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CHLOROPHYLL *A* PRODUCTION MODELLING OF INSHORE WATERS

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ABSTRACT

Increased anthropogenic activity in many coastal regions is placing significant pressures on inshore waters. Many of the estuaries and bays accepting the discharges from large conurbations are heavily polluted preventing their use for recreational activities. The relatively poor water quality in parts of Dublin Bay and Cork Harbour are examples of adversely impacted waters. Discharges of nutrients, in particular, through outfalls and rivers can lead to the occurrence of algal blooms with the associated environmental problems. Computer-based models have been developed to investigate the relationships between nutrient inputs and chlorophyll *a* production in inshore waters. These models have been applied to Cork Harbour and used to perform scenario modelling. Two scenarios are simulated using the models: the first considers the effects of discharging significant nutrient loads into Cork Harbour, and the second scenario considers the water quality when there are no discharges from domestic and industrial outfalls. Good agreement was obtained between model predicted chlorophyll *a* and measured data. There is significant reduction in chlorophyll *a* production during the latter simulation, primarily due to the reduced phosphorus loads. The application of the model to Cork Harbour illustrates how spatially and temporally refined models can be used to assist total water quality management of inshore waters.

INTRODUCTION

Traditionally, the collection and analysis of physical, chemical and biological field data were the main techniques used to quantify the movement and quality of coastal waters. Since about the mid-1980's computer models have been applied with increasing frequency to coastal water quality issues (Townend 1994). Currently, it is common practice to combine field measurements and modelling studies to achieve the best results in estuarine and coastal water quality management. One of the main stimuli leading to research into the development of water quality models has been the significant amount of legislation that was enacted pertaining to water quality. National legislation and EU directives, such as the European Parliament Water Framework Directive (WFD) 2000/60/EC, prescribe limits on various water quality parameters. Integrated modelling and field measurement studies have proven to be amongst the best approaches to ensure compliance with these legal instruments. The central role of water modelling in implementing the WFD has been widely accepted throughout the European Union. The European network EurAqua recently published a review of the modelling as a tool in implementing the Directive (Euraqua 2001). Fourteen EU countries present papers in this review underpinning modelling requirements in water management.

The research community at large have demonstrated their commitment to long-term modelling activities through various international collaborative activities. Large-scale networks such the EU supported European Land-Ocean Interaction Studies (ELOISE) and its global counterpart Land Ocean Interface in the Coastal Zone (LOIZC) demonstrate the significant role that modelling now plays in helping scientist and engineers understand coastal processes and manage this sensitive environment (Mathy 2001).

The trophic status of estuaries is currently a major worldwide environmental issue and is the subject of considerable research. Many Irish estuaries are considered to be eutrophic due to diverse anthropogenic activities which interact with the coastal zone (EPA 2001). Coupled physical/chemical/biological models are required to understand complex marine processes and are used to investigate the effects of anthropogenic activities through scenario modelling. Such models are also useful tools in establishing trophic indices for water bodies.

In the following sections, details are provided of a water quality model and its application to Cork Harbour. In applying the model to Cork Harbour, the author investigated the water quality impacts of current discharge regimes into the Harbour and the likely improvements in water quality if all domestic and industrial discharges were switched off. By comparing the results from these two scenarios it is possible to attempt to ascertain some of the impacts of urbanisation on the water quality of Cork Harbour. This study also illustrates how water quality scenario modelling may be utilised to guide effluent treatment in order to achieve acceptable quality of the receiving waters.

WATER QUALITY MODELLING

The model, DIVAST, used in this research is a two-dimensional model which solves for various parameters on a horizontal plane, assuming homogeneity in the vertical plane. This assumption is generally found to hold well in shallow coastal waters, (Falconer 1986). The model contains three main sub-modules: hydrodynamic, solute transport and water quality. Each of these sub-modules is briefly outlined below.

Hydrodynamic module

The governing differential equations used in the numerical model to determine the water elevation and depth-averaged velocity fields in a horizontal plane are based on integrating the three-dimensional Navier-Stokes equations over the water column depth. This results in a two-dimensional model which resolves variables in two mutually perpendicular horizontal directions (x and y). It is assumed that the vertical accelerations are negligible compared with gravity, i.e. the existence of a quasi-hydrostatic pressure distribution, and that the Reynolds stresses in the vertical plane can be represented by a Boussinesq approximation. The depth-integrated continuity and x-direction momentum equations can be shown (Falconer 1977) to be given by equations (1) and (2) respectively:

$$\frac{\partial \zeta}{\partial t} + \frac{\partial q_x}{\partial x} + \frac{\partial q_y}{\partial y} = 0 \qquad (1)$$

$$\frac{\partial q_x}{\partial t} + \beta \left[\frac{\partial U q_x}{\partial x} + \frac{\partial V q_y}{\partial y} \right] = \qquad (2)$$

$$f q_y - gH \frac{\partial \zeta}{\partial x} + \frac{\tau_{xw}}{\rho} + \frac{\tau_{xb}}{\rho} + 2\frac{\partial q_x}{\partial t} \left[\epsilon H \frac{\partial U q_x}{\partial x} \right] + \frac{\partial}{\partial y} \left[\epsilon H \left[\frac{\partial U}{\partial y} + \frac{\partial V}{\partial x} \right] \right]$$

where
ζ = water surface elevation above mean water level
t = time
q_x, q_y = depth integrated velocity flux components in the x,y directions
β = momentum correction factor
U, V = depth integrated velocity components in the x,y directions
f = Coriolis parameter
g = gravitational acceleration
H = total depth of water column
τ_{xw} = surface wind shear stress component in the x direction
τ_{xb} = bed shear stress component in the x direction
ρ = fluid density

A depth-integrated momentum equation analogous to equation (2) is also developed for the y-direction.

Solute transport module

Solute transport refers to the mechanics of the movement of solutes in water due to the above hydrodynamics and turbulence within the water. These transport effects are incorporated into the model through the well-established advection-diffusion equation (Falconer and Liu 1988). This equation in two-dimensions is:

$$\frac{\partial \phi H}{\partial t} + \left[\frac{\partial \phi Uh}{\partial x} + \frac{\partial \phi Vh}{\partial y}\right] - \frac{\partial}{\partial x}\left[HDxx\frac{\partial \phi}{\partial x} + HDxy\frac{\partial \phi}{\partial y}\right]$$
$$- \frac{\partial}{\partial y}\left[HDyx\frac{\partial \phi}{\partial y} + HDyy\frac{\partial \phi}{\partial y}\right] - H\left[S_o + S_d + S_k\right] = 0 \tag{3}$$

where

ϕ = solute concentration

Dxx, Dxy, Dyx, Dyy = depth averaged longitudinal dispersion and turbulent diffusion coefficients in the x,y directions

S_o = source or sink input

S_d = first order decay rate or growth rate of the solute

S_k = total kinetic transportation rate

Water Quality Module

The water quality module simulates the nitrogen, phosphorus, oxygen and chlorophyll *a* cycles and some of their main interactions. Figure 1 presents a schematic representation of the model used; by inspection, it is clear that the chlorophyll *a* cycle is at the centre of the system. This module includes interactions between the following variables:

salinity	ammoniacal nitrogen
temperature	nitrate nitrogen
BOD	organic phosphorus
dissolved oxygen	orthophosphate
organic nitrogen	chlorophyll *a*

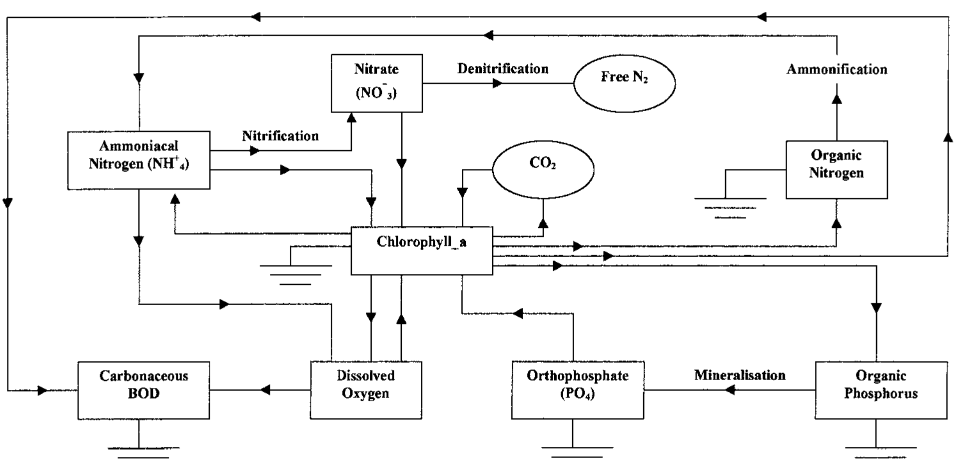

Figure 1. Schematic of system interactions.

These constituents get transported about an estuary through the mechanisms of advection and diffusion as outlined above and, at the same time, also undergo various biochemical reactions. These reactions are represented in the model by partial differential equations. The equations describe the rate at which each constituent changes over time due to various processes (Chapra 1997). All of these reactions and processes are not be presented here in detail; but some aspects of the chlorophyll *a* and nitrogen cycles are presented to give an insight into process modelling.

Chlorophyll *a* cycle

The primary influences on chlorophyll *a* production in the marine environment are nutrients, dissolved oxygen, light and temperature. The model represents these influences through the following partial differential equation describing the growth rate of chlorophyll *a*:

$$\frac{\partial C_P}{\partial t} = \left(G_{PI} - D_{PI} - \frac{V_{S4}}{H}\right) C_P \tag{4}$$

where
C_P = chlorophyll *a* concentration (mgl^{-1})
G_{PI} = specific growth rate constant (d^{-1})
D_{PI} = death plus respiration rate constant
V_{S4} = settling velocity (md^{-1})
H = depth (m)

The growth rate for phytoplankton in a natural environment is a complicated function of species present, solar radiation, temperature, transport processes and nutrient availability (Chapra 1997). In most models the population of phytoplankton is estimated by considering the total phytoplankton biomass. In practice the commonest way of measuring phytoplankton biomass is to measure chlorophyll *a*. The principle advantage of this approach is that the measurement is direct, it integrates all cell types and ages and accounts for cell viability. Thus chlorophyll *a* was modelled to represent phytoplankton.

It is known that the specific growth rate for chlorophyll *a* in equation (4) is related to temperature, light and nutrients; this relationship is expressed as

$$G_{PI} = \acute{G}_{PI} \bullet G_{RTS} \bullet G_{RNU} \bullet G_L \tag{5}$$

where
$\acute{G}_{PI}$ = maximum growth rate given optimum light and nutrients at 20°C
G_{RTS} = temperature correction when water temperature is other than 20°C
G_{RNU} = nutrient limitation factor.
G_L = light limitation factor.

The manner in which the above factors are incorporated within the model is presented below.

Temperature correction

G_{RTS} can be obtained from the Arrhenius equation

$$G_{RTS} = \theta^{(T-20)} \tag{6}$$

where

θ = temperature coefficient = 1.068

T = temperature of water

Nutrient limitation factor

The effect of nutrient concentrations on phytoplankton growth rates is complex but acceptable results can be obtained by representing phytoplankton production as a function

$$G_{RNU} = \mathrm{Min}\left[\frac{DN}{K_{MN} + DN}, \frac{DP}{K_{MP} + DP}\right] \tag{7}$$

of the relevant nutrients using the following Monod type growth kinetics (Chapra 1997).

where

DN = concentration of total inorganic nitrogen

DP = concentration of total inorganic phosphorus

K_{MN}, K_{MP} = half-saturation constants for DN and DP respectively and

Light limitation factor

Obviously light is required to generate green chlorophyll. The relationship between the chlorophyll *a* growth rate and light is complicated due to diurnal surface light variations and light attenuation with depth. Brown and Barnwell (1985) developed a relationship for the limiting effects of light on chlorophyll *a* growth. This relationship relates the limiting effect to surface light intensity, photoperiod, water depth and a light extinction coefficient. This general formulation was used in the model during this research.

Phytoplankton take-up of DN has a preference for NH_4^+ over NO_3^-, is incorporated into the model as outlined below in the next section.

The value for D_{PI} , death plus respiration rate, in equation (4) is calculated from:

where

$$D_{PI} = k_r + k_d + k_z \tag{8}$$

k_r = endogenous respiration rate

k_d = death rate due to parasitisation, infection and toxic materials

k_z = death rate due to grazing organisms.

The endogenous respiration rate of phytoplankton is the rate at which the phytoplankton oxidises their organic carbon to carbon dioxide per unit weight of organic carbon. Once again the value is temperature dependent and is described by the Arrhenius equation at any temperature T

$$k_r = k_r(20)\,\theta^{(T-20)} \tag{9}$$

where
$k_r(20)$ = endogenous respiration rate at 20°C
$\theta = 1.045$.

Finally, sedimentation is an important contributor to phytoplankton removal from the water column, particularly in lakes and coastal waters. The actual value of settling velocity, V_{S4}, can be calculated using Stokes Law. However, in practice phytoplankton tend to behave as a flocculent, Class 2 suspension, and this together with vertical turbulence, density gradient and whether or not the phytoplankton are flagellated greatly influence the settling velocity. A settling velocity of $0.2 m.d^{-1}$ was chosen from Bowie *et al.* (1978).

NITROGEN CYCLE MODELLING

The nitrogen cycle normally employed to represent the important interactions in modelling studies is shown schematically within Figure 1. This model considers the following five different forms of nitrogen: phytoplankton nitrogen, organic nitrogen (ON), ammoniacal nitrogen (NH_4^+), nitrate nitrogen (NO_3^-) and free nitrogen (N_2). The governing equations for these processes, and their interactions, are described by Brown and Barnwell (1985).

Phytoplankton Nitrogen

It is assumed that the amount of nitrogen in a given concentration of phytoplankton is given by ($C_p \times A_{NC}$), where A_{NC} is the fraction per unit mass of phytoplankton. Thus, from equation (4), the temporal rate of change in phytoplankton is given by:

$$\frac{\partial}{\partial t}(C_p A_{nc}) = \left(G_{PI} - D_{PI} - \frac{V_{S4}}{D} \right) C_p \bullet A_{nc} \tag{10}$$

Organic Nitrogen (C_{ON})

When phytoplankton decays the phytoplankton nitrogen will be recycled and hence contribute to the pools of organic nitrogen and ammoniacal nitrogen. The amount recycled to organic nitrogen is in proportion to a preference factor, f_{ON}. Some of the organic nitrogen will be converted to ammoniacal nitrogen and some will settle. Thus the net temporal rate of change of organic nitrogen is represented as:

$$\frac{\partial}{\partial t}(C_{ON}) = D_{PI} C_P A_{NC} f_{ON} - k_{71} \theta_{71}^{(T-20)} C_{ON} - \frac{V_{S3}}{D} C_{ON} \tag{11}$$

where
k_{71} = decay rate for hydrolysis of organic nitrogen at 20°C
$\theta_{71}^{(T-20)}$ = temperature correction for hydrolysis of organic nitrogen
V_{S3} = settling velocity of organic nitrogen ($m.d^{-1}$)

Ammoniacal Nitrogen (C_{NH3})

The source terms for NH_4^+ are from nitrification of organic nitrogen and conversion of phytoplankton nitrogen. The sink terms for NH_4^+ are due to phytoplankton take-up and denitrification to NO_3^-. The take-up of NH_4^+ by phytoplankton depends on the preference that the phytoplankton has for NH_4^+ over NO_3^-. This preference is represented in the model by the factor P_{HN4}^+. The net temporal rate of change of ammoniacal nitrogen is represented as: NH_4^+ NO_3^-

$$\frac{\partial}{\partial t}(C_{NH4}^+) = k_{71}\theta_{71}^{(T-20)} C_{ON} - G_{PI}P_{NH3}C_PA_{NC} - k_{12}\theta_{12}^{(T-20)} \left(\frac{C_{DO}}{k_{DO} + C_{DO}}\right) C_{NH4}^+ + D_{PI}C_PA_{NC}(1 - f_{ON}) \quad (12)$$

where

k_{12} = rate oxidation of NH_4^+ to NO_3^- at 20°C

$\theta_{71}^{(T-20)}$ = temperature correction for oxidation of NH_4^+ to NO_3^-

C_{DO} = concentration of dissolved oxygen

k_{DO} = half saturation concentration of dissolved oxygen

Nitrate Nitrogen (C_{NO3}^-)

The source term for NO_3^- is from denitrification of NH_4^+. The sink terms for NO_3^- are due to phytoplankton take-up and denitrification to NO_2^- and N_2. The preference for phytoplankton to take-up of NO_3^- is represented in the model by the factor ($1-P_{HN3}^-$). The net temporal rate of change of nitrate nitrogen is represented as:

$$\frac{\partial}{\partial t}(C_{NO3}^-) = k_{12}\theta_{12}^{(T-20)} \left(\frac{C_{DO}}{k_{DO} + C_{DO}}\right) C_{NH4}^- - G_{PI}(1 - P_{NH4}^-)C_PA_{NC} \quad (13)$$

Similar sets of partial differential equations are used to describe the oxygen and phosphorus cycles in the model. The partial differential equations describing the water quality parameters and their interactions are approximated by finite difference expressions and solved for using the computer program DIVAST. Details of the numerical aspects of the model can be found in Falconer and Liu (1999).

CHLOROPHYLL *A* PRODUCTION MODELLING OF CORK HARBOUR

Cork Harbour is located on the south-west coast of Ireland and is one of the most important sea inlets in Ireland. It is a busy seaport, a significant receiver of domestic and industrial waste and a popular recreational resource.

The study area extends from the lower exits of the River Lee in the northwest to the open sea below Roches Point in the south. The River Lee flows into Lough Mahon, which, in turn, enters the main harbour through Passage West and Passage East, located to the west and east of Cobh or Great Island. Passage West and Passage East are deep steep-sided channels running from north to south, with rock and shingle beaches exposed between tides. The main harbour is connected to the open sea through a deep channel to the south.

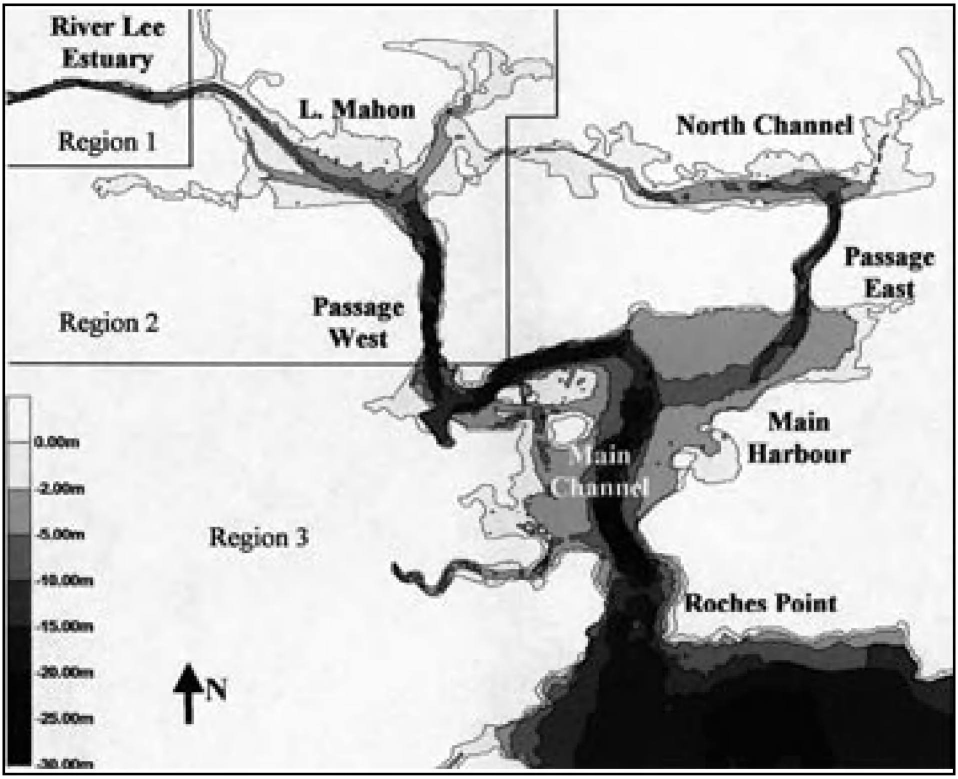

Figure 2. Plan of Cork Harbour model showing water depths.

The model study area measures approximately 354 km^2 (Figure 2). A finite difference grid composed of 565 × 697 (393,805) cells at a 30 m grid resolution was used in the model to represent the study area and provides a high spatial resolution. The finite difference grid was used to construct a detailed bathymetric model of the domain. All hydrodynamic and water quality parameters were resolved at each grid point.

The rise and fall of the tide in Cork Harbour is typical of many Irish coastal locations. The mean tidal range is 3.7m on spring tides and 2.0m on neap tides. Extensive areas of mudflats become exposed within the study area at low water, particularly in Lough Mahon and the North Channel. In Lough Mahon the plan area of the water at low water is *c.* 70% of the high water plan area. Similarly, in the North Channel, the plan area at low water is *c.* 56% of the high water plan area. These areas of mudflats give rise to odours because of decaying organic material deposited there at high water. The four main freshwater inflows to Cork Harbour (Figure 2), are the Rivers Lee, Glashaboy, Owenacurra and Owenabuidhe.

The generic model DIVAST was applied to Cork Harbour to simulate water circulation and water quality processes. The hydrodynamic module required detailed information on the tidal regimes of the study area. These data included tidal ranges and periods for the spring / neap tidal cycle.

The hydrodynamic module was calibrated, in the generally accepted manner, by 'tuning' empirical parameters until good measured current data set was attained. The

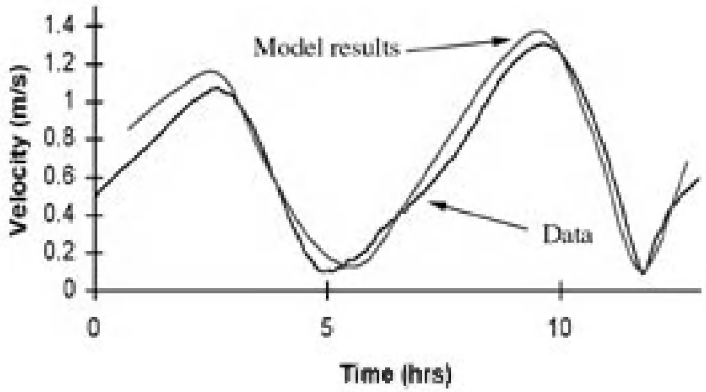

Figure 3. Hydrodynamic model validation.

main tuning parameters were time step, bed friction and coefficient of eddy viscosity. During calibration the model is run for boundary conditions (principally tide, wind and river inflows) that correspond to conditions that prevailed during survey conditions. The above parameters were tuned until good correlation was achieved between predicted and measured water elevations and current speeds and directions. Different data sets were then used for validation to ensure that the model was accurately predicting the hydrodynamic regime in the study area. Figure 3 shows one of the model validation comparisons between measured and predicted current speeds at a point near Roches Point. Validation was carried out at over 10 different sites with similar close agreements between predicted and observed current speeds (Costello *et al.* 2001).

The solute transport module was calibrated using salinity as a tracer; the EPA provided salinity data to the project for this purpose. The water within the model domain was specified as having completely freshwater initially and the water at the sea boundary to the south was specified as having a salinity of 35psu. The solute transport module was then run until steady state conditions were attained with respect to salinity. After steady state was reached strong salinity gradients were observed from south to north throughout the domain. The predicted salinity concentrations were compared against measured salinity data for both summer and winter conditions separately. Figure 4 shows a comparison for summer conditions between measured and predicted salinity at two points in the model, one near Roches Point and at the northern end of Passage West. In Figure 4 the curve represents the model predictions and the two horizontal lines represent minimum and maximum measured salinity. It is seen that the model predicts salinity values close to the minimum and maximum measured data. In all, ten stations were used to compare summer and winter model predictions with salinity data throughout Cork Harbour. Comparisons similar to the above showed that the model accurately predicts the spatial variations in salinity throughout the study area and temporal variations in salinity due to tidal dynamics.

Salinity modelling is important not only with regards to validating the solute transport module but also directly in assessing trophic status. In a recent report (EPA

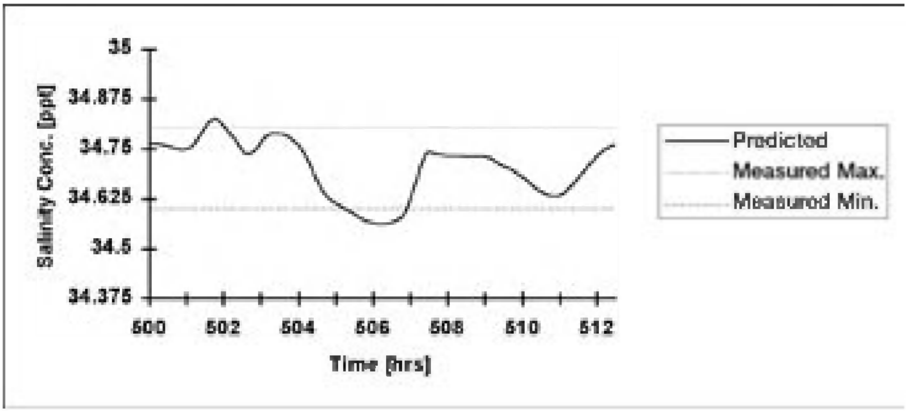

Figure 4. Salinity model validation.

2001) trophic assessment criteria are explicitly related to salinity. It is therefore extremely important that models developed can accurately predict salinity.

The chlorophyll *a* production model was then developed by specifying all the major relevant discharges into Cork Harbour. Concentrations of nutrients, oxygen, BOD and chlorophyll *a* associated with all riverine, domestic and industrial discharges were defined. Further, fluxes of these parameters across the sea boundary were estimated based on fieldwork data collection. The locations of the existing domestic and industrial discharges are shown in Figure 2. The daily loads of organic nitrogen (ON), total ammoniacal nitrogen (TAN), total oxidised nitrogen (TON), organic phosporous (OP) and orthophossphate (MRP) into Cork Harbour during typical summer conditions are presented in Table 1.

Estimates were also made of the influxes of nutrients into Cork Harbour across the sea boundary based on Costello *et al.* 2001 and are presented in Table 2.

Table 1. Daily discharge of nutrients (summer conditions)

Source	ON (kgd^{-1})	TAN (kgd^{-1})	TON (kgd^{-1})	OP (kgd^{-1})	MRP (kgd^{-1})
Domestic & industrial discharges	1218	16063	2217	508	3099
Riverine discharges	1575	97	3434	30	53
Total	2793	16160	5651	538	3152

Table 2. Nutrient loads during the flooding portion of a mean tidal cycle (summer conditions)

ON (kg/d)	TAN (kgd^{-1})	TON (kgd^{-1})	OP (kgd^{-1})	MRP (kgd^{-1})
84564	4374	17010	3159	1701

The loads presented in Table 2 represent the masses of nutrients transported into Cork Harbour from the open sea during the flood half of a mean tidal cycle, approximately 6 h. Much of this will get transported back across the sea boundary into the open sea on the ebb tide. Although it is often difficult to obtain good data for sea boundary water parameters, it is necessary to specify these data as accurately as possible in order to fully describe nutrient budgets of the domain.

Ignoring nutrient fluxes across the sea boundary, Table 1 shows the relative contributions to nutrient loadings from the two main sources. In particular, it is seen from this table that the domestic and industrial discharges contribute substantially more to the total phosphorus load than for the riverine discharges. This is very significant for water quality since the Irish EPA has concluded that phosphorus is the limiting nutrient in the upper reaches of Cork Harbour (EPA 2001).

RESULTS

The DIVAST model was used to simulate chlorophyll *a* dynamics throughout Cork Harbour at each model grid point every 40 seconds throughout model simulations. Two discharge scenarios were modelled to investigate the system response of chlorophyll *a* production to nutrient inputs:

Scenario I – all discharges shown in Table 1

Scenario I – all domestic and industrial discharges in Table 1 set to zero

The main emphasis during the analysis of the model results is on the Lough Mahon region since it is known to suffer from adverse water quality and, in particular, from the occurrence of algal blooms. In order to consider the impacts of switching off the domestic and industrial discharges it was decided to compartmentalise Cork Harbour into 3 regions as shown in Figure 2. These regions are:

Region 1 River Lee Estuary

Region 2 River Lee Estuary, Lough Mahon and Passage West

Region 3 Entire model domain

There were a number of reasons for subdividing the study area in this manner. Firstly, most of the nutrient inputs are discharged along the estuary of the River Lee; thus it was considered necessary to analyse this area of the model in detail. Secondly, it is known that there is relatively little water exchange between Lough Mahon and the Main Cork Harbour area and so it is instructive to look at Region 2 in isolation from the greater Cork Harbour. Thirdly, the EPA has defined Region 2 as a distinct water region for water quality monitoring and analysis. In order to compare model predictions with EPA measurements it was necessary to consider similar water regions. Finally, it was considered best to assess the differences between Scenarios I and II over a regional basis, rather than at individual locations, to obtain a more complete understanding of the marine response to the two loading conditions.

The model was used to determine average concentrations of nutrients and chlorophyll *a* over each of the three regions at each computational timestep. This was achieved using the following expression

$$C = \frac{\sum_{i=1}^{N} C_i V_i}{\sum_{i=1}^{N} V_i} \tag{14}$$

where

C = average concentration of parameter over the region

C_i = concentration of parameter in the i^{th} cell of the model

V_i = the volume of the i^{th} cell of the model

N = is the total number of cells in the region

The results from Scenario I were firstly compared with measurements of water quality parameters produced by the Irish EPA for the Lough Mahon region. The EPA had a number of measurement stations in the Lough Mahon region where they sampled water for levels of % dissolved oxygen saturation (DO), chlorophyll *a* (CHL), orthophosphate (MRP) and dissolved inorganic nitrogen (DIN). From these data the EPA deduced average concentration levels for the four water quality parameters throughout Region 2 for summer and winter conditions. Results from Scenario I modelling were averaged and compared with the EPA values for the more critical summer period. Table 3 presents this comparison for the four water quality parameters chosen by Irish EPA.

Model MRP values appear to be under-predicted, but there is very close agreement between the EPA and model values for chlorophyll *a* and the other parameters. There are a number of reasons for the discrepancies between the EPA data and model results. The locations of the EPA data collection stations in Region 2 were biased towards the estuary of the River Lee close to the major sources of phosphorus loading. The averaging technique used to obtain both sets of results in Table 3 were different. The EPA data were simply averaged arithmetically over all the six sampling stations, whereas the model averages were computed over the entire area of Region 2 using equation (14). However, in general, the above comparison provides good confidence in the ability of the model to predict chlorophyll *a* production in the region.

This good agreement may be attributed to:

- Good calibration and validation of the hydrodynamic model
- Good calibration and validation of the solute transport model
- Access to extensive data sets of nutrient inputs into the harbour

Figures 5–7 show comparisons of predicted chlorophyll *a* concentrations between the model results for Simulation I and II for the 3 regions. During a mean tidal cycle, amplitude 1.5m, the volume of water contained within Cork Harbour at high water is approximately $8.12 \times 10^8 m^3$ and at low water the volume is approximately $5.69 \times 10^8 m^3$, giving a per tidal exchange volume of approximately $2.43 \times 10^8 m^3$. Thus the model results as presented exhibit diurnal variations due to tidal dilutions.

Figure 5 shows that throughout the Lee Estuary there is a significant reduction in the production of chlorophyll *a* when the outfalls are switched off. Scenario I predicts maximum chlorophyll *a* concentration levels in the order of $18 mg.m^{-3}$ and still increasing after 500 hours of simulation, whereas Scenario II predicts maximum chlorophyll *a*

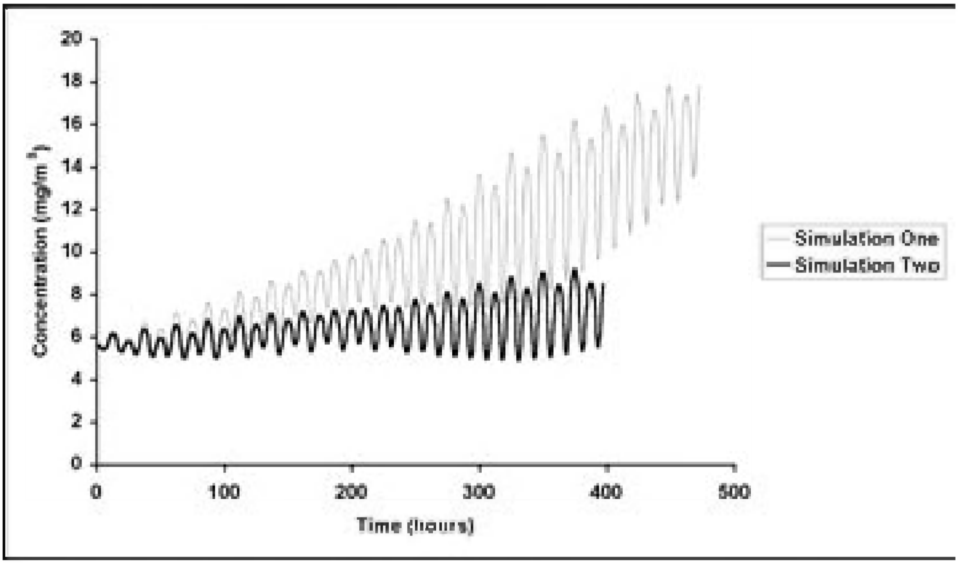

Figure 5. Chlorophyll *a* comparisons–Region 1.

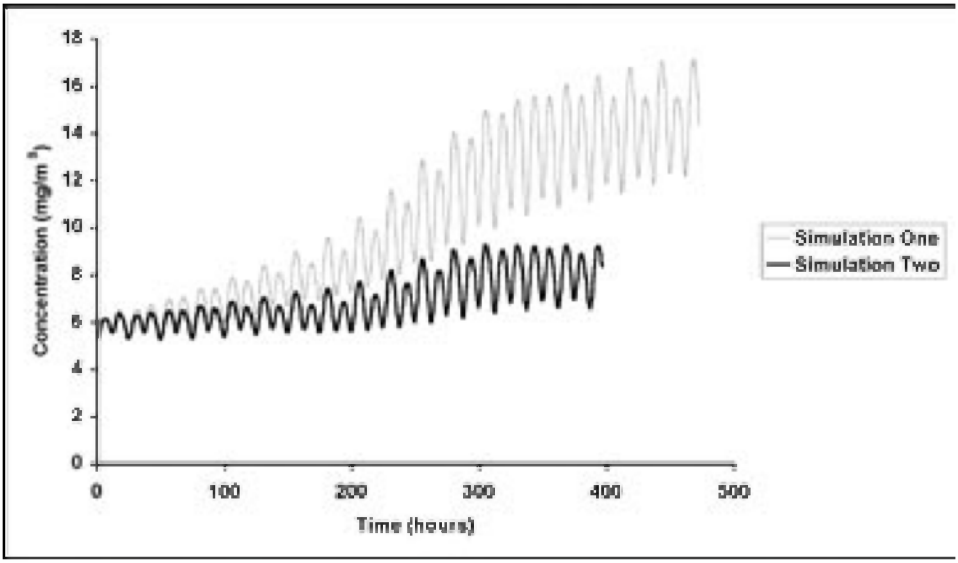

Figure 6. Chlorophyll *a* comparisons–Region 2.

Table 3 Comparison between model and EPA results (see text for abbreviations)

Parameter	EPA Data	Model
DO (% saturation)	80.6	84.3
CHL (mg m^{-3})	12.9	12.8
MRP (mg m^{-3})	76.3	56.2
DIN (mg l^{-1})	1.2	1.4

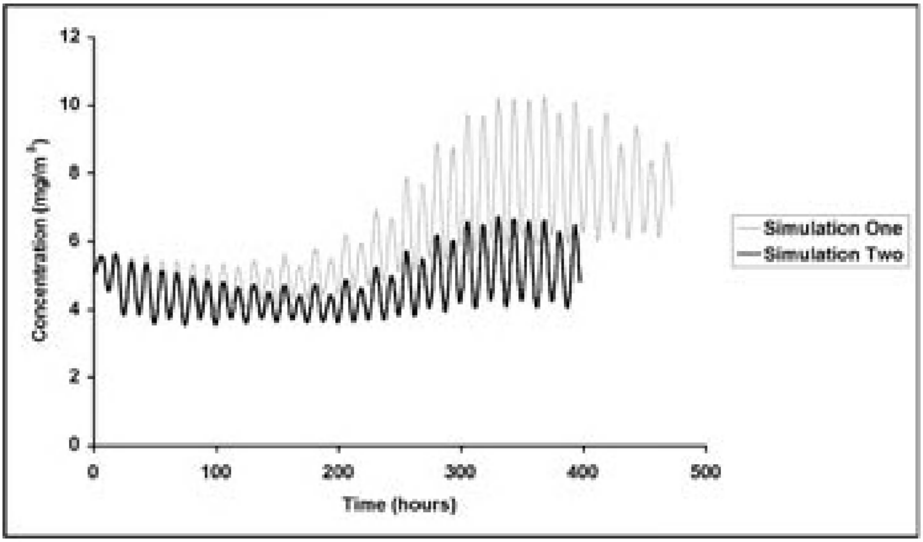

Figure 7. Chlorophyll *a* comparisons–Region 3.

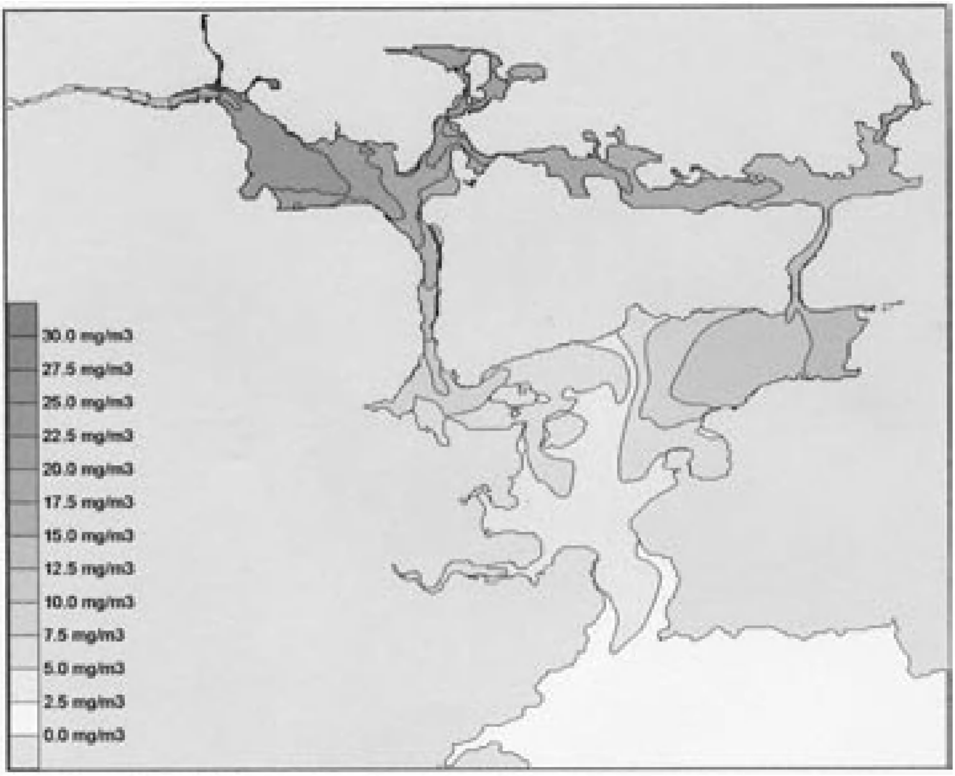

Figure 8. Chlorophyll *a* concentrations at low water, Scenario 1.

concentration levels in the order of 8mg m^{-3} and steady state conditions appear to have been reached. Figure 6 illustrates that a similar situation prevails in Region 2, but the chlorophyll *a* levels appear to have reached steady state during Scenario I also. Again, in Region 3 steady state appears to have been reached for both simulations with a noticeable decrease in concentrations during Scenario II. Although Scenario I results have not yet reached steady state, 500h simulation times for both scenarios is adequate to indicate the significant differences between the two model simulations.

Figure 8 shows a snapshot of chlorophyll *a* concentrations at low water throughout the entire domain for Scenario I. This figure illustrates that much higher chlorophyll *a* concentration levels are predicted in the Lee Estuary and Lough Mahon regions than the rest of Cork Harbour. This is because the major nutrient discharges from the outfalls are in this region and because there is limited water exchange between Lough Mahon and the greater Cork Harbour area due to the restriction of Passage West. This mechanism has previously been studied and is well documented during various water quality investigations (Costello *et al.* 2001).

CONCLUSIONS AND DISCUSSION

The model of Cork Harbour detailed above is highly resolved both spatially and temporally and well defines the main nutrient discharges into Cork Harbour. The collation of the physical and discharge information to develop and run the model was undertake through an EPA funded research project entitled 'Investigation of nutrient inputs, fluxes and productivity in selected brackish water bodies' and was quite time consuming. The project integrated conventional data collection, remote sensing and modelling to provide a detailed analysis of the nutrient budget in the River Lee Estuary and Cork Harbour. The model developed resolves the main components of chlorophyll *a* production and decay (Figure 1) and hence is quite complex to solve. Because of the large number of model cells and small timestep, the model simulation time is only twice as fast as real time and hence computationally very expensive. Currently there are developments at National University of Ireland, Galway to parallelise the computer code to allow it to run on high performance computers with parallel processors and allow web access to perform simulations. Preliminary developments improve computational efficiency by a factor of three.

The above model results show that by eliminating domestic and industrial discharges a significant reduction in chlorophyll *a* levels could be expected throughout Cork Harbour. When compared with other nutrient sources, particularly riverine sources, domestic and industrial discharges provide significant percentages of the overall MRP budget. Domestic and industrial inputs of MRP represent approximately 98% of the freshwater inputs (Table 1) and approximately 64% of all MRP inputs (Table 1, Table 2). The predicted reductions in chlorophyll *a* by eliminating these sources concurs with EPA (2001) that chlorophyll *a* production in Lough Mahon is phosphorus limited.

While complete elimination of domestic and industrial discharges into Cork Harbour is not realistic, the model results illustrate how scenario modelling could be used to comply with water quality standards. Such scenario modelling could also be used to assess the likely effects on water quality of reduced nutrient runoff from agricultural activities.

A GIS based tool, MarGIS has been developed by the modelling company MarCon to store all relevant environmental data pertaining to water bodies and to interface with the DIVAST model of specific water bodies. DIVAST can be run easily through MarGIS and displayed in the ArcGIS environment. Through MarGIS all data and model results are integrated in one management tool. Often monitoring programmes for water bodies are devised before models have been developed and it is recommended that water quality models be developed at the planning stages for two fundamental reasons. Firstly, model results from tools like MarGIS will provide useful information on the strategic deployment of instruments programmes so that the data from these programmes can then be applied with more confidence in assessing the trophic status of water bodies. Secondly, the results of the monitoring programmes can subsequently be used for further model validation. A final recommendation is that, in general, more long-term data acquisition is required to aid model calibration and validation. The close agreement between model predictions and data obtained in this work was possible only because good data sets were available for model boundary conditions. In this recommendation, the author agrees with one of the primary conclusions of Jones *et al.* (2002) on their appraisal of the role of decision support systems in estuarine management.

It is possible that the current EPA sampling stations in the estuary of the River Lee overestimate average MRP concentrations and underestimate average DO. Thus the EPA water quality figures are probably quite conservative.

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sedimentary shores were later and fewer than those on rocky shores and lacked an overall synthesis of ecosystem functioning.

Work on interactions between organisms on the rocky shore has in fact provided two major ecological paradigms. The first, chronologically speaking was Connell's (1961) work on the barnacles *Balanus* (now *Semibalanus*) *balanoides* and *Chthamalus stellatus* while the second was Paine's (1974) experiments with the mussels *Mytilus californianus* and the starfish *Pisaster ochraceus*. The first (Connell 1961) demonstrated that the upper limit of distribution on the shore was set by physical tolerance limits, while the lower limit was a function of biological factors, in this case competition. The second (Paine 1974) advanced the concept of 'keystone' species, in which the structuring of the community, both physical and biological, is determined by the control (usually top-down) of species which might be numerically insignificant, but whose interactions (in this case predation of the mussels by the starfish) determine the current status. Other work has yielded valuable insights into the functioning of the wider marine system, ranging from the mechanisms of habitat selection by larvae (e.g. Crisp, 1961) to the pressures of and responses to evolutionary driving forces (e.g. McMahon, 2001), and a useful review of community function is given in this volume by Crowe.

In contrast, work on sedimentary shores has tended to be dominated by the habitat itself, in that many of the factors which apply directly on the rocky shore are buffered or mediated by the sediments. One aspect of the buffering is that sedimentary environments display a spatial homogeneity (Wilson 1977, Thrush *et al.* 1989) and this in turn has made it a lot easier to put together budgets and derive metrics for the system (e.g. Baird and Milne 1981).

There have been many studies since Baird and Milne's (1981) first energy budget, but the majority have tended to focus on estuaries and other, mainly sedimentary, locations (e.g. Brown and McLachlan 1990, Wilson 2002). These investigations have been extended into consideration of the systems' reaction to anthropogenic and other pressures such as resource exploitation or pollution (Baird *et al.* 1991, Wilson and Parkes 1999). However, the number of systems for which sufficient information exists for ecological modelling is still extremely limited (e.g. Soetart and Herman 1995). Energy budgets have also been used to investigate system properties through network analysis (Baird and Ulanowicz 1993, Wilson and Parkes 1999), from which metrics can be derived quantifying properties such as system throughput, capacity and stability (Wulff *et al.* 1989).

McArthur (1955) was one of the first to articulate the view that system stability was linked to diversity through the provision of parallel, often redundant, functions. Sanders (1968) 'stability-time' hypothesis developed this and thus would predict low stability for littoral systems. This view has also been supported by other studies, for example those of Roth and Wilson (1998) who calculated system metrics signifying low stability in the littoral communities of Dublin Bay. Nevertheless, palaeontological records and archaeological records show some constancy in the general species' associations at particular coastal locations (e.g. Wilson 1993b), while workers such as May (1973), McNaughton (1977) and Tilman (1996) have teased out the diversity/stability relationship and emphasised the distinction between the responses at population and at community level.