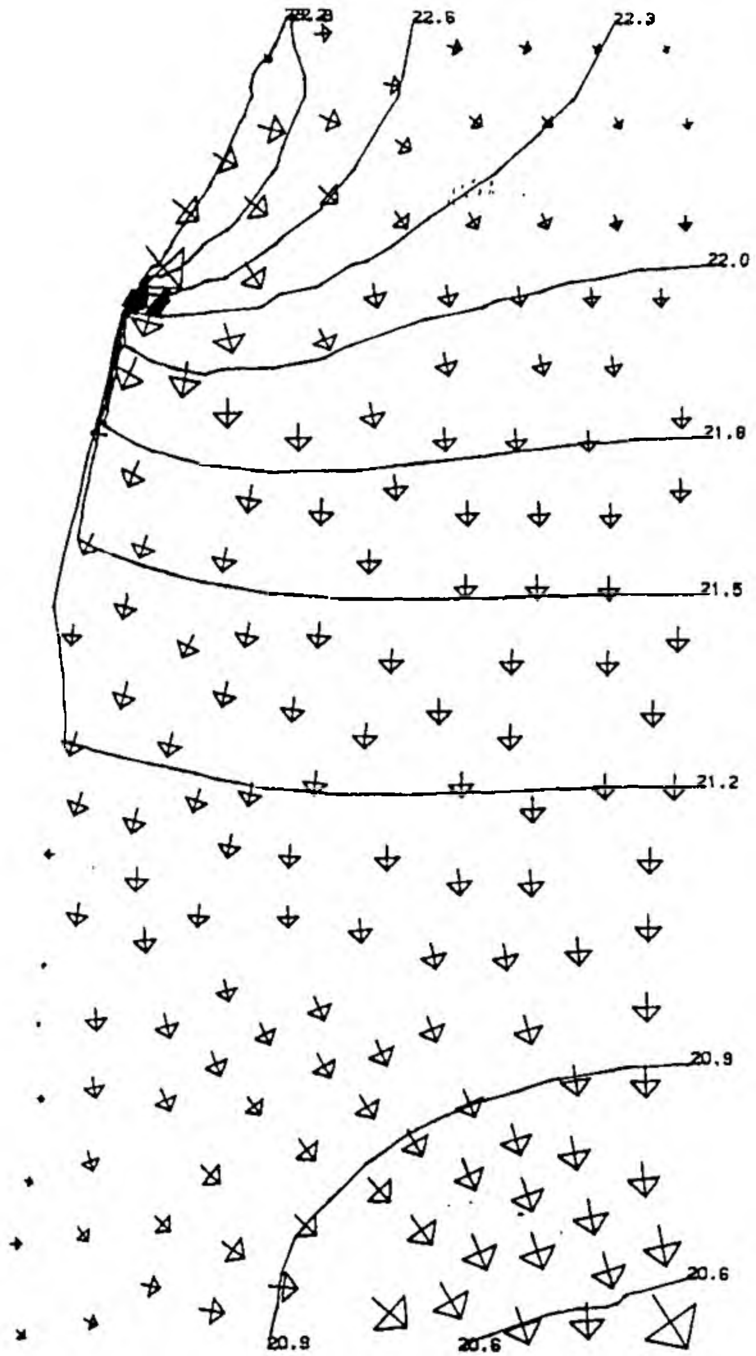


A4.3.3 Nitrate Infiltration Rates for Aquifer Model








22.2
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21.2
20.9
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20.9
20.6

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A 4.5.1

Groundwater Levels, No Channel, West Section.

Groundwater contours (m aod)

A 4.5.2

Groundwater Levels, No Channel, Central Section.

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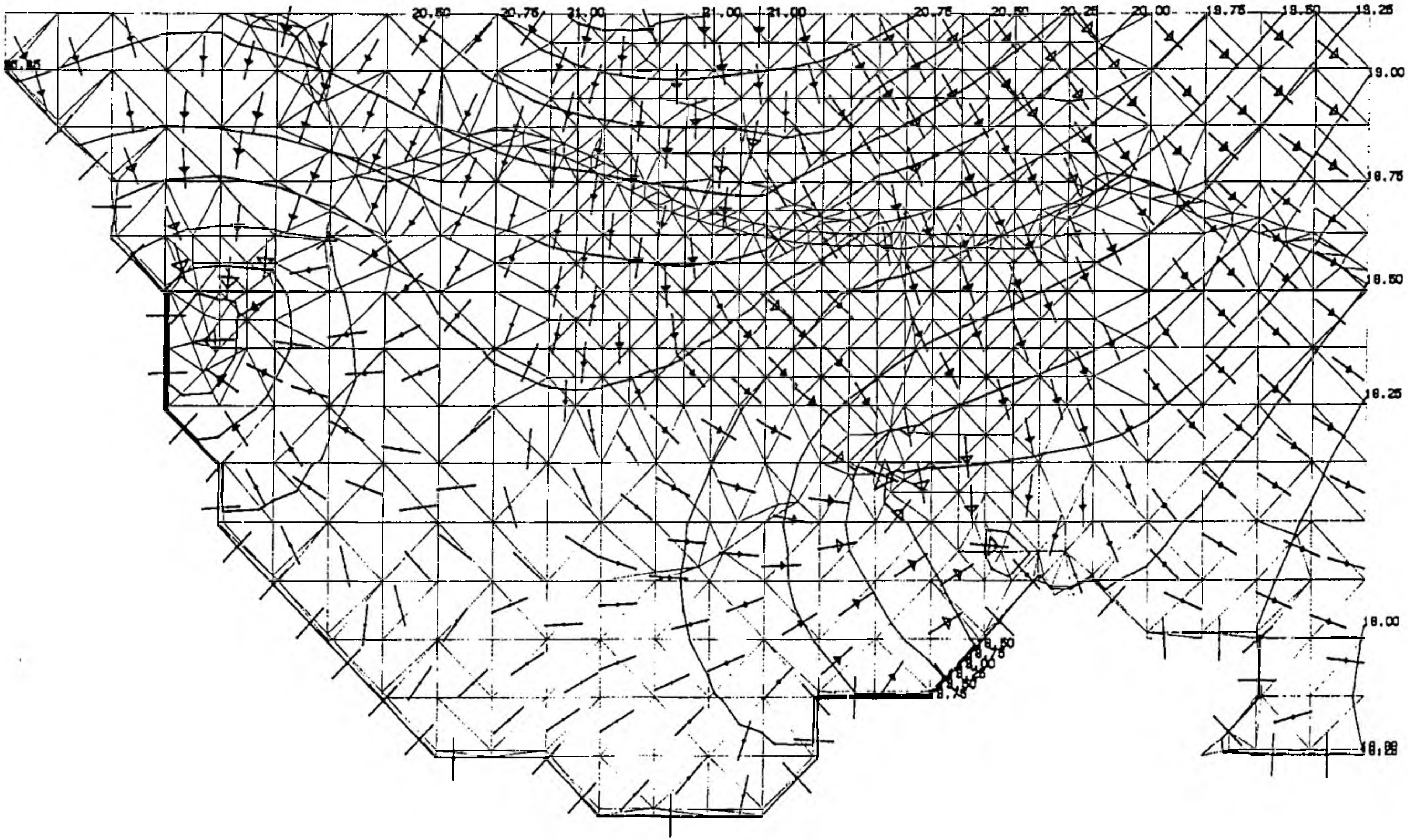
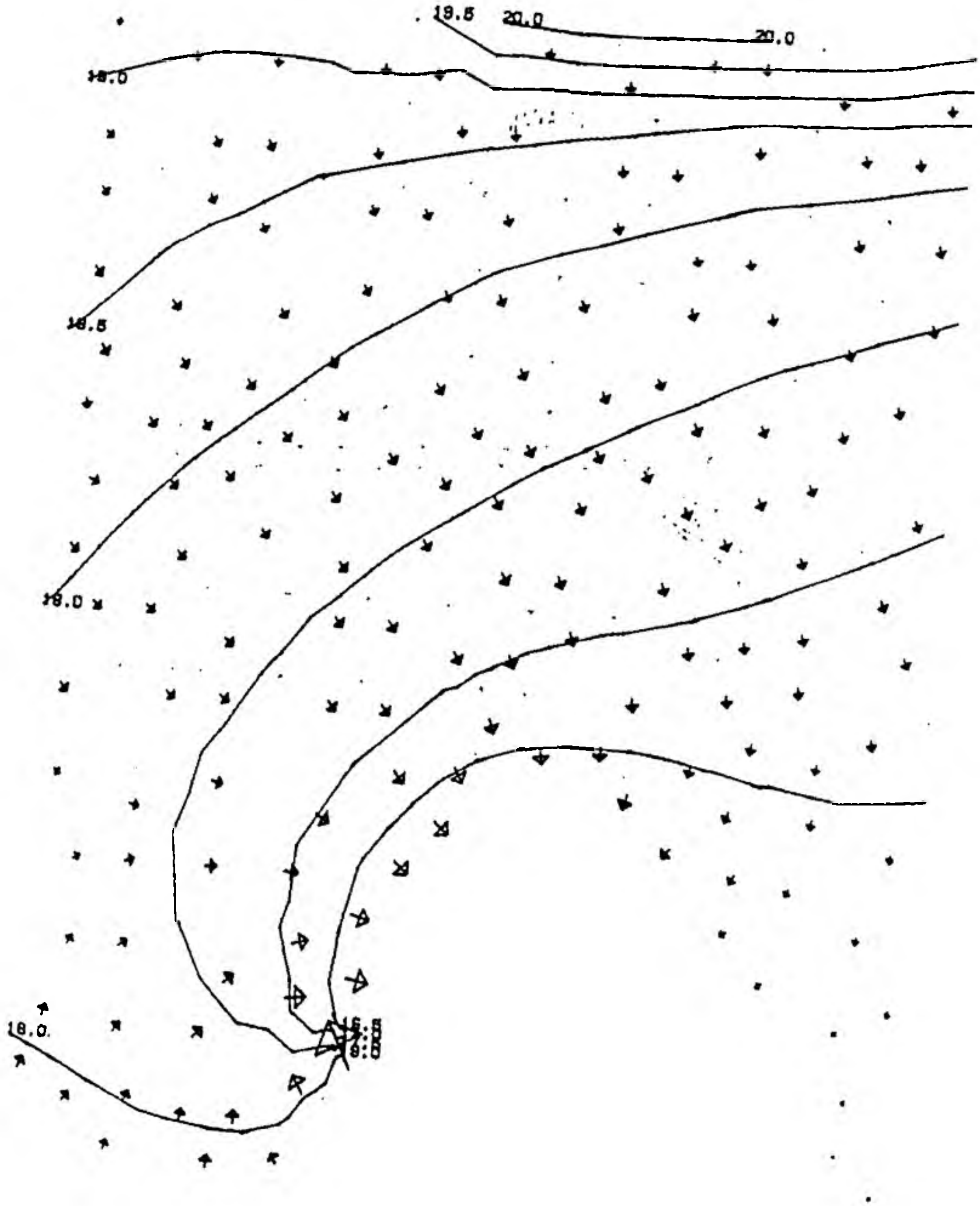
Groundwater contours (m aod)




19.5
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20.0
18.5
18.0

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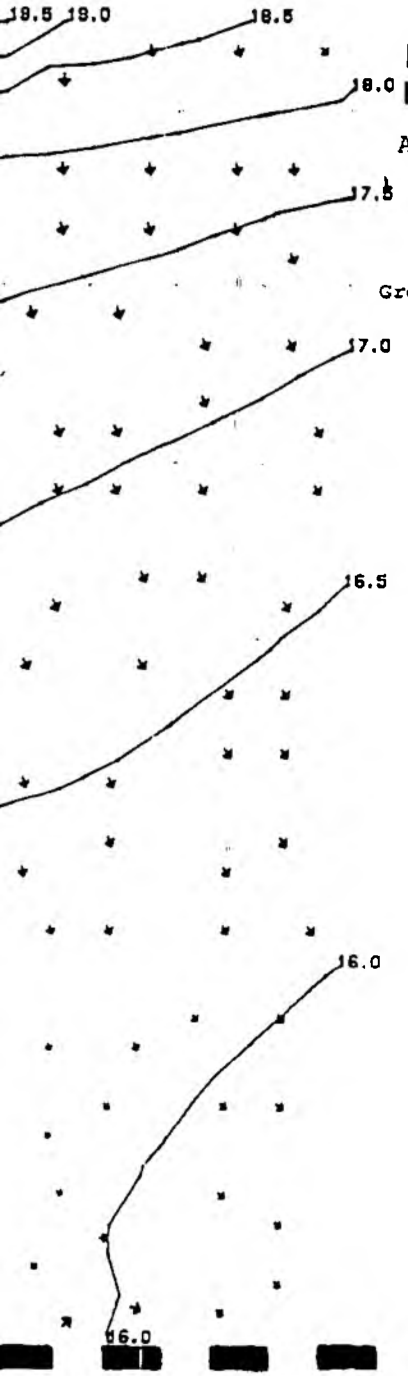
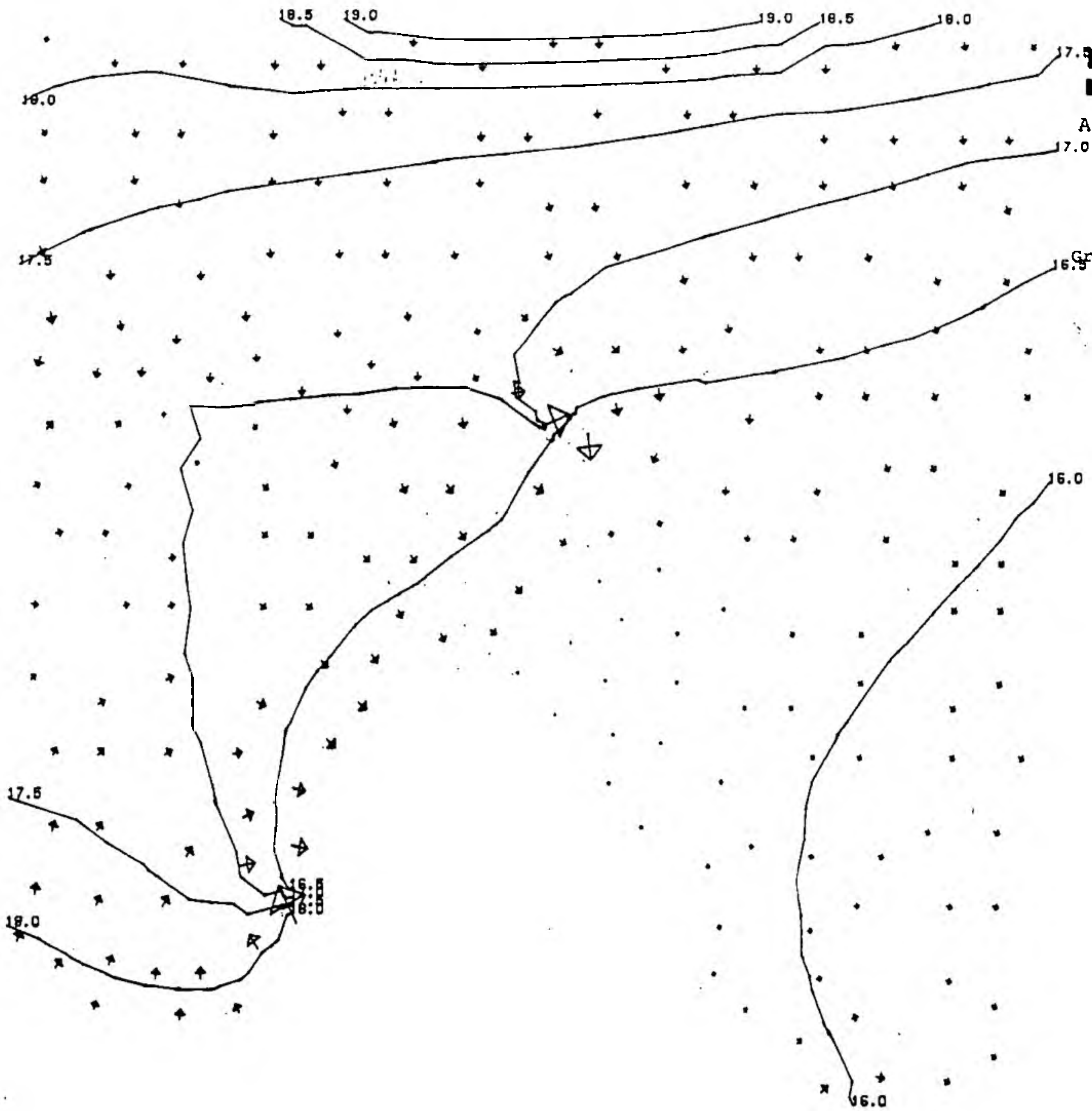
A 4.5.3

Groundwater Levels, No Channel, East Section.

Groundwater contours (m aod)





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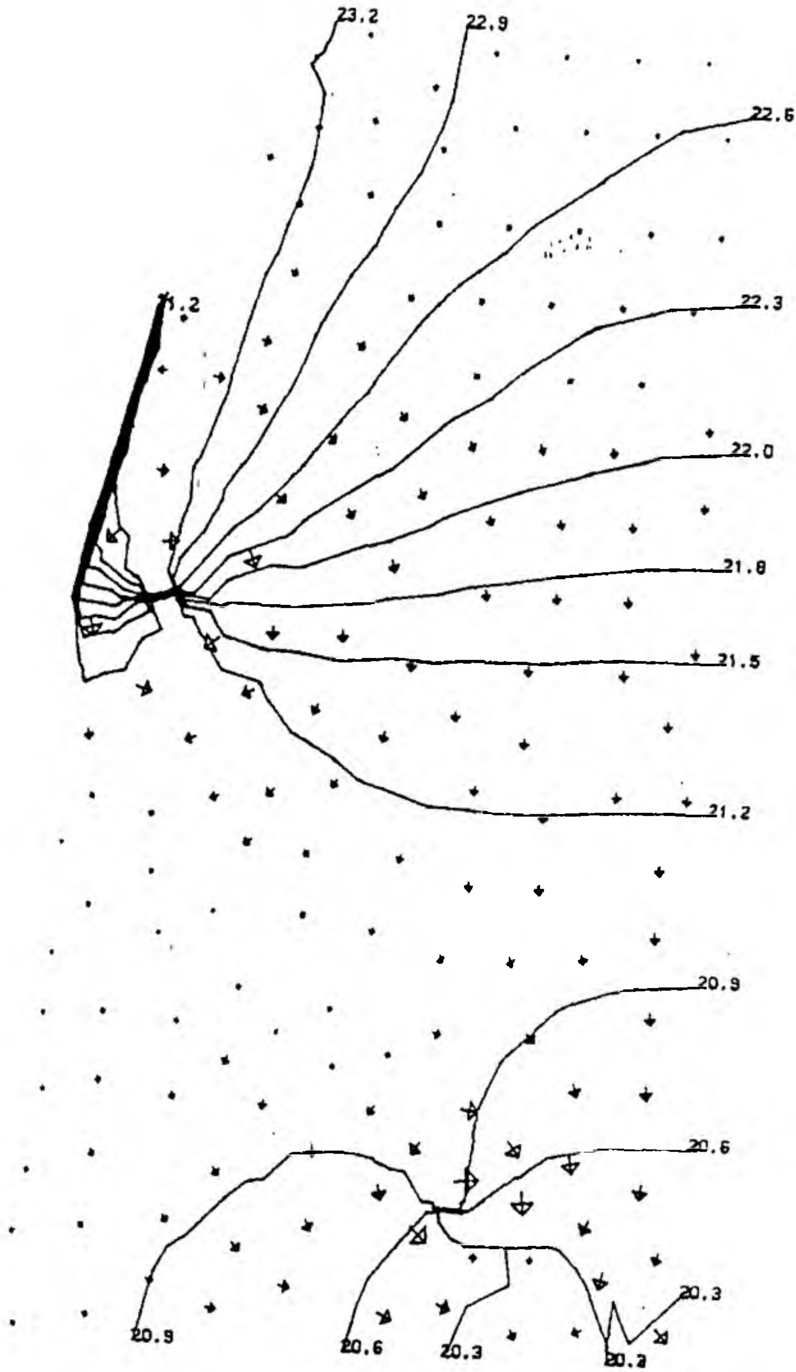
THE ENVIRONMENTAL CONSULTANCY

A 4.5.4

Groundwater Levels, Channel In West Section.

Groundwater contours (m aod)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20.9
20.6
20.3

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A 4.5.5 Groundwater Levels, Channel in Place, East Section.

Groundwater contours (m aod)

A 4.5.6 **Groundwater Levels, Channel in Place, Central Section**

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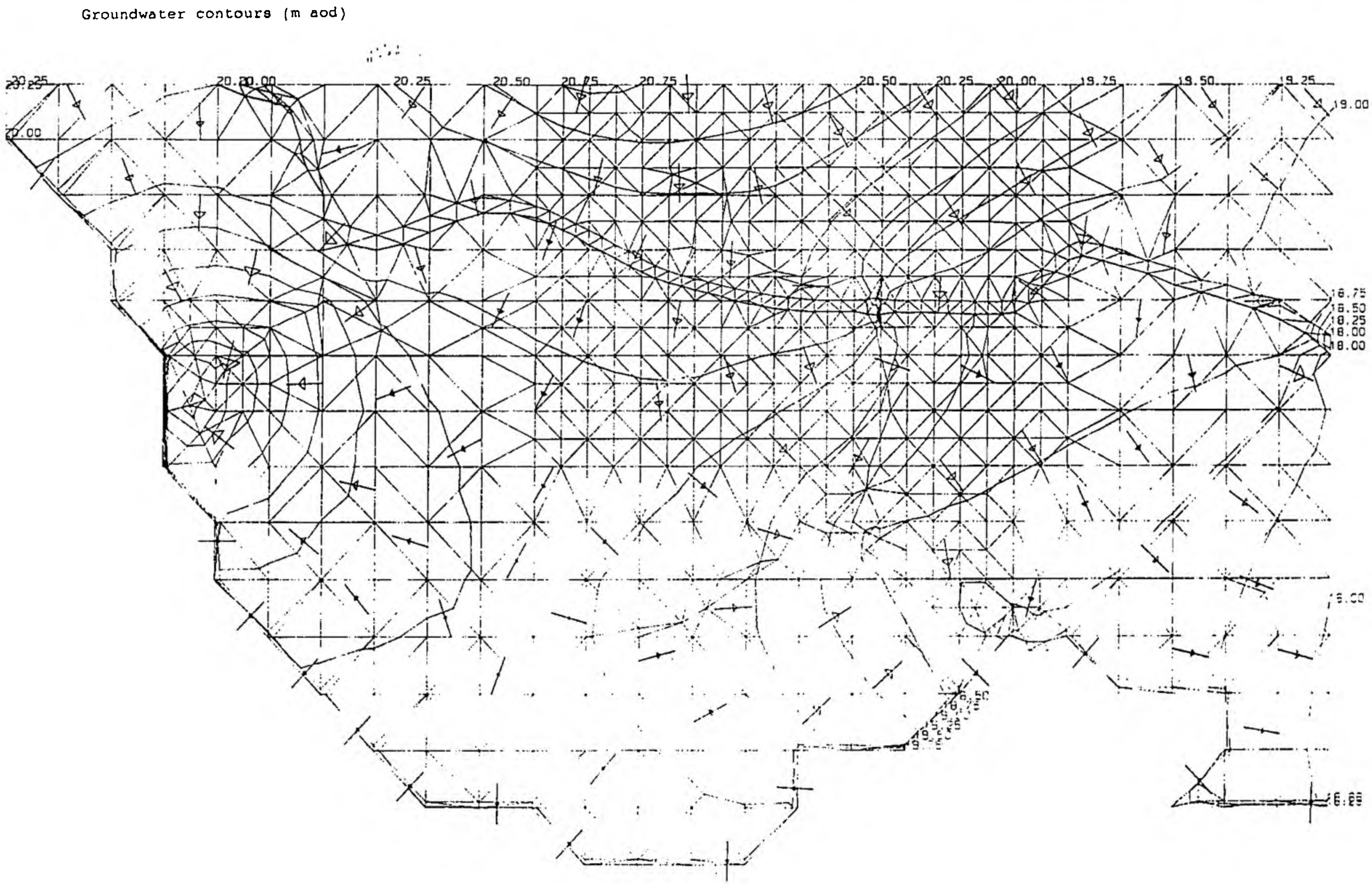

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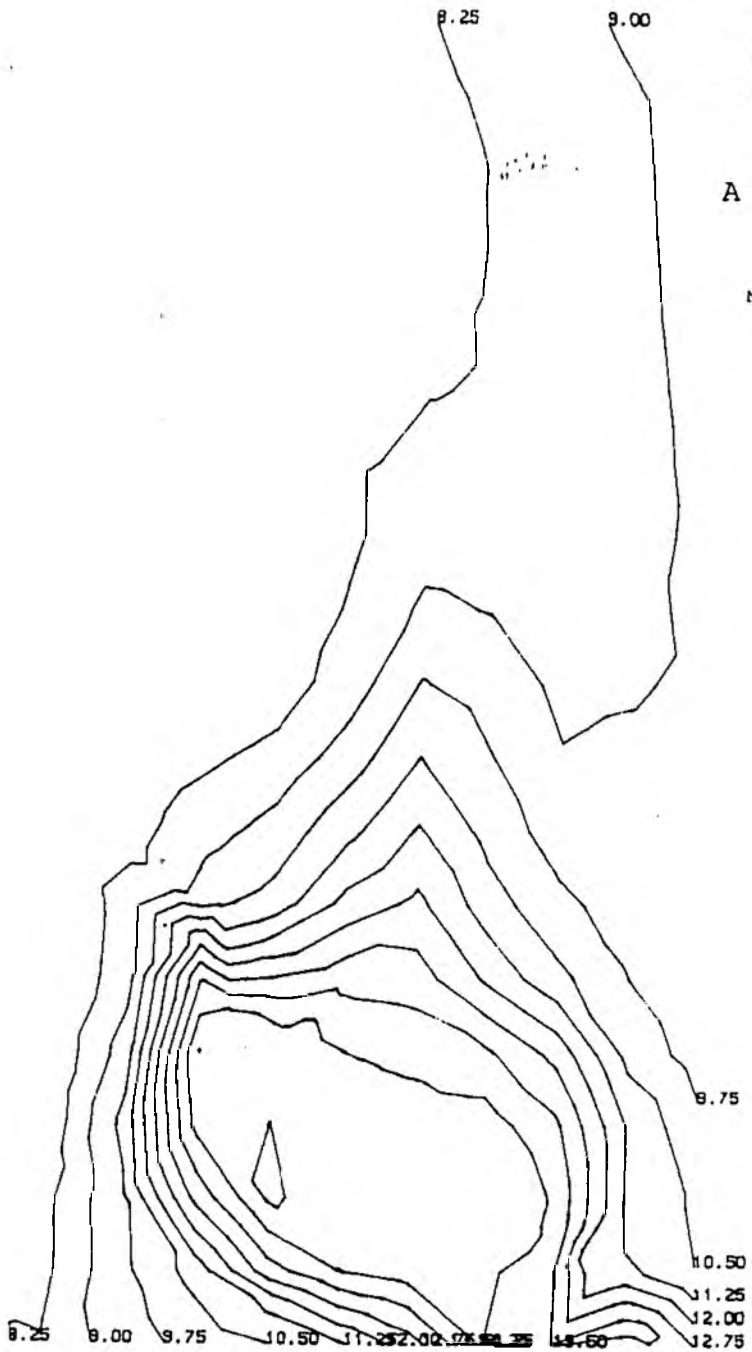
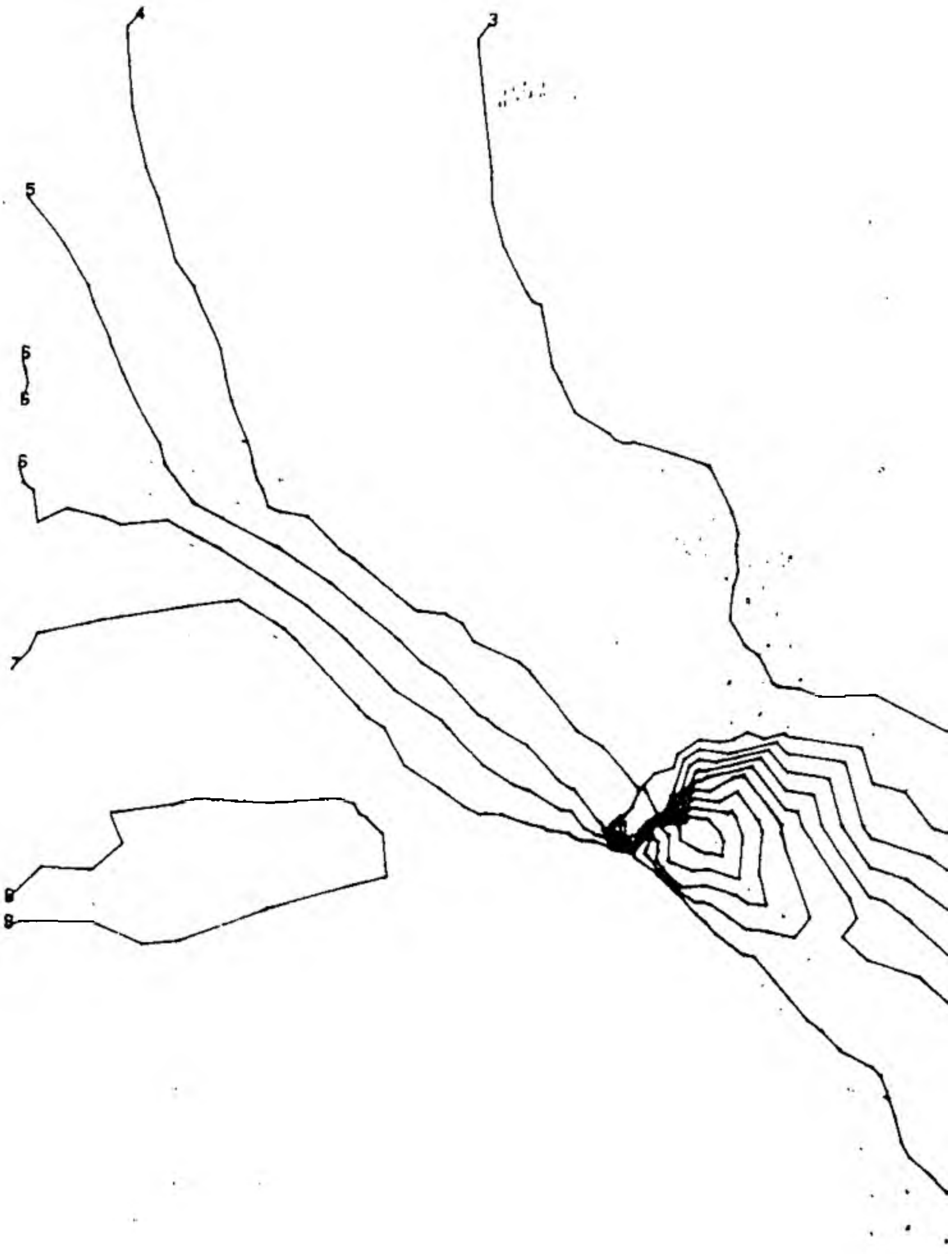
THE ENVIRONMENTAL CONSULTANCY

A 4.5.7 Nitrate Concentrations, Channel in Place, West Section.

Nitrate Contours (mg l^{-1})

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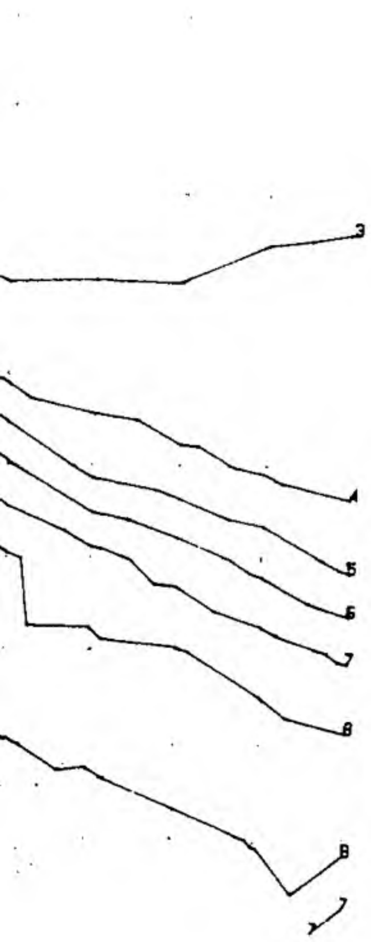
RPS CLOUSTON
THE ENVIRONMENTAL CONSULTANCY

A 4.5.8

Nitrate Concentrations, Channel in Plac
East Section

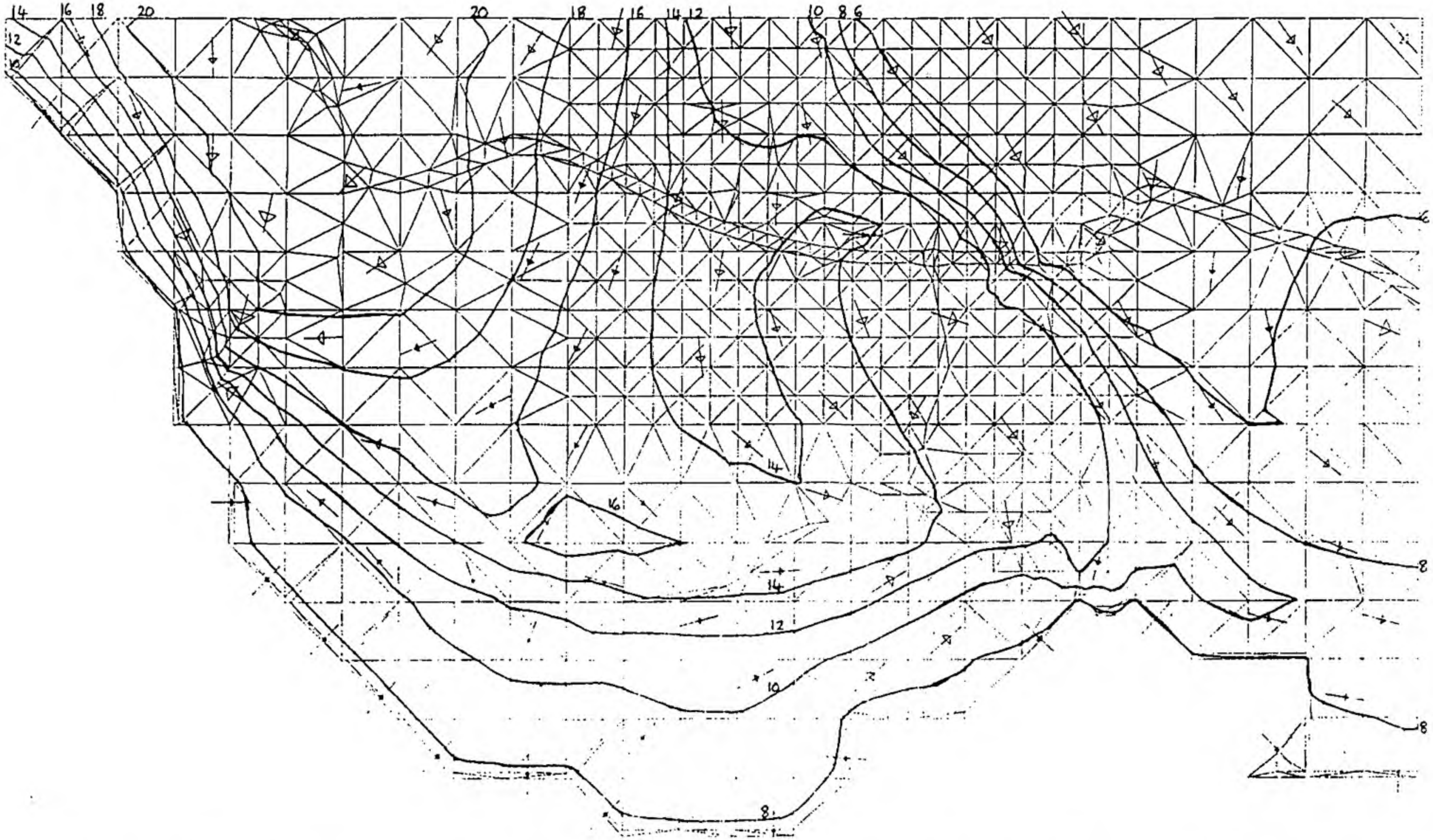
Nitrate Contours (mg l^{-1})



A 4.5.9 **Nitrate Concentrations, Channel in Place, Central Section**

RPS CLOUSTON
THE ENVIRONMENTAL CONSULTANCY

Nitrate Contours (mg l^{-1})



Appendix 5

Algal growth model and factors affecting the growth of algae

Appendix 5 Algal growth model and factors affecting the growth of algae

Dr C S Reynolds, Institute of Freshwater Ecology

A5.1. ALGAL GROWTH MODEL

A5.1.1 Introduction

In this study the approach to the prediction of the characteristics of phytoplankton in the proposed relief channel regulate the assembly of suspended biomass. In this section, the approach has been founded on the assumption (for which there is good evidence, as shown in Warwick, 1984; see also Whitehead & Hornberger 1984) that nutrients scarcely limit concentrations while biotic influences (e.g. grazing, parasitism) act, at best, spasmodically. Thus, the only ultimate limit to supportable biomass is likely to be the underwater light availability and the rate of its attainment will be influenced by temperature, turbidity and the rate of downstream displacement and dilution; these factors are discussed in Section A5.2. Indeed, this upper carrying capacity for phytoplankton can be modelled from first principles with reasonable accuracy and is of value in defining a "worst-case" scenario, that is to say, "the plankton supported will not exceed ....". By ascribing growth rates to phytoplankton in the channel, under different model-flows and travel times with differing turbidity loads, it is possible to determine 'output concentrations' at the downstream end against given 'intake' concentrations at the upstream end.

A5.1.2 Model algorithms

We should define two quantities: the chlorophyll-carrying capacity and the rate of its attainment. It is then possible to judge the extent of growth downstream in the time available for transit.

The simplified capacity model used by Reynolds in Warwick (1984) and in analogous deductions for a customer report on the River Tees is given by (1):

$$N_{max} = \frac{1}{e_s} \left\{ \frac{P_{max}\ 0.7\Gamma\ \mathrm{Ln}(\Gamma_o/0.5I_k)}{24\ R\ h_m} - \epsilon_R \right\} \qquad (1)$$

The equation assumes that light-sustained photosynthesis (P) during the daylight period (T hours) in the small portion of the water column, h_m, to which sufficient light penetrates, is exactly balanced by respirational losses (R) through the entire column over 24 hours. e_s is the extinction coefficient per unit chlorophyll (~0.013 $m^2\ mg^{-1}$ for *Stephanodiscus hantzschi*) while ϵ_R is the residual extinction due to water colour and to non-living matter in suspension. For clay and silt, this is estimated to be around 20 $m^2\ kg^{-1}$ but we prefer to use the value of ϵ_R fitted to discharge data for the Thames (see A6.2.1.3), viz.

$$\epsilon_R = 0.11q - 1.10 \tag{2}$$

where q = discharge in m^3s^{-1}.

The ratio of maximum photosynthetic rate to dark respiration rate P_{max}/R can fall between 7 and 25; Reynolds (1984a) has found 15 to be a typical ratio in many freshwaters and this value is adopted here. I_k the instantaneous light level required to saturate P_{max}, is highly variable and can be substantially lowered in light adapted cells. For river borne diatoms, it is known to fall below 50 μmol photon $m^{-2}s^{-1}$ and, on theoretical grounds, below 25. Here I_k is put at 25 μmol proton $m^{-2}s^{-1}$. The highest incoming radiation intensity will occur at the summer solstice when, under a clear sky, a maximum of 1.33 mmol photon $m^{-2}s^{-1}$ is experienced. The mean, I_0 over (T=) 16h would be 972 μmol proton $m^{-2}s^{-1}$.

With these components and while q is <10 m^3s^{-1} (ϵ_R = 0), N_{max} solves at 2344 mg chl m^{-2}; if the mean depth of the relief channel is 3.2 m, the concentration is equivalent to ~733 mg chl m^{-3}.

At greater discharges, the suspended load shades out algal capacity. At 40 m^3s^{-1}, for example, ϵ_R = 3.30 and N_{max} is only 478 mg m^{-3}. At 100 m^3s^{-1}, N_{max} ~0.

The rate of attainment of N_{max} depends upon $\underline{r}^*$, the maximum sustainable rate of increase in cell mass under the conditions obtaining. Reynolds (1989) has proposed equations to predict $\underline{r}^*$ for given algae against the effects of temperature and light. For *S. hantzschi*, continuously light-saturated growth rate at 20°C is 1.18 d^{-1}; at 15°C over a 16h period of saturating light it is ~0.53 d^{-1}. The correction for turbidity, equivalent to the fraction of the daylight period, T, that the algae spend in light (>0.5 I_k) is h_k/h_m; the greatest average depth of h_k can be solved from the extinction coefficient, ($N\epsilon_s + \epsilon_R$), as:

$$h_k = (\mathrm{Ln}Io - \mathrm{Ln}\ 0.5I_k) \ / \ (N\epsilon_s + \epsilon_R) \tag{3}$$

Using the same values, $h_k \geq h_m$ while ϵ_R = 0 and N <104 mg m^{-3}; as ϵ_R increases above q = 10 m^3s^{-1}, so the maximum rate is sustained through the full channel depth. At higher flows (ϵ_R increases) or higher algal populations (N increases) so $\underline{r}^*$ is diminished in the ratio h_k/h_m.

A5.1.3 Simulation

Simulation of algal growth can now be attempted for different algae at different temperatures and light incomes. As an example, Fig A5.3.1, traces the 'worst case' effects of downstream increase of an inoculum, N_0, into the channel at Maidenhead by the time it is discharged back to the mainflow at Eton. The plot could be regenerated for different algae and different seasons of the year but they would show less extreme effects than this 'worst case'. What is clearly illustrated is that at channel flows of >50 m^3s^{-1}, the combination of high ϵ_r values and the shortness of the travel period <0.33 d do not permit any significant downstream increase to occur: the output concentration is unchanged with respect to the input. Not all these combinations are necessarily realistic, since populations >300 mg Chl m^{-3} at discharge >100 m^3s^{-1} would be as impossible to sustain in the main river as they have been argued to be in the proposed relief channel. Against this, even quite modest concentrations (~10 mg Chl m^3) might increase fourfold flowing at <5 m^3 s^{-1} and thirty fold at <2 m^3s^{-1}. When N_0 is 100 mg Chl m^{-1} (well within experienced values), maximum self-shaded populations of ~700 mg Chl m^{-3} are obtained within the length of the channel.

A5.2 FACTORS AFFECTING THE GROWTH OF ALGAE

A5.2.1 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^3s^{-1} at Reading (Lack 1971), ~50 m^3s^{-1} at Medmenham (Lack *et al.*, 1978) and nearer 70 m^3s^{-1} at Walton-on-Thames (from data presented by Bowles & Quennell, 1971). The generalized implication is that when critical discharge is exceed, then plankton maxima are swept downstream. Yet the simultaneity of bloom and collapse events that have occurred along vast lengths of river, 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.

A5.2.1.1

(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 x 10^9 cells m^{-3} of *S. hantzschii* at a station near Reading, when the discharge was 12.7 m^3s^{-1}. One week later, discharge had increased to 85 m^3s^{-1} and the population had fallen to 3.6 x 10^9 cells m^{-3}. In both instances, the discharge of diatoms (337 x 10^9, 306 x 10^9s^{-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.

A5.2.1.2

(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, 36 h is theoretically required to travel 128 km from Oxford to Walton to maintain a discharge of 263 m^3s^{-1} (mean velocity 0.99 m s^{-1}); at 52.5 m^3s^{-1}, 96 h is required (0.37 m s^{-1}); at 13 m^3s^{-1}, the requirement is 168 h, (0.21 m s^{-1}). The equivalent travel-times in the lower Thames (Windsor- Walton, 32 km) are 9, 24 and 42 h 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, ~375 to ~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 24 h, 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 mean flow is reseeded' continuously (cf. Wawrik, 1962). 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.

5.2.1.3

(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* (z_{eu}, above the 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 (vis., $3.7/z_{eu}$, 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 discharge were generally low. This gave:

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

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

$$\epsilon_R = 0.11\, q - 1.10$$

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^3s^{-1}, 1.1 m^{-1} at 20 m^3s^{-1} and 3.4 m^{-1} at 40 m^3s^{-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.

A5.2.2 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 periods 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 presented. 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 A5.1 apply to a river depth of ~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 (c) above and that chlorophyll concentration is given by (ϵ_{min} - ϵ_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 (~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 A5.1

		Chlorophyll, mg m^{-3}		
Time	ϵ_{min} m^{-1}	at 10 $m^3 s^{-1}$	at 20 $m^3 s^{-1}$	at 40 $m^3 s^{-1}$
Winter Solstice	2.27	174	90	0
Equinox	3.88	298	213	37
Summer solstice	5.50	422	338	161

A5.2.3 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 litre. 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 *Manoraphidium* 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 16^6 *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 cocentration. Nevertheless, Lack's (1971) data showed the 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 ~ 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.

A5.2.4 The effect of water temperature

Temperature of Thames water generally fluctuates within the range of 4-20°C and may be considered unlikely ever to prevent algae from growing. The physiological acitivity 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°, 16h 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.

A5.2.5 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 seeding 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 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 (1972) referred to above (5.2.3) 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.

References

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

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 funcrtion. Oecologia (Berl/) 18, 63-84

Bottrell H.H. (1975b) Generation teme, lengeth 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) Furvey of the River Tees. Part 3: The non-tidal reaches - chemical and biological. Wat. Pollut. Res. Tech. Pap. No 6, 1-189

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

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

Keymer, I.F., Smith, G.R., Roberts, T.A., Heaney, S.I., Hibberd, D.J., 1972. Botulism as a factor in water fowl mortality at St. James' Park, London, The Vet. Record, 90, 111- 114

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. (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

Lloyd, C.S., Thomas, G.J., MacDonald, J.W., Borland, E.D., Standring, E.D., Smart, J.L.,1976. Wild bird mortality caused by botulism in Britain, 1975, Bird Conservation, 10, 119-129.

MacLean, D.D., 1946 Duck disease at Tulare Lake, Calif. Fish Game, 32, 71-80

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

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

Reynolds, C.S. (1989). Physical determinants of phytoplankton succession. In Sommer, U. (ed.) Plankton Ecology, pp 9-56. Springer, Madison

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

Warwick, R.M. (1984). Predicted effects of proposal changes in patterns of water abstraction on the ecosystems of the lower River Thames and its tidal estuary. Unpublished report to Thames Water NERC Institute of Marine Environmental Research, Plymouth

Wawrik, F. (1963) Zur Frage : Fuhrt der Donaustrom autochthones Plankton Arch. Hydrobiol. Suppl. 27, 28-35

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

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

Appendix 6

Ornithology

Appendix 6 Evaluation of the ornithological impact of the proposed Maidenhead Flood Relief Channel, with particular reference to botulism in birds

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A6.1 INTRODUCTION

The purpose of this project was to evaluate the likely consequence for birds of constructing a flood relief channel for the River Thames between Maidenhead and Windsor, Berkshire. Since the field survey was of short duration and limited to August, it was not possible to do a full evaluation of the ornithological interest of the area. Rather, one site visit was made in early August 1990 in order to assess the likely impact of the scheme on the area, based simply on a brief survey of both the birds and of the kinds of habitats occurring there. Particular attention was paid to the possibility that the scheme might have an effect on the incidence of the disease botulism in birds in the area, and suggestions are made as to how the risk might be reduced.

A6.2 BOTULISM

A6.2.1 Symptons of the disease

It is a bacterial disease caused by several strains of Clostridium botulinum. It is a soil organism that occurs widely in anaerobic conditions. Most infections of birds are caused by the ingestion of the toxin botulin produced by the bacteria. The effects on the birds can be considerable, though not necessarily fatal. For some birds, drowsiness and a weakness of the legs and wings, and associated immobility, are all they suffer and recovery can occur within days. The critical condition seems to be a paralysis of the neck, called "limber-neck". This is fast-acting and death highly likely to occur, frequently through drowning. Why some birds are able to recover while others do not is not understood, but a difference in the dosage ingested could be the reason.

A6.2.2 Incidence of the disease

It is especially prevalent in the USA but even there it was not reported on a large scale until 1930. Since 1970, the disease has regularly affected birds in the Netherlands where C. botulinum types C and E were largely responsible. There have also been deaths from botulism recorded since 1973 in the Spanish wetland Cota Donana.

Confirmed outbreaks of botulism in the United Kingdom before 1970 were uncommon and usually restricted to poultry. However, in 1975 a high mortality was caused by the disease in wild birds in at least six sites throughout England, Wales and Scotland. These were confirmed outbreaks, with type C being identified. It was suspected of killing birds in a further ten sites. Botulism was found in that outbreak in roosts, along shore-lines and in rubbish tips.

Since then, confirmed outbreaks have been recorded during the late 1970's and early 1980's in the Thames region from the estuary up to Oxford and elsewhere in the Midlands. Outbreaks have occurred in a variety of habitats including rivers, ponds, gravel pits, lakes, ornamental lakes and brackish estuaries.

The 1975 outbreak concerned primarily gulls. However, botulism infects ducks and other waterfowl, including rare species such as the avocet and purple heron in Holland. The species that have been affected are listed in Table 1.

A6.2.3 Lethality of the toxin

The toxin in the strain A, B and E to which human being are exposed is lethal to man in very small doses. In fact it is the most lethal bacterial toxin known. Only 0.00000000033 mg of the strain that kills man will kill a mouse. The lethal dosage of type C and E for birds has not been measured but may be of a high order. Clearly it is a hazardous substance.

A6.2.4 Natural history and transmission of the disease

The bacterium is a widespread soil-organism which thrives in anaerobic conditions, particularly where there is rotting vegetation and animal matter. Because of this, the organism can prosper in stagnant water and mud: type C, the commoner infective strain found in birds, seems to be particularly associated with mud. In water, the optimum ph range appears to be 5.8-8.3. The bacterium is often associated with the presence of small particles of organic matter. The organism also does particularly well at high temperatures which affect it in several ways. First it directly drives out oxygen in the water and so intensifies its deoxygenisation. Second, warm temperature accelerates the growth of vegetation which, when it dies and rots, further depletes the concentration of oxygen. Finally, warm temperature accelerates the growth of the bacterium itself. Although not precisely measured, the optimum temperatures for the production of toxin seems to be in the region of 20-30 degrees Centigrade. It can also be abundant in rubbish tips. The juxta-position of stagnant water and rubbish tips, along with high summer temperatures, would seem likely to raise the possibility of an outbreak occurring.

It is possible that the bacterium is ingested directly by birds. But according to our literature search, the most likely cause is birds eating rotting vegetation and animal matter that contains the toxins produced by the bacteria. Birds that probe in the mud seem to be particularly at risk. On rubbish tips, carrion-eating birds eat rotting organic matter. In aquatic environments, they can eat rotting invertebrates and vertebrates, such as fish. In fact, infected fish offal has been established as being a cause of outbreaks of botulism.

Micro-scopic invertebrates also may contain the toxin and can therefore be another source of the poison.

The birds can also ingest the toxin by eating other live animals that themselves have been contaminated by the toxin. For example, fly maggots eating carrion ingest meat infected with the toxin but are not affected themselves. But by eating the maggots, the birds take in the toxin. Although no case has to our knowledge been reported, it would be possible in principle for predatory birds to become infected by eating prey, such as other birds and fish, that have already ingested a dose of toxin.

In principle it would also be possible for birds to take in a lethal dose of toxin by drinking which they do particularly in hot weather. The circumstances would probably have to be extreme. For example, a rapidly growing bacterial population in an evaporating, small stagnant pool might cause concentrations of the toxin in the water to increase so much that the ingestion of a small amount would prove to be lethal. Also infected microscopic invertebrates may be ingested as food or. in drinking water.

There is a particular problem with gulls, mainly the herring gull, which frequently scavenges on rubbish tips during the day and roosts on, or by, water at night. These birds may transfer infected material from the tip to the water by carrying it in their beaks and in their crops from whence it might be ejected as undigested pellets. Furthermore, birds infected with a lethal dose are likely to die when on the water by drowning. The water could then become an additional focus for the transmission of the disease as the dead birds will themselves become carrion for other scavengers. Because mud is a source of botulism, mud-feeding birds, such as snipe and lapwing, or birds that forage on the bottom of infected waters, such as mallards, other ducks, geese, swans, rails, coots and moorhens, are also at risk.

A6.2.5 Botulism and the flood channel

Experience in the United States may provide a precedent of how best to minimise the risk of an outbreak of botulism occurring in the flood channel. In 1941, 250 000 ducks out of a population of 2 million died of botulism at Lake Tulare, California. By 1944, a process of sytematic flooding through artificial dykes and drains appeared to be responsible for the virtual elimination of the disease from the duck population of up to 3 million birds (McLean 1946). It seems that by controlling the water level in the lake, anaerobic conditions and high temperatures in the water were reduced to a level which apparently eliminated the disease.

From this, it seems reasonable to expect that the proper choice of water management in the flood channel can substantially reduce the possibility of a botulism outbreak. The need is to avoid the combination of high temperatures and anaerobic conditions, especially in alkaline environments, in which the bacterium flourishes. It is not within our field of expertise to precisely define what that water management programme should be in this case, but we are able to make some suggestions.

We suggest that a combination of water monitoring and a capacity to move water, if necessary, is required. Concerning water movement, it is a matter of choosing an optimum. Too rapid and frequent movement would reduce vegetation growth and diminish the attractiveness of the channel to both man and animals. Too sluggish a water movement would encourage the high-temperature anaerobic conditions favoured by the bacterium. It is not within our expertise to put values to these suggestions; in fact, the knowledge of the conditions that favour the bacterium is so imprecise that the precise information may not exist. An empirical approach, based on monitoring oxygen content, temperature and levels of C. botulinum, particularly at times of high summer temperatures, is all we feel able to recommend. We would also like to suggest a contingency plan in case unacceptable levels build up and bird deaths occur. This would require having the capacity to remove corpses and rotting vegetation. In the latter case, this might be done by flushing the channel or by mechanical means.

Early symptons should be looked for. Gallinaceous birds seem to be particularly sensitive to the toxin and may provide early indications of poisoning.

The fact that botulism outbreaks have occurred on the Thames itself in places not far from the proposed flood channel suggests that this could be an important environmental issue. Because it occurs in the river itself where the water-flow is expected to be generally faster, the risk has to be taken particularly seriously. The channel is also expected to attract high-risk species, such as mallards, swans, moorhens, coots and herring gulls, which are probably common in the area. In the case of herring gulls, there is an enhanced risk due to the presence of rubbish tips in the area.

Some of the conditions that favour botulism also favour blooms of blue-green algae, notably high temperatures. Such blooms may also kill birds in fresh-water sites.

The present state of knowledge does not allow us to estimate the size of the risk of birds dying from either botulism or ingesting blue-green algae in eutrophic conditions. This is why we suggest that a monitoring programme be implemented.

A6.3 ORNITHOLOGICAL AND HABITAT APPRAISAL

A6.3.1 The site visit

This was made from August 6-8 1990. The weather was hot, sunny and dry. There had been little rain for sometime so the terrain was very dry. The survey was conducted between 0730 and 1930 BST. The whole length of the proposed flood relief route was walked by both authors. The main habitat types encountered were marked on an OS map and any birds seen or heard were noted. The habitat types were those that on our general experience suggested would characterise the nature of the bird communities living there.

A6.3.2 Survey results

The proposed route of the flood relief channel was divided into 20 sections, labelled A-T in Fig. A6.1. The habitats and birds encountered in each section were as follows. If no birds are mentioned, none of interest were seen.

Section A

This was the east side of the Thames at Taplow where the inlet to the flood channel would be sited. It was viewed from the top of the hill, overlooking the line of the flood channel itself. The area that could be seen from here was largely mature woodland. There was a mixture of mature trees and saplings with few intermediate aged trees. The saplings have not been recently thinned-out. The tree species included sweet chestnut, sycamore, ash, copper beach, scots pine and oak with an understory in places of rhododendron. The birds encountered were typical woodland or ubiquitous species and included green woodpecker, rook, wren, robin, goldcrest, mistle thrush, magpie, chiffchaff and jay. Roe buck and hare were also present.

Section B

The channel follows an existing ditch crossing Mill Road near the waste-paper site. It is marked by a line of willow trees through a typical river valley grassland landscape.

Section C

This was a section of the west bank of the Thames running alongside, or opposite to, sections A and B of the flood relief scheme itself that had not been directly visible from the top of the hill. From the vantage point of Boulter's Lock, we could confirm that the parts adjacent to the river were an extension of the mature woodland seen from the hill top. It was also interspersed with business premises and recidences. The trees by the river itself were predominantly willow and poplar. The birds seen alongside the river were grey wagtail, kingfisher, great-crested grebe, coot, mandarin duck and black-headed gulls.

Section D

This was below Taplow and adjacent to a lake used for boating and fishing. The lake attracted mallard, black-headed gulls, great-crested grebe, an adult common tern feeding two juveniles, coot and mute swans. On the day it was visited, there was a notice warning people that there was a bloom of toxin-producing blue-green algae in the lake and that therefore swimming was dangerous.

Section E

This was mainly set-aside agricultural land including fields formerly used for growing leeks and cereals. This is the area where it is proposed that an amphibian and reptile reserve be created. The only birds observed were house martin and goldfinch.

Section F and G

Both were set-aside agricultural land, formerly used for oats but now heavly weeded.

Section H

This consisted of cereal fields with mature hedge boundaries.

Section I

This included a small wood by the M4 amongst cereal fields and mature hedgerows. The trees in the wood included ash, hawthorn, elder and alder. No mature trees were seen. This could be an important wildlife refuge. It connects up with a mature hedge system which could provide a system of linear habitats joining the wood with otherwise isolated groups of trees. Birds seen included jay, sand martin amd bullfinch.

Section J

This was an area of cereal fields.

Section K

This was a market-gardening area.

Section L

Both market gardening and cereal growing occurred here. It is the proposed site for a waterfowl refuge area. However the site is very close to overhead power cables which actually pass over the corner of the proposed site. The cables may cause some fatalities amongst waterfowl leaving or returning to the area, especially amongst the heavier and less manoeuvrable species, such as mute swans.

Section M

This was a market gardening area. Section N

Also a market gardening area, it included set-side cereal fields, a rubbish tip and much semi-derelict grazing land. It is the site of another proposed waterfowl refuge and is also near to overhead power cables on the northern side.

Section O

This was river-valley grazing land surrounded by some fairly mature trees, particularly willows. The channel here would run south of a sewage farm where heavy metals are thought to enter the ground-water flow (H Dawson (IFE); presonal communication).

Section P

This was river-valley grazing land.

Section Q

This was rough-grazing land and paddocks.

Section R

This was an area of cereal growing and grazing, sandwiched between two major roads. It is the proposed site of another waterfowl refuge.

Section S

This was a grazing area and was immediately adjacent to a small ornamental wood. The wood might be good for butterflies, the species seen including the speckled wood (numerous), the gatekeeper and the holly blue. Birds encountered included jay, long-tailed, blue and great tits. The wood may have originated as a small ornamental tree plantation which included the following indigenous trees; willow, hawthorn and elder. The wood was on the northern edge of Eton School's cricket field.

Section T

This area included a golf-course and riverside woodland containing willow, ash, elder and sycamore. There were a number of mature trees at this location. It was the final section, the point at which the channel would enter the Thames.

A6.4 AN OVERVIEW OF THE SURVEY

The bird species were typical of the habitats encountered. In our opinion, none of the species would be significantly affected by the channel. In fact, a number could benefit from an additional stretch of water, particularly waterfowl as refuges for them are proposed. In our opinion, the refuge in section L is dangerously close to overhead power lines and we recommend that this area be put to other uses. In our opinion,
the potential hazard to birds arsing from the cables near the propsed refuge in section N is much less because they would be situated some hundreds of yards away. We therefore think that this refuge should be included along with the proposed refuge in section R.

The planting of trees along the channel route should be undertaken with care, especially if the channel is to be maintained in a stagnant state for lengthy periods. This is because the leaf-fall into the water increases the probablity of anearobic conditions developing in the mud. As was discussed above, this provides one of the conditions that encourage the growth of the toxic bacterium Clostridium botulinum and the death of birds from botulism. A number of species known to be at risk to this disease were recorded during the survey, including coot, black-headed gull, mallard, mandarin duck, great-crested grebe and swan. The presence of refuse tips servicing the neighbouring towns of Maidenead, Slough and Windsor could also facilitate an outbreak of this disease.

The blue-green algal bloom on the boating lake near Taplow (section C) draws our attention to another potential hazard for the birds. Death through ingesting their toxins could occur were the channel to be stagnant during prolonged periods of hot weather.

The proposal to establish a steep bank to encourage kingfishers and sandmartins to breed at a point along the channel is welcome. Both species were seen during the site visit.

On the basis of our site visit, we suggest that whereever possible, damage to mature trees and hedgerows should be avoided. They probably form important linear habitats across the valley along which wildlife can move and colonise new places. It is particularly recommended that the small wood in section I should be left intact as it may be important in this respect. We would also like to point out that the channel could contribute usefully to this dispersive process. By providing a ribbon of trees, bushes and other vegetation, and by being itself sheltered, it should provide a useful route along which many species might disperse.

References

Keymer, I.F., Smith, G.R., Roberts, T.A., Heaney, S.I., Hibberd, D.J., 1972. Botulism as a factor in water fowl mortality at St. James' Park, London, The Vet. Record, 90, 111- 114

Lloyd, C.S., Thomas, G.J., MacDonald, J.W., Borland, E.D., Standring, E.D., Smart, J.L.,1976. Wild bird mortality caused by botulism in Britain, 1975, Bird Conservation, 10, 119-129.

MacLean, D.D., 1946 Duck disease at Tulare Lake, Calif. Fish Game, 32, 71-80

Species References

Mallard*
Tufted duck
Pochard
Pintail
Common teal
Shelduck Pink-footed goose
Canada goose
Mute swan*
Great-crested grebe* Water rail
Coot*
Cormorant Herring gull*
Lesser black-backed Black-headed gull*
Little gull
Lapwing gull
Snipe
Ruff Woodpigeon*
Blackbird*
Song thrush*
Starling*

Sections used for ornithological field survey

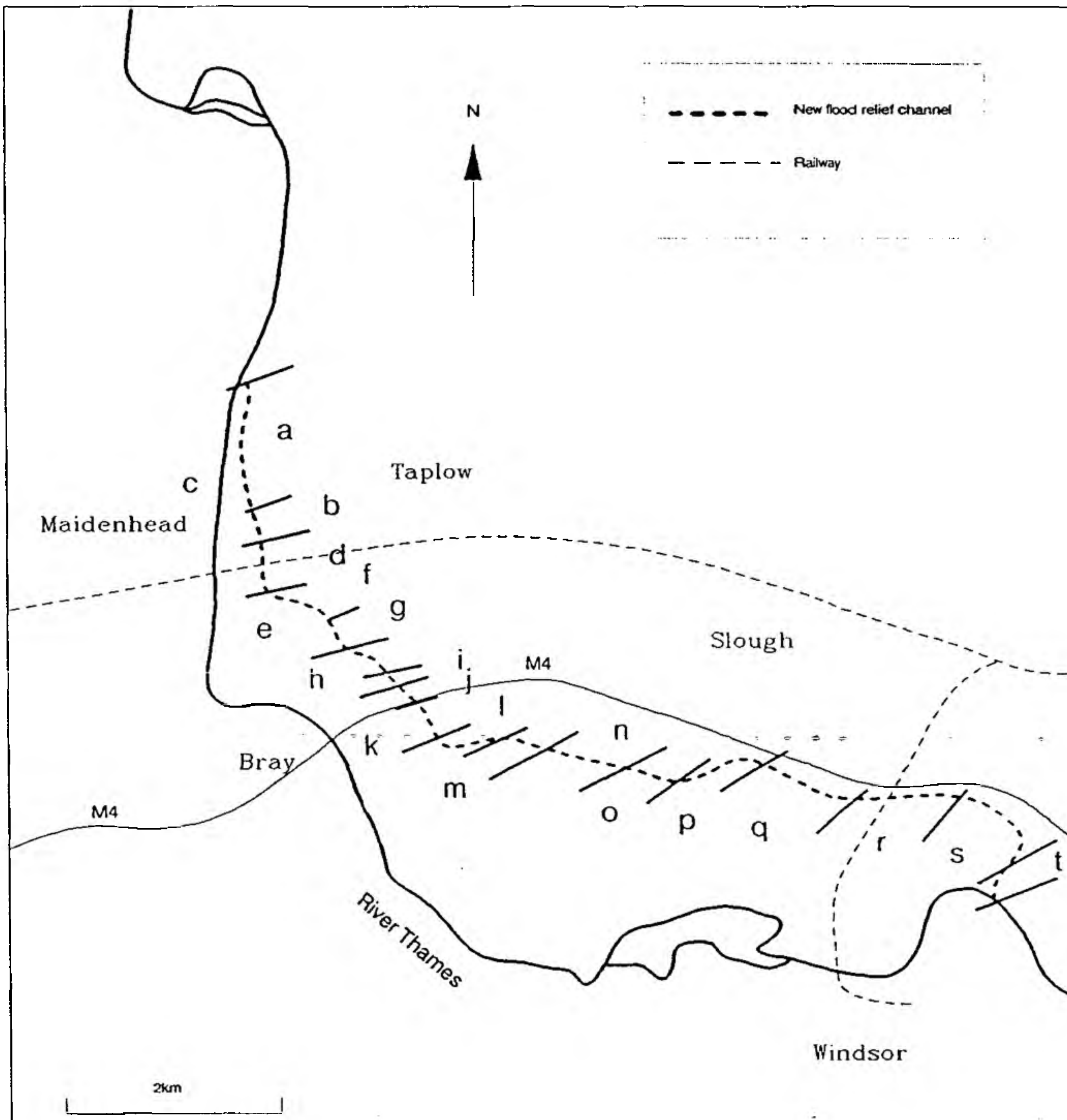

Figure A6.1

Appendix 7

Detailed results of model runs

Appendix A7 Detailed Results of Model Runs

Table No.	Determinand	Year
A7.1	Flow	1974
A7.2	Nitrate	1974
A7.3	Dissolved xygen	1974
A7.4	BOD	1974
A7.5	Ammonium	1974
A7.6	Orthophosphate	1974
A7.7	Unionized Ammonia	1974
A7.8	Flow	1975
A7.9	Nitrate	1975
A7.10	Dissolved xygen	1975
A7.11	BOD	1975
A7.12	Ammonium	1975
A7.13	Orthophosphate	1975
A7.14	Unionized Ammonia	1975
A7.15	Flow	1976
A7.16	Nitrate	1976
A7.17	Dissolved xygen	1976
A7.18	BOD	1976
A7.19	Ammonium	1976
A7.20	Orthophosphate	1976
A7.21	Unionized Ammonia	1976
A7.22	Flow	1983/4
A7.23	Nitrate	1983/4
A7.24	Dissolved xygen	1983/4
A7.25	BOD	1983/4
A7.26	Ammonium	1983/4
A7.27	Orthophosphate	1983/4
A7.28	Unionized Ammonia	1983/4
A7.29	Flow	1989
A7.30	Nitrate	1989
A7.31	Dissolved xygen	1989
A7.32	BOD	1989
A7.33	Ammonium	1989
A7.34	Orthophosphate	1989
A7.35	Unionized Ammonia	1989

Units used in the following tables
Flow m^3s^{-1} all other determinands mg ℓ^{-1}

Table A7.1

Determinand : Flow
YEAR : 1974

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	60.4	30.2	54.1	58.9	51.4
	95%	166.9	86.6	161.9	166.9	156.9
	5%	12.5	6.1	7.6	12.5	9.4
Romney	Mean					
	95%					
	5%					
Datchet	Mean	61.3	61.0	60.9	61.3	60.9
	95%	167.7	167.4	167.4	167.7	167.4
	5%	13.4	13.2	13.1	13.3	13.3
Marsh Lane	Mean		30.1	6.18	1.7	8.9
	95%		83.1	4.94	0.7	9.9
	5%		6.3	4.94	0.3	3.1
Black Potts	Mean		30.3	6.34	2.0	9.1
	95%		83.3	5.14	1.8	10.1
	5%		6.5	5.11	0.5	3.4

Table A7.2

Determinand : Nitrate
YEAR : 1974

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Bovency	Mean	6.5	6.5	6.5	6.5	6.5
	95%	13.1	13.1	13.1	13.1	13.1
	5%	4.9	4.9	4.9	4.9	4.9
Romney	Mean	6.7	6.8	6.7	6.7	6.7
	95%	13.2	13.2	13.2	13.2	13.2
	5%	5.2	5.3	5.2	5.2	5.2
Datchet	Mean	6.6	6.7	6.7	6.6	6.7
	95%	13.1	13.1	13.0	13.1	13.0
	5%	5.1	5.3	5.3	5.1	5.3
Marsh Lane	Mean		6.8	7.2	15.1	7.1
	95%		13.2	13.4	17.2	13.3
	5%		5.2	5.7	13.6	5.5
Black Potts	Mean		6.7	6.9	6.1	6.9
	95%		12.8	11.8	9.6	12.4
	5%		5.2	5.5	4.6	5.5

Table A7.3

Determinand : Dissolved Oxygen
YEAR : 1974

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	9.8	9.7	9.7	9.8	9.8
	95%	12.3	13.2	12.5	12.3	12.6
	5%	7.4	6.8	7.1	7.4	7.2
Romney	Mean	9.8	9.8	9.8	9.8	9.9
	95%	12.5	13.7	12.8	12.5	12.9
	5%	7.5	6.7	7.2	7.5	7.3
Datchet	Mean	10.5	10.7	10.5	9.9	10.5
	95%	13.4	13.1	13.4	12.6	13.6
	5%	8.2	8.6	8.2	7.5	8.1
Marsh Lane	Mean		11.0	11.9	7.5	11.8
	95%		12.0	14.9	10.2	14.2
	5%		9.8	10.2	6.1	9.8
Black Potts	Mean		11.6	13.2	9.1	12.7
	95%		12.3	16.7	10.7	14.9
	5%		10.4	11.8	7.6	10.8

Table A7.4

Determinand : BOD
YEAR : 1974

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	3.8	3.7	3.7	3.8	3.7
	95%	9.8	9.3	9.7	9.8	9.6
	5%	2.0	2.0	2.0	2.0	2.0
Romney	Mean	3.7	3.6	3.7	3.7	3.7
	95%	9.5	8.7	9.5	9.5	9.3
	5%	2.0	2.0	2.0	2.0	2.0
Datchet	Mean	3.7	3.6	4.0	3.6	4.0
	95%	9.6	9.4	10.4	9.3	10.4
	5%	2.1	2.0	2.3	1.9	2.2
Marsh Lane	Mean		3.8	5.5	0.9	5.0
	95%		10.3	15.8	1.3	14.5
	5%		2.1	2.2	0.6	2.0
Black Potts	Mean		3.8	3.9	0.8	3.9
	95%		10.4	10.8	1.3	10.7
	5%		3.8	2.2	0.6	2.2

Table A7.5

Determinand : Ammonium
YEAR : 1974

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Bovency	Mean	0.16	0.18	0.17	0.16	0.17
	95%	0.43	0.43	0.42	0.43	0.43
	5%	0.06	0.08	0.07	0.06	0.07
Romney	Mean	0.17	0.19	0.17	0.17	0.18
	95%	0.42	0.42	0.42	0.42	0.42
	5%	0.07	0.08	0.08	0.07	0.08
Datchet	Mean	0.16	0.15	0.15	0.16	0.15
	95%	0.41	0.39	0.40	0.41	0.40
	5%	0.06	0.05	0.05	0.06	0.06
Marsh Lane	Mean		0.13	0.11	0.04	0.12
	95%		0.41	0.33	0.07	0.37
	5%		0.03	0.03	0.02	0.02
Black Potts	Mean		0.11	0.07	0.03	0.09
	95%		0.36	0.18	0.05	0.27
	5%		0.02	0.02	0.02	0.02

Table A7.6

Determinand : Orthophosphate
YEAR : 1974

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Boveney	Mean	1.08	0.83	1.01	1.08	1.00
	95%	2.13	1.74	2.10	2.13	2.06
	5%	0.42	0.27	0.31	0.42	0.35
Romney	Mean	1.00	0.77	0.94	1.00	0.92
	95%	1.98	1.52	1.94	1.98	1.89
	5%	0.42	0.33	0.35	0.41	0.37
Datchet	Mean	0.91	0.95	0.89	0.90	0.88
	95%	1.86	1.88	1.81	1.86	1.81
	5%	0.32	0.39	0.33	0.31	0.31
Marsh Lane	Mean		1.50	1.36	0.11	1.41
	95%		0.81	2.33	0.21	2.49
	5%		2.61	0.75	0.06	0.74
Black Potts	Mean		1.33	0.93	0.09	1.08
	95%		2.45	1.53	0.13	2.02
	5%		0.65	0.50	0.05	0.47

Table A7.7

Determinand : Unionized Ammonia
YEAR : 1974

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	0.0061	0.0068	0.0063	0.0061	0.0064
	95%	0.0125	0.0147	0.0133	0.0125	0.0133
	5%	0.0028	0.0029	0.0054	0.0028	0.0028
Romney	Mean	0.0050	0.0047	0.0049	0.0050	0.0049
	95%	0.0098	0.0087	0.0095	0.0098	0.0095
	5%	0.0025	0.0025	0.0025	0.0025	0.0026
Datchet	Mean	0.0047	0.0042	0.0044	0.0047	0.0044
	95%	0.0094	0.0089	0.0091	0.0094	0.0090
	5%	0.0024	0.0019	0.0021	0.0024	0.0022
Marsh Lane	Mean		0.0043	0.0037	0.0019	0.0040
	95%		0.0109	0.0091	0.0022	0.0101
	5%		0.0018	0.0017	0.0012	0.0016
Black Potts	Mean		0.0037	0.0024	0.0017	0.0029
	95%		0.0099	0.0055	0.0018	0.0075
	5%		0.0013	0.0012	0.0013	0.0012

Table A7.8

Determinand : Flow
YEAR : 1975

		Without Channel	50%	FS5	FNS	FS10
					SCENARIO	
Boveney	Mean	54.1	27.1	48.4	53.1	46.9
	95%	182.7	91.5	177.7	182.7	172.7
	5%	12.4	6.1	7.4	12.4	9.3
Romney	Mean					
	95%					
	5%					
Datchet	Mean	55.3	55.0	54.9	55.3	54.9
	95%	184.0	183.7	184.4	187.0	183.7
	5%	13.5	13.2	13.2	13.5	13.2
Marsh Lane	Mean		27.0	5.6	1.2	7.1
	95%		91.1	5.0	1.1	9.9
	5%		6.3	4.9	0.3	3.1
Black Potts	Mean		27.5	6.1	1.7	7.6
	95%		91.5	5.6	2.6	10.4
	5%		6.7	5.4	0.7	3.6

Table A7.9

Determinand : Nitrate
YEAR : 1975

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	5.8	5.8	5.8	5.8	5.8
	95%	7.1	7.1	7.1	7.1	7.1
	5%	4.3	4.3	4.3	4.3	4.3
Romney	Mean	6.0	6.1	6.0	6.0	6.0
	95%	7.2	7.4	7.3	7.2	7.3
	5%	4.5	4.7	4.6	4.5	4.6
Datchet	Mean	5.9	6.1	6.0	6.0	6.0
	95%	7.2	7.3	7.3	7.2	7.3
	5%	4.5	4.6	4.6	4.5	4.6
Marsh Lane	Mean		6.1	6.6	18.1	6.5
	95%		7.5	7.9	21.0	7.9
	5%		4.7	5.1	12.5	5.1
Black Potts	Mean		6.1	6.4	7.2	6.3
	95%		9.0	7.6	9.2	7.6
	5%		4.5	5.0	6.2	5.0

Table A7.10

Determinand : Dissolved Oxygen
YEAR : 1975

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Boveney	Mean	10.5	10.6	10.5	10.5	10.5
	95%	16.1	18.5	17.4	16.1	16.9
	5%	3.9	3.2	3.3	3.8	3.5
Romney	Mean	10.5	10.7	10.6	10.5	10.6
	95%	16.5	19.2	17.2	16.6	17.5
	5%	3.6	3.2	3.3	3.6	3.4
Datchet	Mean	10.7	11.4	11.4	10.4	11.3
	95%	16.9	17.7	17.7	16.5	17.4
	5%	4.1	5.6	6.2	3.7	5.6
Marsh Lane	Mean		11.1	12.0	2.2	12.3
	95%		14.3	15.8	7.5	16.8
	5%		6.3	7.3	0.0	7.4
Black Potts	Mean		12.1	13.9	4.5	13.8
	95%		16.4	17.3	9.6	17.3
	5%		8.5	11.4	1.0	11.1

Table A7.11

Determinand : BOD
YEAR : 1975

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Boveney	Mean	3.5	3.4	3.5	3.5	3.5
	95%	6.9	6.6	6.8	6.9	6.8
	5%	1.4	1.4	1.4	1.4	1.4
Romney	Mean	3.5	3.4	3.4	3.5	3.4
	95%	6.7	6.5	6.6	6.7	6.6
	5%	1.5	1.5	1.5	1.5	1.5
Datchet	Mean	3.5	3.3	3.6	3.3	3.6
	95%	6.7	6.4	6.7	6.5	6.7
	5%	1.5	1.5	1.7	1.4	1.7
Marsh Lane	Mean		3.4	3.3	1.2	3.4
	95%		6.8	6.6	1.7	6.7
	5%		1.3	1.4	0.8	1.5
Black Potts	Mean		3.3	3.8	0.9	4.3
	95%		6.5	7.3	1.1	7.5
	5%		1.5	1.5	0.7	1.5

Table A7.12

Determinand : Ammonium
YEAR : 1975

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	0.13	0.16	0.14	0.13	0.14
	95%	0.29	0.37	0.33	0.29	0.32
	5%	0.05	0.07	0.06	0.05	0.06
Romney	Mean	0.14	0.17	0.15	0.14	0.15
	95%	0.34	0.43	0.39	0.34	0.38
	5%	0.05	0.06	0.05	0.05	0.06
Datchet	Mean	0.13	0.11	0.12	0.13	0.12
	95%	0.31	0.24	0.27	0.30	0.27
	5%	0.04	0.03	0.04	0.04	0.04
Marsh Lane	Mean		0.07	0.06	0.06	0.06
	95%		0.17	0.14	0.10	0.15
	5%		0.01	0.01	0.02	0.01
Black Potts	Mean		0.06	0.03	0.02	0.04
	95%		0.16	0.06	0.03	0.10
	5%		0.01	0.01	0.02	0.01

Table A7.13

Determinand : Orthophosphate
YEAR : 1975

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	0.66	0.50	0.59	0.66	0.60
	95%	0.98	0.86	0.98	0.98	0.97
	5%	0.43	0.26	0.31	0.43	0.36
Romney	Mean	0.63	0.50	0.57	0.62	0.58
	95%	0.95	0.81	0.93	0.95	0.91
	5%	0.42	0.33	0.36	0.42	0.38
Datchet	Mean	0.54	0.35	0.43	0.54	0.43
	95%	0.89	0.69	0.82	0.89	0.78
	5%	0.32	0.15	0.18	0.32	0.22
Marsh Lane	Mean		0.69	0.4	0.03	0.44
	95%		0.99	0.55	0.03	0.68
	5%		0.46	0.29	0.01	0.26
Black Potts	Mean		0.28	0.04	0.02	0.07
	95%		0.63	0.05	0.01	0.15
	5%		0.06	0.03	0.01	0.01

Table A7.14

Determinand : Unionized Ammonia
YEAR : 1975

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	0.0046	0.0059	0.0050	0.0046	0.0050
	95%	0.0121	0.0161	0.0138	0.0120	0.0138
	5%	0.0021	0.0024	0.0024	0.0021	0.0023
Romney	Mean	0.0035	0.0036	0.0035	0.0035	0.0036
	95%	0.0060	0.0055	0.0057	0.0060	0.0057
	5%	0.0017	0.0015	0.0016	0.0017	0.0017
Datchet	Mean	0.0033	0.0027	0.0030	0.0033	0.0030
	95%	0.0052	0.0041	0.0048	0.0053	0.0047
	5%	0.0014	0.0009	0.0011	0.0014	0.0012
Marsh Lane	Mean		0.0020	0.0017	0.0052	0.0018
	95%		0.0036	0.0030	0.0109	0.0031
	5%		0.0003	0.0003	0.0009	0.0003
Black Potts	Mean		0.0016	0.0011	0.0015	0.0012
	95%		0.0030	0.0021	0.0019	0.0021
	5%		0.0004	0.0004	0.0010	0.0004

Table A7.15

Determinand : Flow
YEAR : 1976

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Boveney	Mean	22.5	11.2	18.0	22.2	18.1
	95%	87.6	43.7	82.6	87.6	77.6
	5%	3.9	1.8	1.8	3.9	2.8
Romney	Mean					
	95%					
	5%					
Datchet	Mean	23.3	23.0	23.0	23.3	23.0
	95%	88.5	88.1	88.1	89.6	88.1
	5%	4.7	4.6	4.6	4.7	4.5
Marsh Lane	Mean		11.2	4.4	0.5	4.1
	95%		43.9	4.9	0.5	9.9
	5%		2.1	2.1	0.3	1.0
Black Potts	Mean		11.3	4.5	0.7	4.3
	95%		44.1	5.1	1.0	10.1
	5%		2.3	2.1	0.5	1.2

Table A7.16

Determinand : Nitrate
YEAR : 1976

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Boveney	Mean	7.3	7.3	7.3	7.3	7.3
	95%	14.3	14.1	14.3	14.3	14.2
	5%	4.4	4.3	4.3	4.4	4.4
Romney	Mean	7.7	8.1	8.0	7.7	7.8
	95%	14.3	14.1	14.3	14.3	14.3
	5%	5.0	5.4	5.3	5.0	5.1
Datchet	Mean	7.7	7.9	7.9	7.6	7.9
	95%	14.2	14.2	14.2	14.2	14.2
	5%	5.0	5.8	5.8	5.1	5.7
Marsh Lane	Mean		8.0	8.2	18.6	8.9
	95%		14.5	14.5	20.9	14.5
	5%		6.0	6.0	16.4	6.9
Black Potts	Mean		8.1	8.0	8.3	8.7
	95%		14.2	12.6	9.3	13.6
	5%		6.2	6.2	6.7	6.7

Table A7.17

Determinand : Dissolved Oxygen
YEAR : 1976

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Boveney	Mean	11.0	11.5	11.5	11.0	11.1
	95%	17.5	19.9	19.1	17.6	18.3
	5%	0.2	0.2	0.2	0.2	0.2
Romney	Mean	11.0	11.5	11.5	11.0	11.2
	95%	17.8	20.4	19.6	17.8	18.7
	5%	0.2	0.3	0.3	0.2	0.2
Datchet	Mean	10.7	9.6	9.6	10.8	9.9
	95%	17.5	14.5	14.9	17.6	14.7
	5%	0.3	0.2	0.2	0.3	0.2
Marsh Lane	Mean		7.9	7.6	1.2	6.5
	95%		12.8	12.3	5.4	12.5
	5%		0.0*	0.0*	0.0	0.0*
Black Potts	Mean		6.8	6.4	2.5	5.2
	95%		12.9	12.5	8.6	12.5
	5%		0.0*	0.0*	0.0	0.0*

* The zero dissolved oxygen concentrations occur as a result of high BOD levels. These elevated levels are as a result of the large algal blooms which it has been predicted may occur in the channel.

Table A7.18

Determinand : BOD
YEAR : 1976

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Boveney	Mean	3.5	3.3	3.3	3.5	3.4
	95%	6.1	6.1	6.1	6.3	6.2
	5%	1.9	1.9	1.9	1.8	1.9
Romney	Mean	3.5	3.4	3.4	3.5	3.4
	95%	6.2	5.9	6.0	6.2	6.0
	5%	2.0	2.2	2.1	2.0	2.1
Datchet	Mean	3.8	4.6	4.7	3.3	5.3
	95%	6.4	9.1	9.1	5.9	9.4
	5%	2.0	1.9	2.0	1.9	2.1
Marsh Lane	Mean		4.0	3.9	1.2	4.9
	95%		6.8	6.8	1.5	10.3
	5%		1.6	1.6	0.8	1.7
Black Potts	Mean		6.4	6.5	1.1	11.8
	95%		17.7	17.7	1.4	30.4
	5%		1.6	1.9	0.7	2.3

Table A7.19

Determinand : Ammonium
YEAR : 1976

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Boveney	Mean	0.19	0.23	0.21	0.19	0.21
	95%	0.44	0.58	0.51	0.44	0.49
	5%	0.04	0.03	0.03	0.04	0.03
Romney	Mean	0.22	0.27	0.25	0.22	0.24
	95%	0.51	0.64	0.57	0.51	0.55
	5%	0.07	0.06	0.06	0.07	0.07
Datchet	Mean	0.20	0.16	0.17	0.19	0.18
	95%	0.47	0.38	0.41	0.46	0.43
	5%	0.04	0.04	0.04	0.04	0.04
Marsh Lane	Mean		0.10	0.08	0.07	0.09
	95%		0.29	0.24	0.10	0.26
	5%		0.01	0.01	0.02	0.01
Black Potts	Mean		0.08	0.05	0.05	0.08
	95%		0.25	0.11	0.10	0.10
	5%		0.01	0.01	0.02	0.02

Table A7.20

Determinand : Orthophosphate
YEAR : 1976

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Boveney	Mean	0.26	0.19	0.21	0.26	0.23
	95%	0.42	0.33	0.37	0.42	0.38
	5%	0.03	0.05	0.03	0.03	0.03
Romney	Mean	0.33	0.33	0.33	0.33	0.32
	95%	0.46	0.47	0.47	0.46	0.45
	5%	0.05	0.09	0.05	0.05	0.05
Datchet	Mean	0.23	0.13	0.15	0.23	0.18
	95%	0.36	0.20	0.26	0.35	0.27
	5%	0.05	0.04	0.05	0.05	0.05
Marsh Lane	Mean		0.20	0.16	0.01	0.11
	95%		0.37	0.31	0.01	0.23
	5%		0.00	0.00	0.01	0.00
Black Potts	Mean		0.03	0.02	0.01	0.01
	95%		0.09	0.03	0.01	0.03
	5%		0.00	0.01	0.01	0.01

Table A7.21

Determinand : Unionized Ammonia
YEAR : 1976

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	0.0056	0.0056	0.0055	0.0056	0.0057
	95%	0.0118	0.0142	0.0136	0.0117	0.0131
	5%	0.0019	0.0017	0.0017	0.0019	0.0018
Romney	Mean	0.0057	0.0054	0.0053	0.0057	0.0056
	95%	0.0111	0.0107	0.0108	0.0111	0.0111
	5%	0.0021	0.0020	0.0019	0.0021	0.0022
Datchet	Mean	0.0047	0.0036	0.0038	0.0047	0.0043
	95%	0.0094	0.0067	0.0071	0.0093	0.0084
	5%	0.0025	0.0017	0.0017	0.0019	0.0018
Marsh Lane	Mean		0.0021	0.0019	0.0055	0.0024
	95%		0.0034	0.0034	0.0130	0.0053
	5%		0.0009	0.0008	0.0006	0.0008
Black Potts	Mean		0.0026	0.0023	0.0049	0.0034
	95%		0.0064	0.0064	0.0137	0.0092
	5%		0.0009	0.0006	0.0007	0.0008

Table A7.22

Determinand : Flow
YEAR : 1983/4

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Bovency	Mean	43.9	22.0	38.8	43.8	36.4
	95%	122.1	63.3	117.1	122.1	112.1
	5%	15.5	7.7	10.5	15.2	11.6
Romney	Mean					
	95%					
	5%					
Datchet	Mean	44.8	44.5	44.4	44.8	44.4
	95%	123.1	122.7	122.7	123.1	122.7
	5%	16.4	16.1	16.4	16.2	16.2
Marsh Lane	Mean		21.8	4.9	0.3	7.4
	95%		60.7	4.9	0.3	9.9
	5%		7.7	4.9	0.3	3.9
Black Potts	Mean		22.0	5.1	0.5	7.5
	95%		60.9	5.1	0.5	10.1
	5%		7.9	5.1	0.5	4.1

Table A7.23

Determinand : Nitrate
YEAR : 1983/4

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Boveney	Mean	7.3	7.2	7.3	7.3	7.3
	95%	·10.4	10.4	10.4	10.4	10.4
	5%	4.7	4.5	4.6	4.7	4.6
Romney	Mean	7.4	7.4	7.4	7.4	7.4
	95%	10.5	10.4	10.5	10.5	10.5
	5%	4.8	4.7	4.7	4.8	4.7
Datchet	Mean	7.4	7.4	7.4	7.4	7.4
	95%	10.4	10.4	10.4	10.4	10.4
	5%	4.8	4.9	5.0	4.8	5.0
Marsh Lane	Mean		7.5	8.1	19.7	7.9
	95%		10.4	10.9	21.4	10.7
	5%		5.3	5.7	17.0	5.6
Black Potts	Mean		7.6	7.7	8.1	7.6
	95%		10.5	10.0	9.2	10.2
	5%		5.3	5.9	7.o	5.8

Table A7.24

Determinand : Dissolved Oxygen
YEAR : 1983/4

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Boveney	Mean	9.6	9.5	9.5	9.6	9.5
	95%	13.9	14.6	14.0	13.9	14.1
	5%	3.9	3.3	3.6	3.9	3.7
Romney	Mean	9.5	9.4	9.4	9.5	9.5
	95%	14.0	14.9	14.1	14.0	14.3
	5%	3.9	3.2	3.5	3.9	3.6
Datchet	Mean	9.4	9.6	9.3	9.4	9.3
	95%	13.9	12.5	10.6	14.0	12.9
	5%	3.8	4.2	3.4	3.7	3.4
Marsh Lane	Mean		9.8	8.7	10.8	9.1
	95%		12.4	11.7	2.3	12.5
	5%		5.3	3.5	0.10	3.5
Black Potts	Mean		9.8	8.0	2.0	8.8
	95%		12.5	11.9	4.3	12.2
	5%		5.8	0.12	0.0	3.0

Table A7.25

Determinand : BOD
YEAR : 1983/4

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	2.7	2.6	2.7	2.7	2.7
	95%	7.0	6.9	7.0	7.0	7.0
	5%	1.0	1.0	1.0	1.0	1.0
Romney	Mean	2.7	2.6	2.7	2.7	2.7
	95%	7.0	6.8	6.9	7.0	6.9
	5%	1.0	1.0	1.1	1.0	1.1
Datchet	Mean	2.7	2.5	2.7	2.6	2.7
	95%	7.0	6.9	7.1	6.9	7.1
	5%	1.0	1.0	1.1	1.0	1.0
Marsh Lane	Mean		2.7	2.6	1.1	2.6
	95%		7.2	6.9	1.6	7.1
	5%		1.0	1.0	0.8	1.0
Black Potts	Mean		2.5	3.1	0.9	2.8
	95%		7.1	8.6	1.2	7.9
	5%		1.0	1.0	0.7	0.9

Table A7.26

Determinand : Ammonium
YEAR : 1983/4

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	0.12	0.11	0.12	0.12	0.11
	95%	0.28	0.26	0.27	0.28	0.37
	5%	0.03	0.02	0.03	0.03	0.03
Romney	Mean	0.14	0.14	0.14	0.14	0.14
	95%	0.27	0.26	0.27	0.27	0.27
	5%	0.05	0.06	0.05	0.05	0.05
Datchet	Mean	0.13	0.10	0.12	0.13	0.11
	95%	0.24	0.21	0.24	0.27	0.23
	5%	0.04	0.03	0.04	0.04	0.04
Marsh Lane	Mean		0.09	0.07	0.07	0.08
	95%		0.23	0.18	0.10	0.20
	5%		0.01	0.01	0.02	0.01
Black Potts	Mean		0.07	0.04	0.05	0.05
	95%		0.18	0.08	0.10	0.12
	5%		0.01	0.01	0.02	0.01

Table A7.27

Determinand : Orthophosphate
YEAR : 1983/4

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	0.51	0.38	0.46	0.51	0.46
	95%	0.70	0.54	0.65	0.70	0.66
	5%	0.33	0.28	0.32	0.33	0.32
Romney	Mean	0.52	0.45	0.49	0.52	0.49
	95%	0.69	0.57	0.65	0.69	0.64
	5%	0.34	0.32	0.34	0.34	0.34
Datchet	Mean	0.45	0.28	0.36	0.45	0.34
	95%	0.61	0.38	0.51	0.61	0.45
	5%	0.33	0.22	0.28	0.33	0.27
Marsh Lane	Mean		0.52	0.3	0.01	0.35
	95%		0.73	0.46	0.01	0.48
	5%		0.32	0.14	0.01	0.21
Black Potts	Mean		0.18	0.03	0.01	0.05
	95%		0.30	0.04	0.01	0.10
	5%		0.09	0.02	0.01	0.02

Table A7.28

Determinand : Unionized Ammonia
YEAR : 1983/4

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	0.0024	0.0021	0.0023	0.0024	0.0023
	95%	0.0048	0.0042	0.0047	0.0048	0.0045
	5%	0.0011	0.0008	0.0010	0.0011	0.0010
Romney	Mean	0.0029	0.0029	0.0029	0.0029	0.0029
	95%	0.0053	0.0055	0.0053	0.0053	0.0053
	5%	0.0013	0.0013	0.0013	0.0013	0.0013
Datchet	Mean	0.0027	0.0021	0.0024	0.0027	0.0024
	95%	0.0051	0.0040	0.0047	0.0050	0.0044
	5%	0.0012	0.0010	0.0011	0.0012	0.0011
Marsh Lane	Mean		0.0018	0.0015	0.0056	0.0016
	95%		0.0041	0.0035	0.0116	0.0036
	5%		0.0006	0.0005	0.0010	0.0004
Black Potts	Mean		0.0013	0.0011	0.0044	0.0012
	95%		0.0031	0.0027	0.0155	0.0027
	5%		0.0004	0.0004	0.0011	0.0004

Table A7.29

Determinand : Flow
YEAR : 1989

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Boveney	Mean	43.7	21.9	38.3	43.2	36.5
	95%	148.3	75.2	143.3	148.3	138.3
	5%	13.8	6.9	8.8	13.5	10.3
Romney	Mean					
	95%					
	5%					
Datchet	Mean	44.7	44.3	44.3	44.6	44.3
	95%	149.2	148.8	148.9	149.2	148.8
	5%	14.7	14.3	14.3	14.4	14.4
Marsh Lane	Mean		21.8	5.3	0.8	7.2
	95%		73.1	4.9	0.4	9.9
	5%		6.9	4.9	0.3	3.4
Black Potts	Mean		21.9	5.5	1.0	7.3
	95%		73.3	5.1	0.8	10.1
	5%		7.1	5.1	0.5	3.6

Table A7.30

Determinand : Nitrate
YEAR : 1989

			SCENARIO			
		Without Channel	50%	FS5	FNS	FS10
Boveney	Mean	6.4	6.3	6.4	6.4	6.4
	95%	7.9	7.9	6.9	7.9	7.9
	5%	4.5	4.1	4.3	4.5	4.3
Romney	Mean	6.6	6.5	6.6	6.6	6.6
	95%	8.0	7.9	8.0	8.0	8.0
	5%	4.7	4.6	4.6	4.7	4.7
Datchet	Mean	6.6	6.6	6.7	6.6	6.7
	95%	8.0	8.0	8.0	8.0	8.0
	5%	4.7	5.0	5.1	4.7	5.1
Marsh Lane	Mean		6.9	7.3	19.3	7.2
	95%		8.1	8.7	21.5	8.4
	5%		5.4	5.6	16.5	5.7
Black Potts	Mean		6.9	7.2	7.2	7.2
	95%		8.0	8.2	8.4	8.2
	5%		5.4	6.1	6.6	5.8

Table A7.31

Determinand : Dissolved Oxygen
YEAR : 1989

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	9.9	10.3	10.1	9.9	10.1
	95%	12.4	13.8	12.7	12.4	12.8
	5%	1.5	1.4	1.4	1.5	1.4
Romney	Mean	10.0	10.3	10.2	10.0	10.1
	95%	12.7	14.4	13.4	12.7	13.3
	5%	1.5	1.5	1.5	1.5	1.5
Datchet	Mean	9.0	9.8	9.6	10.0	9.3
	95%	11.6	12.2	12.2	12.8	10.6
	5%	1.2	1.4	1.2	1.5	0.6
Marsh Lane	Mean		9.2	8.3	1.1	7.8
	95%		11.5	11.1	3.0	11.0
	5%		1.3	0.4	0.0	0.0*
Black Potts	Mean		9.1	7.1	2.5	6.5
	95%		11.7	11.6	6.7	11.2
	5%		1.2	0.0*	0.0	0.0*

* The zero dissolved oxygen concentrations occur as a result of high BOD levels. These elevated levels are as a result of the large algal blooms which it has been predicted may occur in the channel.

Table A7.32

Determinand : BOD
YEAR : 1989

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	2.3	2.3	2.3	2.3	2.3
	95%	5.0	4.5	4.8	5.0	4.8
	5%	1.1	1.2	1.1	1.1	1.1
Romney	Mean	2.4	2.5	2.5	2.4	2.5
	95%	4.8	4.3	4.5	4.8	4.7
	5%	1.3	1.5	1.3	1.3	1.4
Datchet	Mean	2.5	2.5	2.8	2.4	2.7
	95%	4.8	4.9	5.4	4.6	5.5
	5%	1.3	1.2	1.3	1.3	1.3
Marsh Lane	Mean		2.3	2.4	1.2	2.4
	95%		5.3	5.6	1.5	5.7
	5%		1.0	1.1	0.9	1.1
Black Potts	Mean		2.5	3.9	1.1	3.7
	95%		6.0	7.8	1.5	9.1
	5%		1.1	1.2	0.9	1.2

Table A7.33

Determinand : Ammonium
YEAR : 1989

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	0.17	0.18	0.17	0.17	0.17
	95%	0.45	0.47	0.46	0.45	0.47
	5%	0.06	0.06	0.06	0.06	0.06
Romney	Mean	0.20	0.23	0.21	0.20	0.21
	95%	0.47	0.51	0.47	0.47	0.47
	5%	0.08	0.10	0.08	0.08	0.08
Datchet	Mean	0.18	0.15	0.17	0.18	0.17
	95%	0.45	0.40	0.43	0.45	0.43
	5%	0.07	0.06	0.07	0.07	0.07
Marsh Lane	Mean		0.12	0.09	0.07	0.10
	95%		0.33	0.28	0.10	0.28
	5%		0.03	0.03	0.02	0.03
Black Potts	Mean		0.08	0.05	0.05	0.06
	95%		0.20	0.11	0.10	0.11
	5%		0.02	0.02	0.02	0.02

Table A7.34

Determinand : Orthophosphate
YEAR : 1989

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	0.41	0.37	0.40	0.41	0.40
	95%	0.59	0.59	0.59	0.59	0.59
	5%	0.26	0.21	0.23	0.26	0.24
Romney	Mean	0.45	0.45	0.45	0.45	0.45
	95%	0.59	0.59	0.58	0.58	0.59
	5%	0.36	0.37	0.37	0.36	0.36
Datchet	Mean	0.39	0.26	0.33	0.39	0.32
	95%	0.57	0.47	0.54	0.55	0.53
	5%	0.27	0.16	0.19	0.27	0.22
Marsh Lane	Mean		0.33	0.18	0.02	0.22
	95%		0.53	0.22	0.01	0.34
	5%		0.19	0.15	0.01	0.11
Black Potts	Mean		0.14	0.02	0.02	0.04
	95%		0.39	0.02	0.01	0.07
	5%		0.03	0.02	0.01	0.01

Table A7.35

Determinand : Unionized Ammonia
YEAR : 1989

		Without Channel	SCENARIO 50%	FS5	FNS	FS10
Boveney	Mean	0.0084	0.0079	0.0082	0.0084	0.0083
	95%	0.0149	0.0138	0.0145	0.0149	0.0145
	5%	0.0035	0.0037	0.0036	0.0035	0.0035
Romney	Mean	0.0076	0.0067	0.0073	0.0076	0.0073
	95%	0.0127	0.0115	0.0123	0.0127	0.0122
	5%	0.0036	0.0038	0.0036	0.0036	0.0037
Datchet	Mean	0.0071	0.0057	0.0062	0.0070	0.0062
	95%	0.0119	0.0097	0.0105	0.0119	0.0104
	5%	0.0035	0.0031	0.0032	0.0035	0.0032
Marsh Lane	Mean		0.0071	0.0056	0.0058	0.0059
	95%		0.0137	0.0115	0.0132	0.0115
	5%		0.0029	0.0021	0.0012	0.0025
Black Potts	Mean		0.0048	0.0032	0.0045	0.0039
	95%		0.0091	0.0071	0.0132	0.0088
	5%		0.0014	0.0013	0.0013	0.0015