

## Prediction of water quality in lakes and reservoirs: Part II – Model calibration, sensitivity analysis and application

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### Abstract

A range was assigned to each of the parameters used in the ecological component of the DYRESM Water Quality model based on values found in the literature. The sensitivity of the model to changes in these parameters was determined by individually adjusting parameters to the maximum or minimum of their assigned ranges whilst keeping all other parameters at their assigned means. The effects of these changes were quantified through the mean, the vertical distribution and the temporal variation in chlorophyll *a* and dissolved oxygen concentrations over a 200 day period, using data for Prospect Reservoir. Parameters that influenced uptake kinetics of phosphorus by phytoplankton, the minimum internal concentration, the half saturation constant and the maximum uptake rate, were amongst the most important determinants of all three measures of chlorophyll *a*. These parameters were also important determinants of the mean concentration and the vertical distribution of dissolved oxygen, through effects on photosynthetic oxygen production. Sediment oxygen demand had a significant effect on the mean concentration and vertical distribution of dissolved oxygen, phytoplankton density altered the vertical distribution of chlorophyll *a*, and rates of phytoplankton growth, respiration and mortality influenced the mean concentration of chlorophyll *a*. The model parameters were calibrated for the 200 day period and the model was validated over an additional 306 days. The mean errors between simulated and measured temperatures, and dissolved oxygen and chlorophyll *a* concentrations, as percentages of the measured values, were 5.2, 9.9 and 24.1% respectively. Simulated nutrient concentrations (PO<sub>4</sub>-P, NO<sub>3</sub>-N, NH<sub>4</sub>-N, total phosphorus and total nitrogen) all reflected the general temporal and spatial trends observed in the measured data from Prospect Reservoir. © 1997 Elsevier Science B.V.

*Keywords:* Coupled hydrodynamic–ecological model; Prospect Reservoir; Validation; DYRESM Water Quality model

### 1. Introduction

In Part I, a combined ecological and hydrodynamic model (DYRESM Water Quality) for lakes

and reservoirs is described. The purpose of the model is to provide a quantitative description of the interactions that occur between physical and ecological processes, and the water quality consequences of these interactions. While the hydrodynamic component requires no site specific adjustment, the ecological submodels require calibration with field data from the system they seek to simulate.

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When there are many interactive state variables, it is important to ascertain those parameters to which the model result is most sensitive. Once identified, these parameters are the ones to which most of the calibration effort should be directed. The sensitivity of a parameter can be quantified in many ways, typically through examining changes in the major state variables.

With large numbers of state variables modelled and correspondingly large numbers of parameters to calibrate, rigorous calibration is often not a realistic objective for large water quality models (Beck, 1986a). However, with due care and diligence, the task may be simplified through repeated sensitivity analysis and trial-and-error calibration, until an acceptable outcome is achieved (Jørgensen, 1993).

The test of the calibration values selected, the validation, is the degree to which the model prediction matches field data. There are numerous mathematical techniques to calibrate model parameters (cf. Beck and Arnold, 1977; Jørgensen, 1993). All require that the range of acceptable parameter limits, determined either from laboratory or field observation, be adhered to in order that the model be imbued with the necessary realism (e.g. Jørgensen et al., 1981, Jørgensen et al., 1986). Iterative calibration using best-judgement tuning of parameters by the modeller, combined with sensitivity analysis, is commonly used to obtain parameter estimates (e.g. Scavia, 1980; Rossi et al., 1986). In the present study, we have opted to use this method.

Validation is generally carried out with a second, independent set of field data. In theory it is possible only to invalidate a model, since it cannot be shown that the model conforms to reality under all circumstances. In practice, validation is achieved when predictions from a model that has been calibrated and verified with one data set, give a good approximation of the behaviour of a second data set (Beck, 1986b). Once complete, validation indicates that the model can be used as a tool to make prognoses about the system for which it was calibrated.

This paper presents the results of a sensitivity analysis, calibration and validation of the water quality model described in Part I. The input data used for the model were collected at Prospect Reservoir, New South Wales, by the Sydney Water Board. Prospect Reservoir was chosen because of the quality of the

model input data; it is a secondary reservoir with inflows and outflows that are regulated and monitored closely.

## 2. Input data

Prospect Reservoir, 30 km west of Sydney, Australia, is a moderately large, mesotrophic reservoir supplying 80% of water used in the Sydney metropolitan area. It has a mean depth of 9 m, maximum depth of 24 m, surface area of  $5.25 \times 10^6$  m<sup>2</sup>, maximum storage volume of  $50 \times 10^6$  m<sup>3</sup>, and a mean hydraulic retention time of 35 days. Under present operating conditions, the water quality is generally high. It has not suffered from severe hypolimnetic oxygen depletion ( $< 2$  mg DO l<sup>-1</sup>) and only minor algal blooms have occurred in the past (Cannon et al., 1970).

All data used for the model were from existing records (Schladow and Hamilton, 1992). In several cases the frequency and spacing of the data fell short of the ideal for calibrating the water quality component of the model. Further, the levels recorded for critical variables, namely PO<sub>4</sub>-P, often approached the limits of resolution. Both of these factors combined to increase the uncertainty associated with the field data although the temporal and spatial resolution of the data was generally still adequate for the purposes of calibration.

Water levels and volumes of the inflows and outflows were from Sydney Water Board records. Presently the reservoir receives gauged inflows from the Warragamba Pipeline and the Upper Canal (see Fig. 1). Warragamba Pipeline water, comprising 75% of the inflow, is usually drawn from hypolimnetic water at the Warragamba Dam, and has a year-round temperature of approximately 14°C. The Upper Canal is an open channel that receives water from reservoirs in the Upper Nepean River catchment (Fig. 1). Water temperatures in the Canal are close to air temperature, fluctuating in the range 14°C–25°C. The canal is chloraminated on its passage to Prospect Reservoir, boosting ammonium concentrations from approximately 10 mg m<sup>-3</sup> to 500–700 mg m<sup>-3</sup>. The catchment surrounding Prospect Reservoir supplies less than 0.5% of the total inflow volume. The water level is normally maintained within 0.15–0.30 m of

full supply level. Three of the four gauged offtakes withdraw epilimnetic water as they are within 2 m of the surface. A fourth offtake is approximately 9 m below the surface. All offtakes usually operate, and each extracts a similar volume of water.

Two weekly vertical profiles of temperature, dissolved oxygen and chlorophyll *a* were taken at 1 m intervals to a depth of 18 m at the Valve House (Fig. 1). Depth integrated epilimnetic and hypolimnetic dissolved inorganic phosphorus, total phosphorus, nitrate, ammonium and Kjeldahl nitrogen were also taken at the Valve House. Measurements of nitrate and Kjeldahl nitrogen were used to estimate the total nitrogen concentration. Measurements of temperature, chlorophyll *a*, dissolved oxygen and nutrients for the inflows were recorded at two weekly intervals. No attempts were made to simulate different functional groups of phytoplankton or to simulate zooplankton because phytoplankton species composition and zooplankton biomass were measured only occasionally.

Meteorological data comprised daily shortwave radiation from Sydney Airport (30 km to the east), and daily rainfall, 9 a.m. and 3 p.m. cloud cover, wind speed, and wet and dry bulb temperatures from Prospect Reservoir. Vapour pressure was determined

from wet and dry bulb temperatures (Tennessee Valley Authority, 1972).

The total period of data suitable for model input extended for 506 days, from 12 April 1989 to 31 August 1990. A 200 day period within this, from 1 November 1989 to 20 May 1990, was used for the sensitivity analysis and calibration. The full 506 day period was used for the validation.

### 3. Sensitivity analysis

The ranges derived in Part I for each of the major model parameters were used to set allowable parameter limits for use in the model calibration (Table 1). As zooplankton biomass was low and had little effect on phytoplankton biomass (D. Cannon, Sydney Water Board, personal communication), parameters for zooplankton grazing were assigned fixed values according to those given in Part I. The use of these zooplankton parameters produced a daily removal rate of approximately 2% of the phytoplankton biomass for a zooplankton concentration of  $0.4 \text{ mg m}^{-3}$ , a chlorophyll *a* concentration of  $5 \text{ mg m}^{-3}$  and a water temperature of  $20^\circ\text{C}$ , values typical of the euphotic zone in Prospect Reservoir.

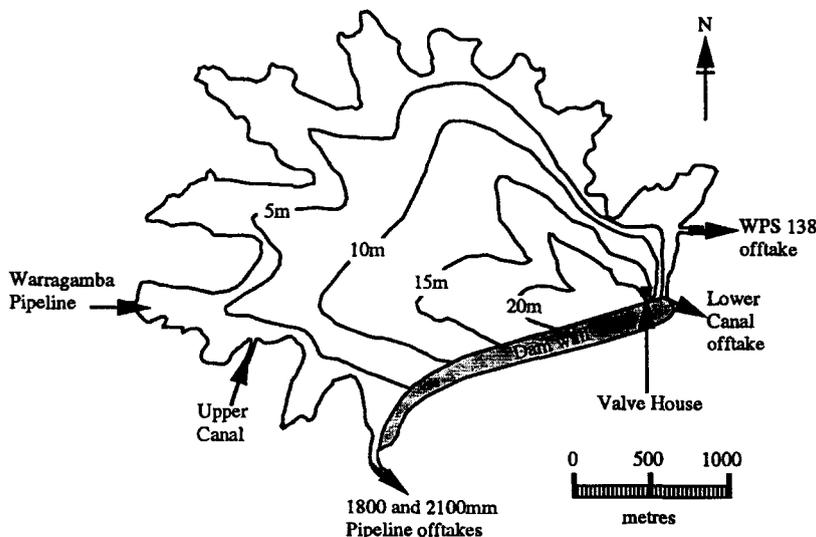


Fig. 1. Map showing location of Prospect Reservoir, inflows, outflows and Valve House.

The sensitivity of model results to changed values of the calibration parameters was quantified with reference to chlorophyll *a* (Chl*a*) and dissolved oxygen (DO) concentrations. These variables were considered to be two of the most critical determinants of water quality. For the sensitivity analysis all but one of the 29 adjustable parameters were fixed at the mean of their defined range given in Table 1. The model was then run twice for the 200 day calibration period, with the remaining free parameter set to the minimum of its assigned range for the first run and to the maximum of its assigned range for the second run. This process was repeated for each parameter in turn. Three measures of sensitivity were used in interpreting the model output. These were designed to test for changes in mean concentration

over the whole water column, changes in the vertical distribution, and changes in the temporal distribution of Chl*a* and DO. These latter two tests were considered necessary, as the mean concentration could remain unchanged in the presence of profound changes in either the vertical or temporal distribution.

To calculate changes in mean concentrations in the water column, the mean depth-averaged concentrations of Chl*a* and DO were calculated at every model timestep, and the maximum, minimum and mean values over the entire 200 day simulation were recorded. The values for Chl*a* and DO are shown in the histograms of Fig. 2a and b. For each of the parameters numbered on the horizontal axis two bars are displayed; the first corresponds to the use of the

Table 1  
Parameters used in the calibration and sensitivity analysis

Parameter	Definition	Units	Assigned range	Assigned value
1 $G_p$	maximum phytoplankton growth rate	day <sup>-1</sup>	1.3–3.6	2.02
2 $k_r$	phytoplankton respiration coefficient	day <sup>-1</sup>	0.05–0.17	0.085
3 $k_m$	phytoplankton mortality coefficient	day <sup>-1</sup>	0.01–0.10	0.02
4 $\vartheta_P$	phytoplankton temperature multiplier		1.02–1.14	1.09
5 $I_s$	phytoplankton saturating light	$\mu\text{E m}^{-2} \text{s}^{-1}$	100–500 <sup>a</sup>	100–220
6 $\eta_C$	specific extinction coefficient, Chl <i>a</i>	$\text{m}^2(\text{mg Chl } a)^{-1}$	0.015–0.025	0.02
7 $\eta_W$	background extinction coefficient	$\text{m}^{-1}$	0.25–0.65	0.45
8 $IP_{\min}$	minimum phytoplankton internal P	mg P (mg Chl <i>a</i> ) <sup>-1</sup>	0.1–1.0	0.20
9 $IP_{\max}$	maximum phytoplankton internal P	mg P (mg Chl <i>a</i> ) <sup>-1</sup>	1.0–5.0	1.24
10 $IN_{\min}$	minimum phytoplankton internal N	mg N (mg Chl <i>a</i> ) <sup>-1</sup>	1.5–4.0	2.35
11 $IN_{\max}$	maximum phytoplankton internal N	mg N (mg Chl <i>a</i> ) <sup>-1</sup>	8.0–15.0	9.0
12 $U_P$	maximum rate of phytoplankton P uptake	mg P (mg Chl <i>a</i> ) <sup>-1</sup> day <sup>-1</sup>	0.05–1.0	0.078
13 $U_N$	maximum rate of phytoplankton N uptake	mg N (mg Chl <i>a</i> ) <sup>-1</sup> day <sup>-1</sup>	0.5–10.0	0.85
14 $K_P$	half saturation constant for phytoplankton P uptake	mg m <sup>-3</sup>	1–25	2.75
15 $K_N$	half saturation constant for phytoplankton N uptake	mg m <sup>-3</sup>	20–200	22
16 $\rho_P$	density of phytoplankton	kg m <sup>-3</sup>	1005–1100	1025
17 $k_n$	rate coefficient for nitrification	day <sup>-1</sup>	0.005–0.05	0.037
18 $k_b$	rate coefficient for sediment oxygen demand	g m <sup>-2</sup> day <sup>-1</sup>	0.02–50.0	3.95
19 $k_{\text{BOD}}$	rate coefficient for breakdown of organic matter	day <sup>-1</sup>	0.001–0.025	0.008
20 $k_{\text{OP}}$	rate coefficient for organic P mineralisation	day <sup>-1</sup>	0.07–0.80	0.099
21 $k_{\text{ON}}$	rate coefficient for organic N mineralisation	day <sup>-1</sup>	0.03–0.3	0.03
22 $\rho_T$	density of particulate nutrients	kg m <sup>-3</sup>	1005–1200	1017
23 $\rho_B$	density of particulate organic matter	kg m <sup>-3</sup>	1001–1100	1013
24 $\vartheta_n$	temperature multiplier for nitrification		1.02–1.14	1.03
25 $\vartheta_B$	temperature multiplier for detrital breakdown		1.02–1.14	1.05
26 $S_P$	sediment phosphorus release	mg m <sup>-2</sup> day <sup>-1</sup>	0.0–5.0	0.013
27 $S_N$	sediment ammonium release	mg m <sup>-2</sup> day <sup>-1</sup>	0.0–10.0	0.04
28 $\vartheta_S$	temperature multiplier for sediments		1.02–1.14	1.04

<sup>a</sup> Values of  $I_s$  were fixed for the purpose of the sensitivity analysis.

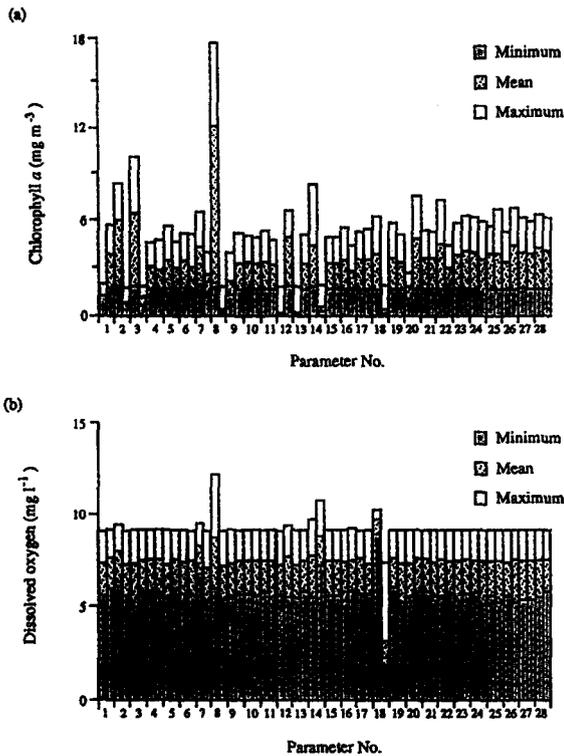


Fig. 2. Minimum, maximum and mean of the depth-averaged, daily concentrations of (a) chlorophyll *a* and (b) dissolved oxygen for the period of the sensitivity analysis (1 November 1989 to 20 May 1990). Each parameter, numbered according to Table 1, was adjusted first to the minimum and then to the maximum of its assigned range of Table 1.

assigned minimum value of that particular parameter, and the second to the use of the maximum value.

The histograms show that the model results for both Chl*a* and DO are only sensitive to a relatively small subset of the 29 parameters. Parameters for which Chl*a* was highly sensitive are those that directly alter growth rates ( $G_P$ ,  $k_r$  and  $k_m$ ) or indirectly affect growth rates through their ability to take up or utilise phosphorus ( $IP_{\min}$ ,  $U_P$  and  $K_P$ ). Chl*a* levels were only mildly sensitive to light penetration in the water column, nitrogen uptake ( $U_N$ ) and sediment oxygen demand ( $k_b$ ) for the assigned parameter ranges.

Even fewer parameters affected the minimum, mean and maximum levels of DO over the selected 200 day period. Sediment oxygen demand, for the range assigned to  $k_b$ , was clearly the dominant

parameter affecting the oxygen concentration. Dissolved oxygen levels were also influenced, however, by parameters that indirectly affected phytoplankton growth by changing phosphorus uptake dynamics (e.g.  $IP_{\min}$  and  $K_P$ ).

Changes in the vertical distribution were measured by examining variations in Chl*a* and DO with depth. For each timestep, the standard deviation in Chl*a* or DO over the depth of the water column was divided by the mean Chl*a* or DO over the water column, thus normalising the data. Fig. 3a and b show the maximum normalised values of Chl*a* and DO over the 200 day simulation period, corresponding to the minimum and maximum values of each parameter. The minimum values are not shown as

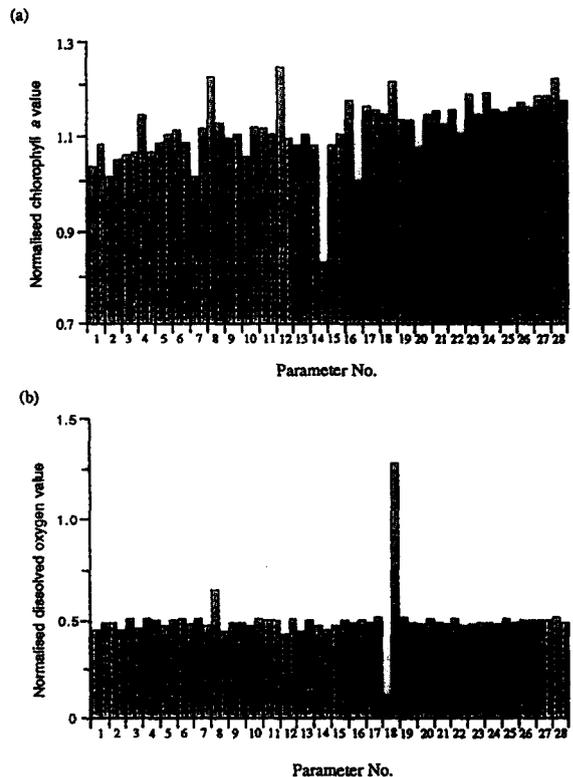


Fig. 3. Maximum of the standard deviation normalised by the mean daily concentration over the water column for (a) chlorophyll *a* and (b) dissolved oxygen for the period of the sensitivity analysis (1 November 1989 to 20 May 1990). Each parameter, numbered according to Table 1, was adjusted first to the minimum and then to the maximum of its assigned range of Table 1.

they were all zero, corresponding to times when the water column was homogeneous.

Five parameters had a large effect on the distribution of Chl *a* in the water column. Phosphorus uptake parameters ( $IP_{\min}$ ,  $U_P$  and  $K_P$ ) were once again important determinants of the vertical distribution of chlorophyll *a*, but changes to the values of  $\vartheta_P$  and  $\rho_P$  could also alter the distribution substantially. All but the latter variable affect the growth rate of phytoplankton. In a phosphorus limited system, uptake of phosphorus by phytoplankton in one part of the water column precludes its transport and utilisation in another part of the water column, thereby altering the vertical distribution. By contrast, changes in  $G_P$ ,  $k_r$  and  $k_m$  had little effect on the spatial distribution as they act uniformly over the vertical distribution. Thus they would only affect the mean concentration, as already shown. In a temperature stratified water column, altering the temperature response through changes in  $\vartheta_P$  would lead to different growth and loss responses over the water column. The remaining parameter,  $\rho_{Chl a}$ , directly affects the settling rate of phytoplankton, but is also influenced by the density and viscosity of the water column. Therefore, for the wide range assigned to this parameter, in a temperature stratified water column there will be different rates of phytoplankton loss over the water column.

The depth distribution of DO in the water column is most affected by the sediment oxygen demand parameter,  $k_B$ . The only other parameter to have a marked effect on the vertical distribution of DO was  $IP_{\min}$ , through its influence on phytoplankton biomass and the resultant oxygen production and consumption.

Sensitivity of the temporal distribution of Chl *a* and DO to changes in the various parameters was quantified from the distribution of the depth-averaged concentrations over the 200 day simulation period. The time when the centre of this distribution (first moment) and one standard deviation on either side of this distribution (second moment) were attained was quantified by the number of days since the start of the simulation.

The temporal distribution of DO was insensitive to changes in any of the parameter ranges, but Chl *a* distributions were variable (Fig. 4a and b). In particular, high values of  $IP_{\min}$ ,  $K_P$  and  $k_B$ , and low

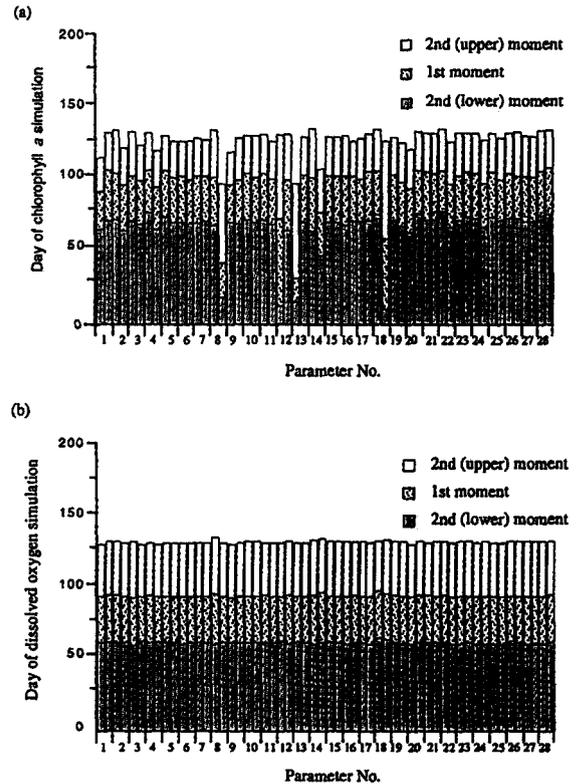


Fig. 4. First moment (mean) and second moments (mean  $\pm$  standard deviation) of the temporal distribution of daily, depth averaged (a) chlorophyll *a* and (b) dissolved oxygen concentrations for the period of the sensitivity analysis (1 November 1989 to 20 May 1990). Each parameter, numbered according to Table 1, was adjusted first to the minimum and then to the maximum of its assigned range of Table 1.

values of  $U_P$  and  $U_N$ , advanced the onset of algal production.

The parameters to which the model results are

Table 2

Sensitive model parameters with respect to mean concentration, vertical distribution and temporal distribution of chlorophyll *a* and dissolved oxygen concentrations. Definitions of parameters are given in Table 1 of Part I

	Chlorophyll <i>a</i>	Dissolved oxygen
Mean concentration	$G_P$ , $k_r$ , $k_m$ , $IP_{\min}$ , $UP_{\max}$ , $K_P$	$IP_{\min}$ , $K_P$ , $k_B$
Vertical distribution	$\vartheta_P$ , $IP_{\min}$ , $UP_{\max}$ , $K_P$ , $\rho_P$	$k_B$
Temporal distribution	$IP_{\min}$ , $UP_{\max}$ , $UN_{\max}$ , $K_P$ , $k_B$	

most sensitive are summarised in Table 2. It should be noted that this result is specific to Prospect Reservoir, although many of the parameters could be expected to produce similar results in other systems. As a consequence of the sensitivity analysis, however, the number of model parameters that formed the focus of the calibration was reduced from 29 to 10. Knowledge of the selective way in which the model result is affected by the most sensitive model parameters (i.e. the temporal or spatial distribution of Chl *a* or DO) further simplified the task of calibration.

#### 4. Calibration and verification

The ecological model was calibrated by trial-and-error adjustment of the most sensitive parameters to give the best match with trends in the measured field data over the 200 day period. Although only dissolved oxygen and chlorophyll *a* concentrations were used in identifying the most sensitive parameters, all the measured data (e.g. PO<sub>4</sub>-P, NO<sub>3</sub>-N, NH<sub>4</sub>-N, TP and TKN) were used in the model calibration. The parameter values found to give the best fit for Prospect Reservoir are given in Table 1.

The hydrodynamic component of the model was verified with nine years (1983–1991) of temperature data for Prospect Reservoir. The model produced accurate simulations of observed temperatures over this period. Because of the limited data available to verify the other state variables in the model, a two year period of relatively complete data (1989–1990) was replicated to give a total simulation period of ten years. This procedure verified the stability of the model and its ability to reproduce repeatable seasonal and annual trends in all of the state variables.

#### 5. Validation

The model validation used the measured input variables to simulate the observed reservoir behaviour over a 506 day simulation period, from 12 April 1989 to 31 August 1990, when a detailed data set was available. This period included the 200 days used for the sensitivity analysis and calibration of the ecological model. Thus, for the greater part of the

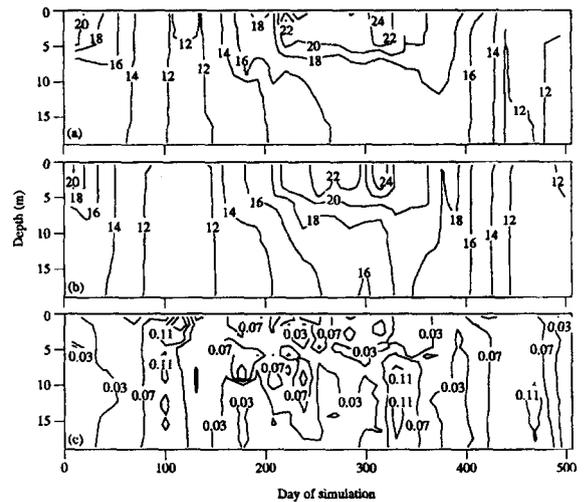


Fig. 5. (a) Measured and (b) simulated temperatures, and (c) error in simulated temperature as a fraction of the measured value, for the period 12 April 1989 and 31 August 1990.

period the model was predicting the reservoir behaviour.

The simulated temperatures of Fig. 5b correspond closely to the measured values of Fig. 5a. There is a strong density stratification during warmer months of the year with a relatively shallow surface mixed layer. Fig. 5c shows a contour plot of the differences between the measured and simulated data, where the differences are determined for each field measurement as (measured value – simulated value)/measured value. The mean difference between the measured and simulated temperatures was 5.2% of the measured temperature, with a standard deviation of 3.6%. This result was achieved independently of the ecological model calibration, although there is a feedback of the ecology to the hydrodynamics via the extinction coefficient (refer to Part I). It should be noted that the contour plot for the field data was constituted from the two weekly profiles taken at 1m intervals to a depth of 19 m whereas the simulated data represents daily output for each Lagrangian layer represented in the model.

The simulated dissolved oxygen concentrations of Fig. 6b are in reasonable agreement with the field measurements of Fig. 6a. The error variation, calculated as above for temperature, is shown as a contour plot in Fig. 6c. The mean difference between mea-

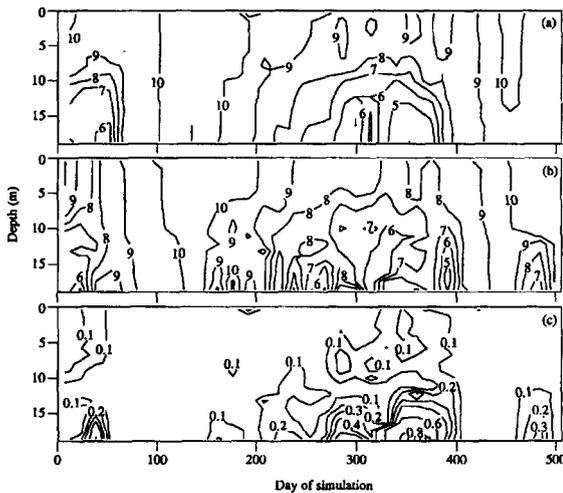


Fig. 6. (a) Measured and (b) simulated dissolved oxygen concentrations, and (c) error in simulated dissolved oxygen as a fraction of the measured value, for the period 12 April 1989 and 31 August 1990.

sured and simulated DO was 9.9% of the measured DO, with a standard deviation of 13.9%. Much of the variation between simulated and measured DO occurs close to the bottom of the water column where periodic insertions of oxygen derived from the inflows significantly elevate the simulated DO but are not seen over the temporal and spatial scale for which measurements are carried out (Hamilton and Schladow, 1994).

The simulation result for *Chl a*, shown in Fig. 7b, was achieved by striking a balance between those parameters affecting the temporal distribution of *Chl a* and those affecting its vertical distribution. Measured *Chl a* concentrations are shown in Fig. 7a and a contour plot of differences between the measured and simulated data over the water column is given in Fig. 7c. For this case, the differences were calculated as:  $[(\text{measured value} + 1.0) - (\text{simulated value} + 1.0)] / (\text{measured value} + 1.0)$  in order to avoid bias for cases when the measured chlorophyll *a* concentration was very close to zero. Using this method the mean difference between the measured and simulated values, as a mean over the water column and the 200 day period, was 24.1% of the measured *Chl a*, with a standard deviation of 19.7%. Variations between measured and simulated tempera-

ture, DO and *Chl a* are somewhat higher than the measurement errors ( $\pm 0.1^\circ\text{C}$ ,  $\pm 0.1 \text{ mg L}^{-1}$  and  $\pm 0.5 \mu\text{g L}^{-1}$ , respectively) but for DO and *Chl a*, consideration should be given to the wide natural variability of phytoplankton biomass, both spatially and temporally.

The distribution of *Chl a* is a complex mix of limiting effects induced by temperature, availability of light and nutrients, and the physical mixing processes that dictate the vertical distribution of a cell in the water column. When the water column in Prospect Reservoir was well mixed, phytoplankton concentrations were nearly evenly distributed vertically. At the onset of stratification the mixed layer provided more favourable conditions for phytoplankton growth while in deeper layers light was the major limiting factor.

Nutrient dynamics are of crucial importance to the ecological components of the model. The calibration procedure required a balance between phytoplankton growth and depletion of available nutrients from the water column. The broad trends of changes in measured nutrient concentrations, as well as temperature and dissolved oxygen, at specific depths, have been simulated in Figs. 8 and 9. Particular attention was placed on the dynamics of  $\text{PO}_4\text{-P}$  as this nutrient was the main one limiting phytoplankton biomass in Prospect Reservoir. Phosphorus concentrations were

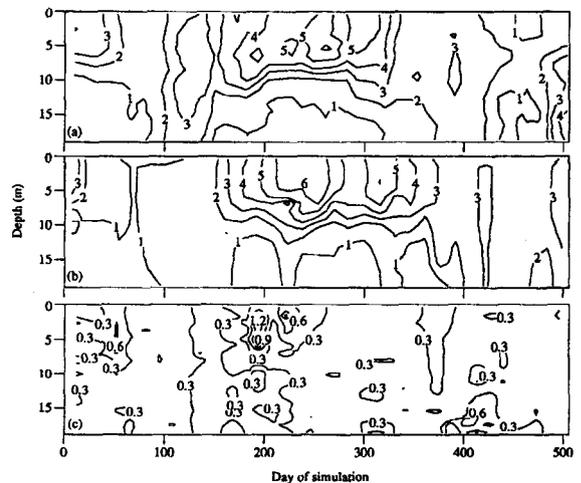


Fig. 7. (a) Measured and (b) simulated chlorophyll *a* concentrations, and (c) error in simulated chlorophyll *a* as a fraction of the measured value, for the period 12 April 1989 and 31 August 1990.

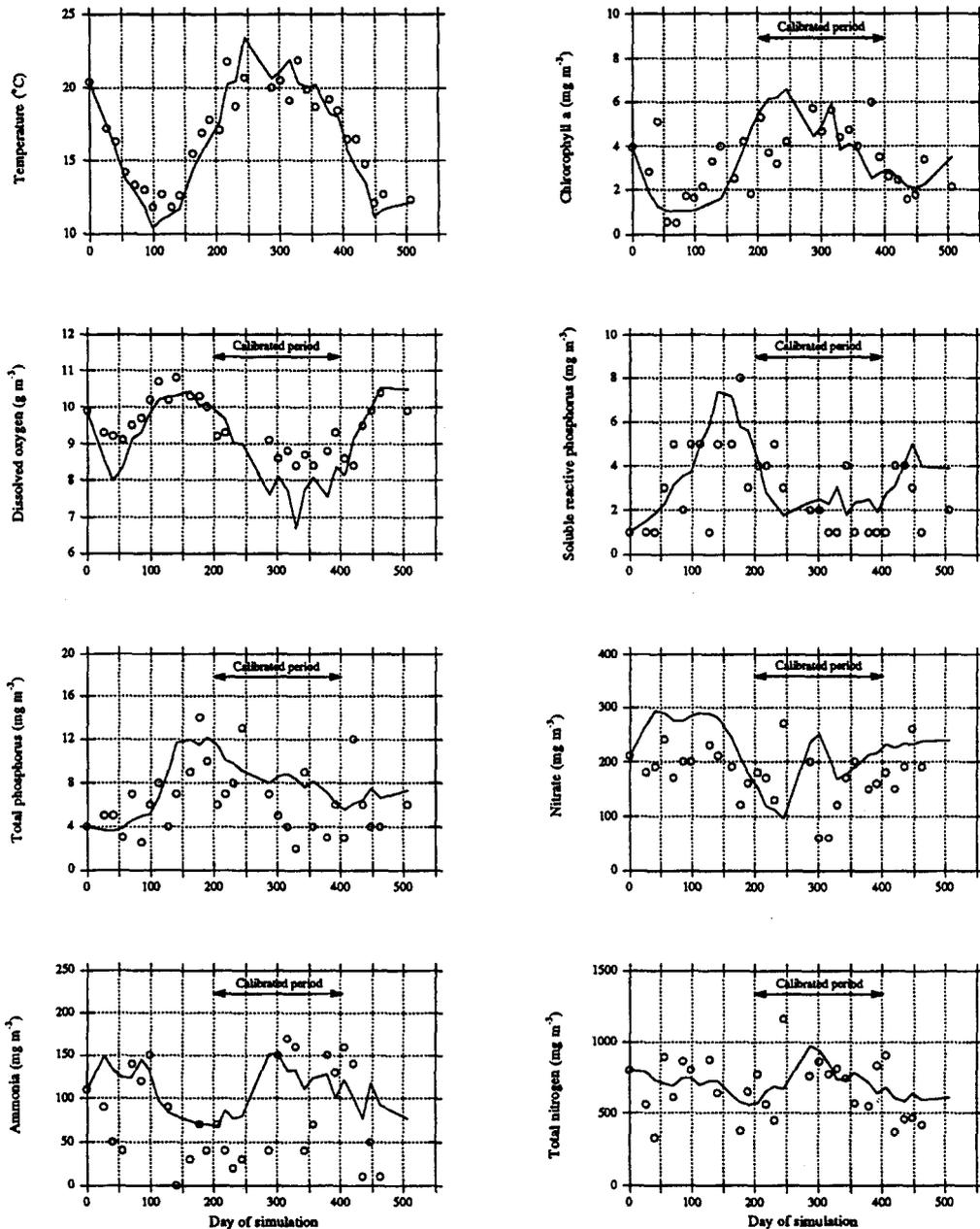


Fig. 8. Measured (o) and simulated (—) temperature, chlorophyll *a*, dissolved oxygen and nutrients for epilimnetic samples from Prospect Reservoir for the period 12 April 1989 to 31 August 1990.

generally lower in the mixed layer, reflecting greater nutrient depletion caused by larger phytoplankton populations near the surface, and higher nearer the

bottom where the population is limited primarily by light availability. Nutrient release from the sediments was not considered important in the phosphorus dy-

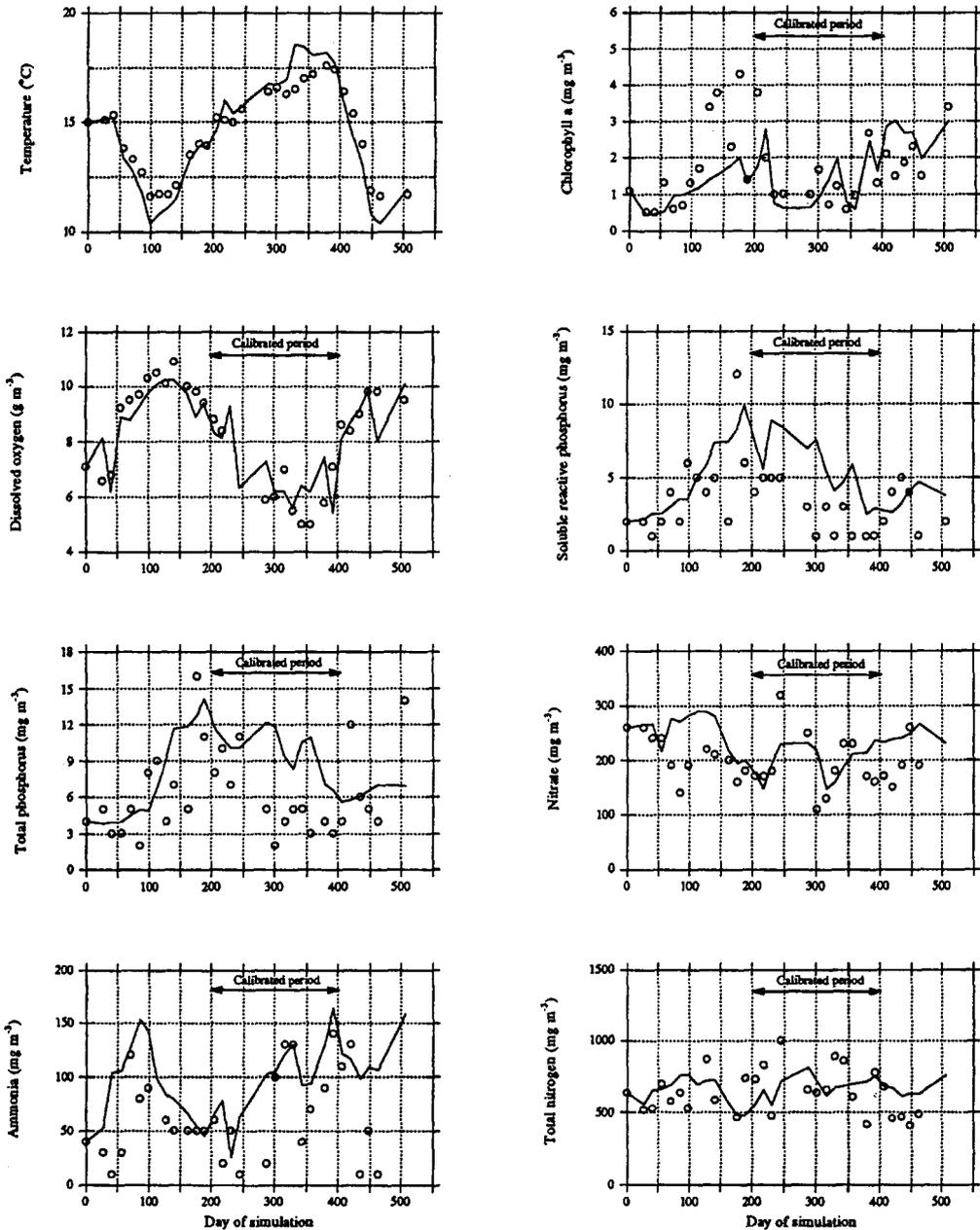


Fig. 9. Measured (o) and simulated (—) temperature, chlorophyll *a*, dissolved oxygen and nutrients for hypolimnetic samples from Prospect Reservoir for the period 12 April 1989 to 31 August 1990.

namics as the overlying water column remained well oxygenated.

## 6. Discussion

Two important benefits were gained from the sensitivity analysis. Firstly it allowed a reduction in

the total number of parameters to be adjusted. Secondly, it identified which parameters affected the mean, the temporal and the spatial distributions of certain variables (i.e. Chl *a* and DO). For example, the minimum internal phosphorus concentration,  $IP_{min}$ , and the half saturation constant for phosphorus uptake,  $K_p$ , exerted obvious effects on the temporal

distribution of Chl *a* over the verification period. When each of these parameters was adjusted to the minimum of its assigned range it allowed phytoplankton to access the available nutrients more rapidly, moving the temporal concentration distribution towards the start of the simulation. The nutrient uptake parameters  $UP_{max}$  and  $UN_{max}$  worked in the opposite way by lowering the respective uptake rates and preserving dissolved inorganic nutrients at high levels until later in the sensitivity analysis period. A high value of the biological sediment oxygen demand,  $k_b$ , shifted the temporal distribution of Chl *a* towards the start of the simulation, triggering rapid onset of sediment anoxia and associated nutrient release. It should be noted, however, that the parameters for sediment nutrient release remained largely unvalidated as Prospect Reservoir had not previously experienced oxygen depletion of a magnitude that would allow substantial anoxic release of nutrients from the sediments (Schladow and Hamilton, 1995).

Under the present flow conditions, the depth of the mixed layer is entirely due to the interplay between the inflows, the outflows and the surface mixing processes. There is a strong thermal stratification and shallow mixed layer depth during warmer months of the year (December to April). The relatively large flow of cold water from the Warragamba Pipeline inserts into the hypolimnion while at the same time the bulk of the withdrawal comes from the surface. Thus the thermocline is maintained at an artificially high level and there is not the characteristic deepening of the mixed layer prior to turnover which is commonly observed in lakes of temperate regions. By contrast, the temperature of the Upper Canal is similar to that of the mixed layer. This inflow therefore inserts near the surface and does not affect the mixed layer depth greatly as it is volumetrically inferior to the Warragamba inflow.

The temporal distribution of dissolved oxygen through the water column is largely a reflection of the prevailing stratification. At turnover, oxygen is redistributed throughout the water column to homogeneous levels around  $2 \text{ mg l}^{-1}$  below saturation. Oxygen levels peak soon after restratification when the cold water is saturated. During the stratified period oxygen produced through photosynthesis is confined primarily to the epilimnion and although oxygen derived from the Warragamba inflow is

highly important to the inventory in the hypolimnion, it does not compensate completely for biochemical utilisation of oxygen in the water column and the sediments. Von Winterberg et al. (1985) suggested that the relatively slow rate of hypolimnetic oxygen depletion in Prospect Reservoir was due to significant photosynthetic oxygen production in the hypolimnion. However, Hamilton and Schladow (1994) traced the various sources and sinks of dissolved oxygen using DYRESM Water Quality and showed that the only significant source of oxygen was from the Warragamba inflow. This inflow is generally saturated in oxygen and, in contrast to the Upper Canal inflow, plunges upon entering Prospect Reservoir. Relatively wide variations in DO in the hypolimnion occur in response to variations in the depth of insertion of the Warragamba inflow (Hamilton and Schladow, 1994) and these variations also contribute to marked deviations between measured and simulated DO near the bottom of the water column (Fig. 6). Vincent et al., 1991 have also shown that inputs of dissolved oxygen from a plunging river inflow can have a significant effect on the overall oxygen budget in a lake.

Under the present flow regime, phytoplankton concentrations in Prospect Reservoir are largely insensitive to maximum sediment  $\text{PO}_4\text{-P}$  release rates in the range  $0\text{--}0.9 \text{ mg m}^{-2} \text{ day}^{-1}$ , even though previous studies (e.g. Petrie and Smalls, 1981; OECD, 1982) and the sensitivity analysis of the present study indicate that phytoplankton biomass is strongly phosphorus limited. The possibility of nitrogen limitation may be further averted by chloramination of the Upper Canal. The hydraulic residence time in Prospect Reservoir is around 35 days but could be considerably lower in the hypolimnion due to the high Warragamba through-flow. Moderate increases in phosphorus input from the sediments are therefore not reflected in increased Chl *a* due to a combination of inhibited mixing across the thermocline and rapid flushing of the hypolimnion, which prevent severe anoxia of the bottom sediments.

The validation showed that the coupled hydrodynamic-water quality model simulated the main temporal and spatial trends in all of the state variables that were modelled (Figs. 8 and 9). Some restratification of oxygen evident around day 470 of the simulation was not observed in the field data. This may

have resulted from inaccuracies in interpolations between temperature measurements in the Warragamba inflow, which would have altered the depth of insertion of the saturated Warragamba inflow. This serves to illustrate the importance of a model that takes account of all the mixing processes when simulating the vertical and temporal distribution of dissolved oxygen and other water quality indicators.

## 7. Conclusion

The calibration, sensitivity analysis and validation adopted in this study form the basis of a procedure to derive an ecologically sound eutrophication model that may eventually be free from substantial calibration. The knowledge acquired from our sensitivity analysis has allowed for identification through quantitative methods of the most critical ecological parameters. Verification of values of these parameters, based on the results of the present study, has been undertaken for Prospect Reservoir (Pickering, 1994). Deterministic process descriptions can then be formulated as specific data to which the model result is most sensitive are obtained. In addition, certain sub-models can be excluded or reduced according to the hierarchy of regulatory mechanisms (Straškraba, 1994). It is still important, however, that the model retains an element of flexibility to encompass potential changes in system behaviour, for example, changes in species composition, self-organisation aspects of the ecosystem or synergistic effects amongst components.

Finally, the model illustrates the importance of the process based hydrodynamic description used in DYRESM Water Quality for investigating the physical processes that influence water quality. An example is provided by Prospect Reservoir where the depth of insertion of the Warragamba inflow plays a key role in oxygenation of the hypolimnion and the resultant biological response. Extension of deterministic descriptions to the ecology will enhance the physical meaning of any remaining parameters and improve capabilities to explore interactions amongst the ecological components.

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