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An Integrated Approach to Trophic Assessment of Coastal Waters Incorporating Measurement, Modelling and Water Quality Classification

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Abstract
Various protocols have been developed for trophic assessment of coastal waters but most incorporate single value compliance criteria for water quality parameters which, in reality, exhibit significant spatial variation. Application of these protocols generally requires the averaging of data recorded at discrete locations to obtain single values for comparison with compliance criteria; however, there is no guarantee that the averaged values are truly representative of overall water quality. The present approach to trophic assessment integrates measurement, remote sensing, flushing analysis and water quality modelling with the aim of identifying sub-regions in coastal waterbodies that demonstrate homogeneity. Averaged water quality data for these homogenous sub-regions should be more representative of the water quality in the sub-region as a whole making the assessment process more accurate. The integrated approach incorporates water quality classification criteria developed by the Irish Environmental Protection Agency but it can incorporate any suitable classification criteria. The approach was validated for a sub-region of Cork Harbour, Ireland and the trophic assessments agreed with previous assessments by the Irish Environmental Protection Agency (EPA). The approach has significant applications in the implementation of the EU Water Framework Directive and was adopted by the Irish EPA in convincing the EU to provide the Irish government with structural funds for water quality management within Cork Harbour.

Keywords: integrated trophic assessment; modelling; monitoring; water quality; classification; remote sensing

1 Introduction
The OSPAR (Oslo and Paris) Commission, the body responsible for guiding international cooperation on the protection of the marine environment of the North-East Atlantic, have defined eutrophication as ‘the enrichment of water by nutrients causing accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms and to the water quality concerned…’ (OSPAR, 2003). In general, eutrophication can have a number of deleterious effects on waterbodies. These can include an increase in the number and duration of algal blooms, among them toxic and nuisance species, depletion of dissolved oxygen and formation of toxic hydrogen sulphide during decay of such blooms, and alteration of the natural composition of the waterbody ecosystem. In estuaries and coastal waters, nutrient enrichment and ensuing eutrophic conditions may occur naturally;
however, in recent decades, anthropogenic nutrient sources such as agriculture, industry and domestic sewage have increased nutrient inputs to estuaries to many times their natural levels, accelerating eutrophication.

Despite increased public concern in recent times and the resulting increase in research effort and expenditure on remedial and management techniques, eutrophication is recognised worldwide as the greatest estuarine environmental problem (Ferreira et al., 2011; Xiao et al., 2007). In Europe, measures such as the Nitrates Directive (91/676/EEC), the Urban Wastewater Treatment Directive (91/271/EEC), the Water Framework Directive (2000/60/EC) and the OSPAR Joint Assessment and Monitoring Programme (JAMP) have been introduced in an attempt to tackle eutrophication (and water quality in general), primarily through the management of nutrient inputs. While these measures have helped, OSPAR (2008) reported that eutrophication is still a problem in 106 defined areas of the North-East Atlantic including the Greater North Sea, some small coastal embayments and estuaries within the Celtic Seas, the Bay of Biscay and the Iberian Coast.

An important step in the solution of the eutrophication problem is quantification of eutrophication. Unfortunately, eutrophication in estuaries is a complex issue involving many different physical, chemical and biological processes and accurate quantification is quite difficult. Historically, the classical freshwater approach (e.g. Carlson, 1977) has been used. This approach is based on the measurement of variables such as nutrients and chlorophyll_a (a measure of algal biomass) and the establishment of nutrient-based classification systems. However, more recently it has become apparent that nutrient-based systems alone are insufficient. While nutrients are the primary cause of eutrophication, there are many other factors that can influence trophic status such as tidal exchange and freshwater inflows. As a result there has been a sustained focus on the development of more integrated approaches to quantification of eutrophication in estuarine waters such as the OSPAR Common Procedure (OSPAR, 1997) or the ASSETS (Assessment of Estuarine Trophic Status) methodology (Bricker et al., 2003) developed jointly by NOAA (US National Oceanic and Atmospheric Administration) and the Portuguese Institute of Marine Research (IMAR).

This paper presents an integrated approach to the assessment of trophic status which combines both measurement, remote sensing, modelling and water quality classification. One of the difficulties in the implementation of trophic assessment protocols is that single value thresholds are specified for spatially and temporally varying water quality parameters. The use of such protocols requires averaging of measured data in both space and time; however, the averaged data may not be truly indicative of the quality of the water in a region. In the present approach a combined hydrodynamic and water quality model is used to identify homogenous sub-regions within a waterbody based on a flushing study for which sub-region averaged values will be representative of the whole sub-region. To validate the integrated
approach, it was applied to Cork Harbour, a brackish waterbody on the south coast of Ireland, and the assessment results were compared with those from previous Irish EPA assessments; for consistency the EPA's data-averaging techniques and classification scheme were used in the integrated approach. The classification scheme and the integrated approach developed by the Authors are described in the following sections. The methodology for the sub-division of the harbour is also presented as are results of the flushing study on which the sub-division was based. Finally, results are presented which demonstrate the accuracy of the model and the effectiveness of the new integrated approach.

2 Trophic Assessment in Ireland
Prior to the mid 1990s, water quality surveys of most estuarine and coastal areas in Ireland were infrequent and were generally intended to establish the effects of discharges from municipal or industrial sources (Toner et al., 2005). The current monitoring programme was initiated at a pilot scale by the Irish EPA (Environmental Protection Agency) in 1992/93. By 1995, the programme had been expanded to include the majority of Ireland’s more important estuaries, bays and inshore coastal waters and today it includes at total of 21 estuarine and coastal areas (OSPAR, 2008). A system for trophic assessment was first implemented in 2001; this has since been developed as the national Trophic Status Assessment System (TSAS) and is based on the OSPAR Common Procedure framework.

2.1 National Trophic Status Assessment System (TSAS)
TSAS comprises two main elements. Trophic status is primarily determined by a classification scheme based on quantitative analysis of a range of water quality parameters. The second element is based on a qualitative assessment of the presence of macroalgae. The second element has only been applied informally to date and a more comprehensive approach based on quantitative criteria is currently being developed. The integrated approach to trophic assessment developed during this research only incorporates the water quality element of the TSAS system, therefore, the remainder of this section focuses solely on the water quality classification scheme. In practice the integrated approach should be applied in combination with a quantitative assessment of macroalgae occurrences.

The detailed structure of TSAS is presented in Table 1. It can be seen that the scheme sets out compliance criteria for a range of water quality parameters: dissolved inorganic nitrogen (DIN), orthophosphate (MRP), chlorophyll_a and dissolved oxygen (DO). These parameters were specifically chosen to capture the cause-effect relationship of the eutrophication process. The parameter criteria listed in the table were determined from monitoring of estuarine and coastal waters and are considered to be indicative of good environmental quality. Importantly the parameter criteria must be scaled to take into account the influence of salinity, an important factor in determining the water quality characteristics of estuarine waters (Toner et al., 2005); the values shown for intermediate waters are for a median salinity of 17.
Following comparative analysis of measured data with the compliance criteria, trophic status is determined on the basis of the following classification scheme:

- **Eutrophic**: compliance criteria for all three categories are breached
- **Potentially Eutrophic**: criteria for two categories are breached and the third falls within 15% of the relevant threshold value(s)
- **Intermediate**: waterbodies are not Eutrophic or Potentially Eutrophic but one or two category criteria are breached
- **Unpolluted**: no criteria are breached

A major problem with the implementation of TSAS, and by extension the vast majority of trophic assessment schemes, is the single value nature of the water quality compliance criteria. Estuarine and coastal waterbodies are highly dynamic systems where environmental quality can vary quite significantly in space and time. Some averaging process must be used to obtain single values for water quality parameters in such regions; a certain degree of uncertainty will inevitably surround any trophic status designations based on region averaged parameter values. The vast majority of trophic assessment schemes incorporate a water quality classification system similar to that described and, while it is true that most assessment schemes are not based solely on such systems, trophic status classification is heavily influenced by these systems. The typical approach to the implementation of water quality compliance criteria, and that adopted by EPA, is to divide the waterbody into smaller sub-regions. Water quality data from monitoring stations within a sub-region are then averaged over time and across stations to obtain a single value for each parameter of interest. The problem with this approach is that, even within a sub-region, water quality can still be quite variable and there is no guarantee that the averaged values are a true representation of the water quality in the sub-region as a whole.

### 2.2 Trophic Status in Cork Harbour

Cork Harbour, shown in Figure 1, was the study area used to develop and test the integrated approach to trophic assessment. It is the deepest, natural harbour in Ireland and one of the largest and most important sea inlets in the country. The main freshwater influence is the River Lee; other freshwater discharges include the Rivers Owenacurra, Owenbuidhe and Glashaboy but their influences are relatively minor. The harbour is macro-tidal with a maximum tidal range of 4.2m and extensive mudflats are exposed at low tides, particularly in Lough Mahon and the North Channel. In general, the harbour waters are quite shallow. Deeper waters are confined to the Main Channel where depths of up to 30m exist; however, water depths within the vast majority of the harbour are less than 5m on a spring tide. From Figure 1 it can be seen that the harbour geometry is quite complex, as a result hydrodynamic circulation patterns in the harbour are also quite complex.
Over the years the harbour has become heavily populated and industrialised. Today, in excess of 100 large-scale industries discharge their effluents to the harbour; including a number of large pharmaceutical plants. Many smaller industries also discharge to the sewers. Nine wastewater treatment plants (WWTP) discharge domestic waste to the harbour and nutrients also enter through freshwater and marine sources. As a result, water quality has suffered. In the trophic status assessment for the period 1995-1999, the Lee estuary, Lough Mahon, and the Owenacurra estuary were all classified as eutrophic while the North Channel was designated potentially eutrophic. For the period 1999-2003, the Lee estuary and Lough Mahon were again classified as eutrophic while the Owenacurra estuary was also downgraded to eutrophic status.

In recent times, however, the situation has improved. In 2004, work was completed on Carrigrennan WWTP at Little Island in Lough Mahon (see Figure 1). The plant provides secondary treatment for sewage from Cork city which previously entered the Lee estuary and Lough Mahon untreated. As a result, a marked improvement in the water quality of the Lee estuary and Lough Mahon was recorded in the most recent assessment for the period 2002-2006 with both waterbodies upgraded to intermediate status. The recorded change in the trophic status of Lough Mahon, as a result of the new WWTP, was considered a suitable test of the validity of the Authors’ integrated approach to trophic assessment.

3 The Numerical Model

The numerical model used for the research was DIVAST (Depth-Integrated Velocity and Solute Transport); a two-dimensional, depth-integrated, time-variant model based on a finite difference solution scheme. The model is capable of simulating hydrodynamics, solute transport and water quality in relatively shallow estuarine and coastal waterbodies (e.g. Falconer, 1984; Lin and Falconer, 1997, Hartnett and Nash, 2004, Gao et al. 2011). The model simulates ten different water quality parameters: temperature, salinity, BOD, organic, ammonia and nitrate nitrogen, dissolved oxygen, phytoplankton, organic phosphorus and orthophosphate.

3.1 Hydrodynamics

The hydrodynamic module is based on the solution of the depth-integrated, Reynolds averaged, Navier-Stokes equations in two mutually perpendicular horizontal directions (x and y). The depth averaged continuity and x-direction momentum equation solved by the model are expressed as follows:

\[ \frac{\partial C}{\partial t} + \frac{\partial q_x}{\partial x} + \frac{\partial q_y}{\partial y} = 0 \]  

(1)
x-direction momentum equation:

\[ \frac{\partial q_x}{\partial t} + \beta \left[ \frac{\partial U q_x}{\partial x} + \frac{\partial U q_y}{\partial y} \right] = f q_y - g H \frac{\partial \zeta}{\partial x} + \frac{\tau_{xw}}{\rho} - \frac{\tau_{xb}}{\rho} \]

(1) \hspace{1cm} (2) \hspace{1cm} (3) \hspace{1cm} (4) \hspace{1cm} (5) \hspace{1cm} (6)

\[ + 2 \frac{\partial}{\partial x} \left[ \frac{\partial H}{\partial x} \frac{\partial U}{\partial x} \right] + \frac{\partial}{\partial y} \left[ \frac{\partial H}{\partial y} \frac{\partial U}{\partial y} + \frac{\partial V}{\partial x} \right] \]

(7)

where \( t \) = time; \( \zeta \) = water surface elevation; \( q_x, q_y \) = depth averaged volumetric flux components in the \( x,y \) directions; \( U, V \) = depth averaged velocity components in the \( x,y \) directions; \( \beta \) = momentum correction factor; \( f \) = Coriolis parameter; \( g \) = gravitational acceleration; \( \rho \) = fluid density; \( \tau_{xw}, \tau_{yw} \) = surface wind shear stress components in the \( x,y \) directions; \( \tau_{xb}, \tau_{yb} \) = bed shear stress components in the \( x,y \) directions and \( \epsilon \) = depth averaged eddy viscosity. The momentum correction factor, the wind and bed shear stresses, and the depth averaged viscosity are described in detail in Falconer (1993). Using the numbering convention of equation (2), the model incorporates (1) local and (2) advective accelerations, (3) Coriolis forces, (4) barotropic and free surface pressure gradients, (5) wind action, (6) bed resistance and (7) a simple mixing length turbulence model.

3.2 Solute Transport

The solute transport module is based on the general depth-integrated advection-diffusion equation which takes the following form:

\[ \frac{\partial S}{\partial t} \left[ \frac{\partial S}{\partial x} + \frac{\partial S}{\partial y} \right] = \frac{\partial}{\partial x} \left[ \frac{\partial H D_{xx}}{\partial x} \frac{\partial S}{\partial x} + \frac{\partial H D_{xy}}{\partial y} \frac{\partial S}{\partial y} \right] + \frac{\partial}{\partial y} \left[ \frac{\partial H D_{yx}}{\partial x} \frac{\partial S}{\partial x} + \frac{\partial H D_{yy}}{\partial y} \frac{\partial S}{\partial y} \right] \]

(1) \hspace{1cm} (2) \hspace{1cm} (3)

where \( D_{xx}, D_{xy}, D_{yx}, D_{yy} \) are the depth averaged dispersion coefficients in the \( x \)- and \( y \)-directions respectively. The numbered terms of equation (3) refer to depth averaged solute variations within the model domain as a result of: (1) local effects, (2) advective effects and (3) turbulent diffusion and dispersion.

3.3 Water Quality

The structure of phytoplankton models can vary quite widely from relatively simple nutrient-phytoplankton-zooplankton (NPZ) models (e.g. Edelvang (2005), Franks and Chen (2000), Nikolaidis et al. (2005)) to much more detailed ecological models (e.g. Spillman et al. (2008),
Nobre et al. (2010)). While additional detail might provide a more realistic model it does not necessarily provide a more accurate model (see Franks (2002)). The main advantages of the simpler NPZ model structure are the limited number of state variables and relatively low number of parameters. NPZ models are therefore easier to initialise and test than other more complex models.

The phytoplankton model chosen for this research (see Figure 2) was of the NPZ variety; however, in the absence of any zooplankton data, zooplankton effects were simply incorporated through the inclusion of a constant grazing rate in the phytoplankton growth equation. The model incorporates the nitrogen and phosphorus nutrient cycles and the dissolved oxygen cycle and includes such processes as nitrification, ammonification, mineralisation and reaeration. The kinetic formulations used to simulate the system dynamics are based on those of the US Environmental Protection Agency’s widely-used QUAL2E model (see Brown and Barnwell (1987) for details). As is common in most primary production models (Huot et al. 2007) chlorophyll_a is used as a measure of phytoplankton biomass. The model therefore computes all four parameters used in the TSAS scheme either directly (orthophosphate, chlorophyll_a and dissolved oxygen) or indirectly (DIN = ammonia + nitrate). While some NPZ models compute phytoplankton concentrations at a species-based level, application of the quantitative element of TSAS simply requires values for total phytoplankton biomass. The added complexity of a species-based phytoplankton model was therefore deemed unnecessary.

Phytoplankton production is influenced by water temperature and the availability of nutrients and light. The rate of change of chlorophyll_a concentration (C_P) with time is represented mathematically in the model as follows:

\[
\frac{\partial C_P}{\partial t} = \hat{G} \cdot G_T \cdot G_N \cdot G_L \cdot C_P
\]  

(4)

where \(\hat{G}\) is the maximum growth rate (at 20°C) measured in d\(^{-1}\), \(G_T\) is the temperature correction factor, \(G_N\) is the nutrient limitation factor and \(G_L\) is the light limitation factor.

As a limiting factor, light availability is particularly important for phytoplankton growth. Light attenuation occurs due to absorption and scatter by water particles and by other particles suspended in the water column. Light attenuation is incorporated in the light limitation factor as follows:

\[
G_L = \left[ \log \left( \frac{I_H + I_Q}{I_H + I_Q^{1/K_A H}} \right) \right] \cdot \frac{f}{K_A H}
\]  

(5)
where $I_i$ is the light level at which phytoplankton growth is half of $I_G$, $I_o$ is the surface light intensity, $f$ is the photoperiod (sunlight fraction of day), and $K_A$ is the light attenuation coefficient. Attenuation due to suspended solids can be particularly important in tidal embayments due to elevated levels of suspended sediment. For this reason, suspended solids and sediments are sometimes included as state variables in NPZ models. In the present research there was insufficient data for either variable to be included directly in the model; however, their attenuating effects were included indirectly by developing a site-specific formula for light attenuation based on recorded secchi disc transparencies. The site-specific formula also incorporates the self-shading effect of phytoplankton biomass and can be written as follows:

$$K_A = 0.79012 + 0.00429dC_P$$ (6)

This site-specific approach to light attenuation was developed by the Authors for a water quality modelling study of a second Irish estuary, Wexford Harbour (Hartnett and Nash, 2004) and was found to significantly improve the accuracy of the phytoplankton model. Similar improvements were found to occur when applied in the present research.

### 4 Methodology

The integrated approach to trophic assessment uses numerical modelling in combination with data collection and monitoring to help address the problem of uncertainty in relation to the application of single-value water quality compliance criteria. Incorporating the numerical model in the assessment strategy allows the identification of homogenous sub-regions within a waterbody. This, in turn, allows the calculation of sub-region average concentrations of water quality constituents that are more representative of the sub-region as a whole. Figure 3 shows a graphical representation of the integrated assessment approach. Firstly, the numerical model is developed for the study area, then calibrated and validated. Secondly, the model is used to determine residence times by means of a flushing study and the waterbody is sub-divided on the basis of residence time homogeneity. Following the identification of homogenous sub-regions, the water quality modelling is performed and sub-region average values for compliance parameters are calculated for both modelled and measured data. Finally, the compliance criteria, in this case TSAS, is applied to the measured and modelled data and trophic statuses of sub-regions are classified accordingly.

#### 4.1 Model Development and Data Collection

The model was developed in a sequential manner: first the hydrodynamic module, second the solute transport module and third the water quality module. Each module was calibrated and validated before the next module was developed; different measured datasets were used for
calibration and validation. Due to the complexity of the harbour and the scale of the processes of interest a high spatial resolution of 30m was used. The model domain covered the full extents of the study area shown in Figure 1. Data requirements for the hydrodynamic module included water depths, tidal forcings and freshwater inflows, as well as calibration and validation data in the form of flow and water level measurements. Dispersion and diffusion coefficients were required for the solute transport module; these were obtained by calibration with dye surveys (see Nash and Hartnett (2010) for more details).

By far, the greatest data requirements were those of the water quality module, these included: nutrient fluxes, initial water quality conditions, ambient environmental conditions, kinetic transformation rates and constants, and water quality data for calibration and validation purposes. Information on nutrient inputs was particularly important. Fluxes for all ten water quality parameters simulated by the model were calculated for all river-boundaries and the sea-boundary; these were specified in the form of monthly-average values. Regarding industrial and domestic inputs, monthly-average values were also specified for rates of discharge and concentrations of constituents. Data was available for 27 companies that discharge directly into the harbour; there was no data, or data was not felt essential for a further 97 companies. Inorganic industrial discharges not considered to affect the estuarine nutrient dynamics were excluded. In cases where industries discharged through the same outfall, inputs were added together. Many smaller industries discharge to the public sewer system and were thus included within the sewage discharges. Treated domestic waste is discharged to the harbour through nine wastewater treatment plants. The composition of the industrial and domestic discharges and associated flow rates specified to the model are listed in Table 2; Figure 4 shows their respective locations. For more complete details of data collection and model development readers are referred to Nash and Hartnett (2010).

4.2 Sub-division of the Waterbody

Water quality in coastal embayments can vary quite significantly in both space and time, particularly in embayments with complex topographies. Such spatial and temporal variations in water quality are strongly influenced by local hydrodynamic and solute transport processes. The aim of the sub-division of the waterbody is the identification and delineation of smaller bodies of water within a large coastal embayment where the hydrodynamic and solute transport regimes suggest some homogeneity in water quality. Hydrodynamic circulation patterns in coastal embayments can be quite complicated and their effects on solute transport difficult to interpret. Flushing characteristics, such as residence time, can be very helpful in better understanding water circulation and, by extension, nutrient dynamics within estuarine systems. In previous modelling studies (e.g. Nash and Hartnett (2010)) the Authors have found that spatial gradients of water quality parameter concentrations can show strong correlation with spatial gradients of residence times. In addition, spatial distributions of residence times can also clearly identify areas where waters are either well-mixed or poorly-
mixed/stagnant. As a result of these findings residence time was chosen as the characteristic for homogeneity of water quality in adjacent waters. A flushing module was therefore added to the model to enable computation of residence times.

Residence times, \( \tau \), were computed using a formula developed by Takeoka (1984):

\[
\tau = \int_0^\infty r(t) \, dt
\]

(7)

where \( r(t) \) is the remnant function further expressed as:

\[
r(t) = \frac{M(t)}{M_0}
\]

(8)

with \( M(t) \) and \( M_0 \) representing the mass of dye within the region of interest at time \( t \) and time zero. The remnant function \( r(t) \) denotes the decrease of the dye mass considered as the ratio of the mass of dye whose residence time is greater than \( t \), to the initial mass of dye.

The remnant function depends on local conditions and must be determined experimentally using numerical models; the remnant function in this research is computed following Murakami (1991). Murakami derived the following remnant function expression:

\[
r(t) = \exp(-A_1 t^{B_1})
\]

(9)

The function represents the dye decay curve for a waterbody with tidal exchange. The empirical constants \( A_1 \) and \( B_1 \) depend on the shape of the decay curve and must be determined in each case. Upon determination of \( A_1 \) and \( B_1 \), equation (1) can then be used to calculate the average residence time for the region of interest. Spatial distributions of residence time were computed by considering each grid cell within the model domain as a small, completely mixed reactor. Homogeneity of residence times was taken as an indication of homogeneity of water quality and the waterbody was divided into homogenous sub-regions accordingly. For each homogenous sub-region, the model was then able to compute sub-region average concentrations for water quality parameters for comparison with the single value compliance criteria of TSAS.

4.3 Calculation of Sub-region Average Values

For a sub-region enclosing \( N \) grid cells of the model domain, the sub-region average concentration, \( C_{ra} \), of a particular water quality parameter was determined by averaging first in space and then in time. The spatially-averaged concentration at timestep \( t \), \( C_{sa} \), is the ratio of
the total mass of a parameter in a region to the total volume of water in the region. \( C_{sa} \) can therefore be calculated as:

\[
C_{sa} = \frac{\sum_{n=1}^{N} C_n H_n \Delta x \Delta y}{\sum_{n=1}^{N} H_n \Delta x \Delta y} \quad \text{… at time } t
\]

where \( C_n \) and \( H_n \) are the parameter concentration and water depth at a grid cell \( n \) and \( \Delta x, \Delta y \) are the model grid spacing in the \( x \)- and \( y \)-directions, respectively. If \( C_{sa} \) is then calculated at each timestep during a simulation time \( T \), the region average concentration, \( C_{ra} \), is the arithmetic mean of the spatially-averaged concentrations over the time period \( T \), that is:

\[
C_{ra} = \frac{1}{T} \int_{0}^{T} C_{sa} \, dt \quad \text{… over time } T
\]

In the present research \( C_{sa} \) was calculated by the model at every timestep (25 seconds). The arithmetic mean of these values was then calculated over a period of two spring-neap tidal cycles to give the region average concentration \( C_{ra} \).

The spatial- and temporal-averaging processes might appear simplistic given the significant water quality variations that can occur in large, complex estuaries; however, they were deemed suitable given the high resolution (30m) of the modelled data and the homogeneity and small spatial scales of the sub-regions to which they were applied. In addition, the EPA assessments used for validation of the present approach employ similar averaging processes and some measure of consistency was thought to be beneficial.

### 4.4 Remote Sensing of Chlorophyll \( a \)

Low-altitude airborne remote sensing techniques were used in this research to monitor chlorophyll \( a \) levels and, in particular, to produce spatial maps of estuary-wide chlorophyll distributions; this is a more accurate approach than using high altitude satellite products such as SeaWiFS. Water constituents absorb different colours of light preferentially. Remote sensing, and in particular the radiometric aspect of remote sensing, allows the measurement of water constituents by analysis and quantification of the different colour components of the light reflected from a waterbody. Light is reflected from both the surface of a waterbody (reflected light) and from within the waterbody (back-scattered light). The back-scattered component contains the light from within the water whose spectrum has been modulated by the coloured effects of absorption by water constituents; it can therefore be analysed to determine the concentration of particular water constituents.
Chlorophyll content in water affects the blue, green and red portions of the light spectrum. In the absence of any other major constituent, satellite remote sensing can give an estimate of the chlorophyll content based on the ratio of blue to green light detected. Such ratios have been used with data from ocean colour satellites for the effective mapping of chlorophyll content in clear open waters. This approach fails in coastal regions where dissolved organic matter content and sedimentation produce blue/green effects that confuse the chlorophyll detection algorithms. The airborne remote sensing system used in this research was specifically developed to overcome these problems and was validated for Irish waters (O'Mongain et al., 1996). Upon application to Cork Harbour, the system was tuned using chlorophyll data obtained from ground-sampling.

5 Results and Discussion

The new integrated approach to trophic assessment was developed and tested using Cork Harbour as the study area. Results are now presented and discussed for model validation, the identification of sub-regions using flushing analysis, and validation of trophic assessment. Due to space limitations, only a selection of results from the extensive calibration and validation processes are presented here; readers are directed to Nash and Hartnett (2010) for further details.

5.1 Model Validation

As shown in the schematic of the assessment approach in Figure 3, the hydrodynamic and water quality models were developed sequentially. Before developing the water quality module, the hydrodynamic module was first extensively calibrated and validated. Due to space limitations, only a single validation plot is presented for each of current velocities and water surface elevations. Validation plots of water elevations in Passage West are presented in Figure 5a while similar plots for current velocities in the Main Channel are presented in Figure 5b. It can be seen that a high level of model accuracy was achieved. Similarly high levels of accuracy were achieved at the other validation stations.

Advection-diffusion processes were calibrated and validated in two stages. Firstly, values for longitudinal dispersion and turbulent diffusion coefficients were obtained by calibrating the model against two dye surveys carried out in the field. Figure 6 compares a snapshot of the measured dye plume at low tide on day two of the four day continuous release study at Marino Point with that computed by the calibrated model. Plume extents and concentration gradients can be seen to be quite similar. Transport processes were validated by modelling salinity, a conservative state variable which is only subject to physical transport processes. Summer (June) and winter (November) salinity scenarios were modelled and maximum and minimum concentrations on spring tides were compared against corresponding measured data collected by EPA. The accuracy of the solute transport model is demonstrated by the
linear regression analyses of the maxima and minima data in Figure 7; the correlation coefficient ($r^2$) was 0.97 in both cases.

In the absence of sufficient nutrient data for validation purposes, chlorophyll_a was used to calibrate and validate the water quality module. The remotely-sensed data proved particularly useful for these purposes. Geo-referenced chlorophyll_a concentrations were recorded at discrete points along the flight paths of the aeroplane allowing development of spatial distributions of chlorophyll_a within the harbour at particular stages of the tide. Once again, due to space limitations validation plots are only shown for chlorophyll_a at the time of spring high tide during the month of September (Figure 8). Linear regression analysis (Figure 8c) of the modelled and measured data at 25 discrete locations gave an $r^2$ value of 0.8. Given the complex nature of the primary production cycle, the Authors believe that this level of correlation is satisfactory. The data points used for the regression analysis (Figure 8a) are spread across the full extents of the harbour domain; some are close to nutrient point sources, others are further away. Nonetheless, Figure 8c shows that, with the exception of 2-3 sites, the level of agreement between measured and modelled values was relatively consistent. Although the chlorophyll results suggest an acceptable degree of accuracy in the water quality model it is recognised that further validation of the nutrient cycles is required.

5.2 Flushing Study and Selection of Sub-regions
For the flushing study, firstly the hydrodynamic model was run until steady state conditions were reached; at this juncture a uniform dye concentration of 35 mg/m3 was specified at each wet grid cell within the harbour. Next, the hydrodynamic and solute transport models were run for a further 3,000 hrs for the flushing analysis. Temporal changes in the mean dye concentrations within the flushing region were computed by the model and graphed to obtain the dye decay curves and the remnant function of equation (3) was fitted to the decay curves (using the least squares method) to allow determination of the coefficients $A_1$ and $B_1$. Equation (1) was then used to compute residence times for each grid cell in the region. Figure 9 shows the spatial distributions of residence times (in days) for annual averaged freshwater inflow conditions.

The spatial variation of residence times provided a very good insight into the hydrodynamic circulation patterns of the harbour; for example they enabled easy identification of areas that are well-mixed with a high rate of tidal exchange and other areas such as the North Channel that are quite poorly-mixed and are thus a potential site for eutrophication. A number of areas were identified within which the residence times were quite similar; it was on this basis that the harbour was divided into the seven sub-regions shown in Figure 9. The names of the sub-regions and their spatially-averaged residence times are:

- 1) Lee Estuary - 3.8 days
- 2) Lough Mahon - 31.4 days
3) North Channel - 64.4 days
4) Owenacurra Estuary - 53.1 days
5) Harbour (west) - 26.2 days
6) Harbour (central) - 10.7 days
7) Harbour (east) - 47.1 days

The boundary limits of the sub-regions were also influenced by the location of EPA water quality monitoring to allow comparison between modelled and measured data; for example in sub-region 2 (Lough Mahon), the area of interest for this research, the white dots indicate the locations of monitoring stations. It can be seen that there is a significant variation in average residence time between regions ranging from 3.8 days in the Lee estuary to 64.4 days in the North Channel. This reflects the complexity of the hydrodynamic regime within the harbour.

The division of estuarine and coastal systems into a number of smaller homogenous waterbodies for monitoring and management purpose is now common practice. For example, the EU Water Framework Directive establishes that surface waters should be divided into waterbodies, or management units, which should then be monitored to establish ecological status. While the splitting of large systems into smaller parts is a common management approach, the highly dynamic nature of estuarine systems makes them quite difficult to compartmentalise. The problem is further complicated by the nature of trophic and ecological assessment protocols which then consider such regions to be spatially uniform.

A variety of approaches can be used to subdivide estuaries, including topography, morphology, salinity structure and water quality or a combination thereof (e.g. Ferreira et al. (2006)). However, a common theme in these approaches is their reliance on measured data for the defining parameters. The exclusive use of measured datasets recorded at discrete locations can result in poor representation of the spatial variations in the defining parameters and therefore the mis-identification of homogenous regions. The benefits of the modelling-flushing approach used in this research are two-fold. Firstly, the spatial variations in the chosen characteristic of homogeneity, i.e. residence time, are produced for the complete estuary; this results in a more-informed decision as regards sub-division. Secondly, residence times were chosen as the characteristic of homogeneity as spatial distribution plots of this parameter capture details of both hydrodynamic circulation and solute transport processes. In its current form, the modelling-flushing approach to sub-division is based solely on natural processes but it could be further developed to incorporate anthropogenic processes in the form of nutrient inputs.

5.3 Validation of Trophic Assessment
The numerical model was used to simulate the discharge conditions before and after the construction of the WWTP. In the pre-WWTP scenario, all of the domestic and industrial
discharges listed in Table 2, except those from the plant at Carrigrennan, were included in the simulation. In the post-WWTP scenario, the discharges from Cork City north sewer, Cork City south sewer, Glounthane, Lough Mahon west, Lough Mahon east, Glanmire and Tramore were collected together, subjected to secondary treatment and discharged through the new outfall at Carrigrennan using the discharge parameters specified for Carrigrennan in Table 2.

For both scenarios, the model was run for typical summer and winter conditions with respect to river flows, domestic and industrial discharges, freshwater and marine nutrient fluxes, temperature, and light intensity; thus, four model simulations were run in total:

- Sim1: Pre-WWTP (summer)
- Sim2: Pre-WWTP (winter)
- Sim3: Post-WWTP (summer)
- Sim4: Post-WWTP (winter)

The summer simulations were run for the two-month period 1st May – 31st June whilst the winter simulations were run for the two-month period 1st October – 31st November. Archived start-up grids were used for all water quality parameters to avoid cold-start effects. For all model simulations, region average concentrations for DIN, MRP, DO and chlorophyll_a were calculated using equations (4) and (5). Region average values were also calculated for EPA data recorded at the three monitoring stations shown in Figure 8. Summer and winter average values were calculated for the period 1995-1999 prior to construction of the plant (Toner et al., 2005) and the period 2002-2006 after construction of the plant (Clabby et al., 2008).

Table 3 compares averaged parameter values of measured and modelled data in sub-region 2, Lough Mahon, for the four model simulations. As previously stated, the TSAS water quality compliance criteria are salinity dependant. Based on model predictions, tidally averaged salinity in the Lough Mahon sub-region was approximately 29 during the summer simulation period and 22 during the winter period. The relevant eutrophic criteria (Toner et al., 2005) for these levels of salinity are also presented in Table 3. It should be noted that the modelled salinity values compared well with the mean values calculated from the measured data; for example, the mean summer and winter values calculated for Lough Mahon for the period 2002-2006 were 30.6 and 22.2 respectively (Clabby et al., 2008).

From Table 3 it can be seen that the level of agreement between the measured and modelled region average values varied. Agreement was particularly good for dissolved oxygen for all four scenarios. Agreement was also good for pre-WWTP summer chlorophyll values and although the level of agreement was slightly poorer for post-WWTP summer values, the level of reduction from pre-treatment to post-treatment values was of the same order of magnitude. The agreement between winter pre-WWTP chlorophyll values was poorer still but this period is less important as biological activity is low in winter; both values reflect this low level of activity.
Regarding nutrients (DIN and MRP), percentage differences between measured and modelled data were more significant; however, the values were of the same order of magnitude. The absence of nutrient input data for some of the industries discharging to the harbour most likely contributed to the poorer level of agreement for nutrients. While this is a perennial problem in water quality modelling studies the Authors acknowledge with further calibration/validation of the nutrient cycles model predictions for these parameters might improve. The level of agreement for nutrients was also lower for the post-WWTP scenario than the pre-WWTP scenario. Nutrient data input to the model was collected prior to the construction of the WWTP; it is quite possible that these inputs may have changed in the period between the first assessment (1999-2003) and the second assessment (2002-2006), thus affecting the post-WWTP modelled values. In addition, the post-WWTP EPA data was averaged over the full assessment period 2002-2006 and therefore includes data collected both pre- and post-construction of the treatment plant; in contrast, the modelled scenario simulated the post-construction period only.

Despite the discrepancies in agreement between the nutrient values, at a trophic assessment level both datasets demonstrated a similar trend of improved water quality following the construction of the Carrigrennan WWTP. For the pre-WWTP scenario, non-compliance of both the measured and modelled averaged values was recorded for DIN, MRP and chlorophyll_a (see Table 3). Non-compliance was also recorded for DO, in both cases, based on a breach of the 5 percentile criteria. Failures in all three categories of the compliance criteria thus resulted in a eutrophic classification for both the measured and modelled datasets. For the post-WWTP scenario, the measured and modelled values were compliant with the chlorophyll_a and MRP criteria. Both datasets were also found to be compliant with the DO criteria. However, neither set of values was fully compliant with the DIN criteria; one category failure was recorded for each dataset resulting in an intermediate status classification. To summarise, both the EPA’s trophic assessment approach (i.e. the measured values) and the integrated trophic assessment approach (i.e. the modelled values) resulted in eutrophic status classification pre-WWTP and an improvement to intermediate status post-WWTP. These results suggest that the integrated approach, incorporating the modelling, has potential as a suitable approach to trophic assessment.

The integrated measurement/modelling approach offers a number of benefits over the traditional assessment approaches, such as ASSETS and the OSPAR Common Procedure, that are primarily measurement-based. The model gives much better spatial and temporal coverage of water quality parameters within the study area. As a result, region average values calculated by the model are more representative of the region, as a whole, than average values calculated using data from a small number of discrete monitoring locations. In addition, the use of the model and, in particular, the residence time approach, for the division
of waterbodies into more manageable sub-regions should result in the selection of more homogenous sub-regions than a sub-division process based on measured data alone. The model can also be used to improve monitoring programmes and the Authors are presently using statistical analysis techniques and model results to develop optimised programmes. Lastly, the predictive nature of the model also means that it can be used to determine the potential impact(s) of management decisions on estuarine water quality and, by extension, trophic status – this is a major advantage of the integrated approach.

6 Conclusions

The integrated approach to trophic assessment links data collection, water quality modelling and generic water quality classification systems. The inclusion of the numerical model in the assessment approach means that important factors for eutrophication such as hydrodynamic circulation, river flows, tidal exchange, light attenuation and nutrient inputs are all incorporated directly in the classification process. The incorporation of the model also has other benefits. Firstly, it can be used to investigate the potential impacts of proposed management decisions on water quality and, by extension, trophic status. Secondly, it can be used to optimise water quality monitoring programmes by enabling identification of discrete locations that might be more representative of particular sub-regions. With this in mind, the Authors are currently undertaking research, funded by EPA, into the use of spatio-temporal random field theory (STRF) for combined analysis of model and measured data with a view to assessing the effectiveness of monitoring programmes and devising optimised programmes.

Water quality compliance criteria plays an important role in most trophic assessment schemes (e.g. OSPAR Common Procedure) but their implementation requires the calculation of single values for spatially- and temporally-varying parameters. The spatial and temporal data produced by the model provides a more complete dataset from which more representative sub-region averages can be calculated. Upon application of the integrated approach to Cork Harbour, the modelled sub-region average values gave the same trophic classification as the measured sub-region average values for two different scenarios suggesting that the integrated approach is a valid one. The generic methodology can be used in conjunction with any water quality classification system that uses compliance criteria and can be applied to any coastal waterbody. However, the approach relates only to the water quality compliance element of trophic assessment schemes; it is not proposed as a complete assessment scheme in its own right but rather should be implemented in combination with other elements such as assessment of presence, type and abundance of algal species.

The modelled residence time approach to sub-division of estuarine systems offers an effective alternative to other measurement-based approaches (e.g. Ferreira et al. (2006)). By using residence time as the characteristic for homogeneity, sub-regions are designated on the basis of similar hydrodynamic and solute transport regimes. The approach has significant
applications in the implementation of the EU Water Framework Directive where estuarine and coastal systems must be divided into smaller waterbodies for monitoring and assessment.

The results of the water quality model and the integrated assessment approach were quite promising; however, further work is required. Firstly, for full confidence in the water quality model, the nutrient cycle models must be validated. Secondly, while the light attenuating effects of suspended solids were incorporated indirectly in the model by the development of a site specific light attenuation coefficient based on measured transparency data, it might prove beneficial to incorporate sediments and/or suspended solids directly in the model as state variables.

Acknowledgements

The Authors would like to thank the many organisations that contributed to this research project. In particular, the Authors wish to thank the Irish Environmental Protection Agency for funding part of this research under the Environmental Monitoring, R&D Sub-Programme, Operational Programme for Environmental Sciences, and also for making data available.

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OSPAR (2003) *OSPAR integrated report 2003 on the eutrophication status of the OSPAR maritime area based upon the first application of the Comprehensive Procedure.* OSPAR
### Tables

Table 1: TSAS water quality compliance criteria.

<table>
<thead>
<tr>
<th>Parameter / Waterbody Type</th>
<th>Criterion / Statistic</th>
<th>Period to which Criterion applies¹</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Category A: Nutrient Enrichment</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Dissolved Inorganic Nitrogen (DIN)</strong></td>
<td>mg/l N</td>
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</tr>
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<td>Tidal Fresh Waters</td>
<td>&gt;2.6 median</td>
<td>winter or summer</td>
</tr>
<tr>
<td>Intermediate Waters</td>
<td>&gt;1.4 median</td>
<td>winter or summer</td>
</tr>
<tr>
<td>Full Salinity Water</td>
<td>&gt;0.25 median</td>
<td>winter or summer</td>
</tr>
<tr>
<td><strong>Orthophosphate (MRP)</strong></td>
<td>μg/l P</td>
<td></td>
</tr>
<tr>
<td>Tidal Fresh Waters</td>
<td>&gt;60 median</td>
<td>winter or summer</td>
</tr>
<tr>
<td>Intermediate Waters</td>
<td>&gt;60 median</td>
<td>winter or summer</td>
</tr>
<tr>
<td>Full Salinity Water</td>
<td>&gt;40 median</td>
<td>winter or summer</td>
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<tr>
<td><strong>Category B: Accelerated Growth</strong></td>
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</tr>
<tr>
<td><strong>Chlorophyll_a (Chla)</strong></td>
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<td>or</td>
<td>&gt;30 90 percentile</td>
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<tr>
<td>Intermediate Waters</td>
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<td>or</td>
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<td>or</td>
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<td><strong>Category C: Undesirable Disturbance</strong></td>
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<tr>
<td><strong>Dissolved Oxygen (DO) % saturation</strong></td>
<td>% sat</td>
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<td>or</td>
<td>&gt;130 90 percentile</td>
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<td>Intermediate Waters</td>
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<td>or</td>
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<tr>
<td>or</td>
<td>&gt;120 90 percentile</td>
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¹ Winter extends from October – March inclusive. Summer extends from April – September inclusive.
Table 2: Domestic and industrial discharge details.

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<th>Discharge</th>
<th>Flow (m³/s)</th>
<th>BOD (mg/l)</th>
<th>Org N (mg/l)</th>
<th>Amm (mg/l)</th>
<th>Nitrate (mg/l)</th>
<th>Org P (mg/l)</th>
<th>MRP (mg/l)</th>
<th>DOX (mg/l)</th>
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<td>370</td>
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<td>0</td>
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<td>24.4</td>
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* Replaced by Carrigrennan discharge following construction of Carrigrennan WWTP
Table 3: Region average values of measured and modelled data pre- and post-WWTP and relevant TSAS compliance criteria (shaded cells indicate non-compliance).

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<tr>
<th>Dataset</th>
<th>DIN (mg/l N)</th>
<th>MRP (μg/l P)</th>
<th>DO (% sat)</th>
<th>Chl_a (mg/m³)</th>
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<td></td>
<td>Summer</td>
<td>Winter</td>
<td>Summer</td>
<td>Winter</td>
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<td>2.2</td>
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<td>Post-WWTP Model Averages</td>
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<td>TSAS Compliance criteria</td>
<td>0.63</td>
<td>1.08</td>
<td>47.0</td>
<td>54.0</td>
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Figure 5: Comparison of (a) spring tide water surface elevations in Passage West and (b) current velocities in the Main Channel.
Figure 6: Comparison of (a) measured (μg/l) and (b) modelled dye plumes at low tide on day two of the four day continuous release dye study at Marino Point.
Figure 7: Linear regression analyses of measured and modelled salinity maxima and minima.
Figure 8: (a) Modelled and (b) remotely-sensed chlorophyll_a concentrations at spring high water, September and (c) regression analysis of values at locations in 8(a).
Figure 9: Spatial plot of residence times (Dabrowski, 2005) showing boundaries of sub-regions and monitoring locations (white dots) in sub-region 2.