Articles   School of Civil and Structural Engineering

2011-8

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** Aodh Dowley had a considerable input to the material in the paper. He passed away before seeing the final version of it.

Abstract

This paper compares a 3-dimensional hydro-ecological model with a 2-dimensional model simulating the distribution and fate of Escherichia Coli (E.Coli) discharges from a sewage treatment plant discharging into Dublin Bay, Ireland. Before being discharged, the effluent from the sewage treatment plant is mixed with cooling water from a thermal generation plant resulting in a warm buoyant sewage plume that can be $7 - 9^{o}$C higher than the ambient water in the Bay. The mixing of the stratified plume is complicated by the tidal currents which transport the plume into and out of the estuary. These processes have a direct impact on the transport and fate of E.Coli and the model comparison demonstrates that a three-dimensional model is required to adequately represent the mixing processes in such a stratified environment.

The modelling followed a two-step procedure. First, hydrodynamic simulations of water levels and flow velocities in Dublin Bay were performed using the three-dimensional model TELEMAC-3D. The resulting water level and flow velocity fields were used by the three-dimensional water quality model, SUBIEF-3D to model the transport and fate of E.Coli in the Bay.

Further simulations were performed in which the wind effects on the E.Coli dispersion were included. The water quality simulation was repeated using the 2-dimensional, depth-averaged, hydrodynamic model TELEMAC-2D to compare with the three-dimensional simulations. The results showed that the three-dimensional model gives an adequate representation of the hydrodynamics and water quality in the Bay while the two-dimensional, depth-averaged, water quality model (in comparison to the three-dimensional model) delays the timing of the delivery of E.Coli to the Bay and seriously underestimats the decay rate of E.Coli and the effect of wind on the movement of the buoyant plume of pollution.

Keywords
Three-dimensional, Bathing Water Directive, Escherichia Coli modelling, depth-averaged, wind effect, cooling water discharges, coastal waters.

1. Introduction

The growing demand for electricity production has caused rapid increases in the volumes of hot water discharged into the marine environment.
Ireland's largest power generating plant, the Electricity Supply Board (ESB)
Power Plant at Poolbeg, resides on the South bank of the Liffey Estuary (Figure 1). The oil-powered plant abstracts 2.1 million m3/day from the estuary for cooling purposes, and discharges it back into the estuary at a higher temperature (7 - 9°C above ambient). Before being discharged, the cooling water from this plant is mixed with the sewage effluent from Ringsend Treatment Works resulting in a warm and less saline pollutant plume that tends to float on the water surface in the Bay. Such heated discharges create thermal stratification in the receiving waters. This phenomenon profoundly affects the assimilation of polluting discharges by preventing the mixing (and thus dilution of the contaminants) between the warm upper levels and the cooler water underneath. When waste is added to the flow with, or subsequent to, the thermal discharge the pollution will, nearly invariably, remain in the upper, warm layer. Therefore the volume, and time, available for the self-purification processes can be substantially limited (Ellis et al., 1989). This has a direct impact on the water quality of Dublin Bay and has been a source of concern to the local authorities with respect to water quality legislation, particularly the EU Bathing Water Directive (2006/7/EC) (EC, 2006) which sets stringent limits to the concentrations of Escherichia Coli and Intestinal Enterococci in recreational waters. Hence a hydro-environmental model is needed to evaluate the environmental impact of the discharge on the beaches in the Bay.

Hydro-environmental models, generally, solve the set of governing physically-based equations describing the flow and the transport of contaminants. The true solution of the flow and water quality constituents depends on how accurately these equations reflect the actual physical conditions in the coastal water (Falconer and Lin, 1997). Depending on the approximation of the governing equations, two main types of models have evolved; two-dimensional, depth-averaged, and three-dimensional.

Depth-averaged models (e.g. DIVAST (Falconer, 1984, 1986), MIKE21 (DHI, 2002), TELEMAC-2D (EDF-DRD, 2001)) integrate the hydrodynamic and/or water quality variables over a vertical water column and thus neglect variations in density of contaminant concentrations. These models are ideal for waters where vertical mixing is well established i.e. where there is homogeneity of the transport variables within a water column e.g. salinity, or E.Coli. Depth-averaged hydro-environmental model have a wide range of applications (e.g. Kashefipour et al. 2002, 2006; Schnauder et al., 2007; Riou et al., 2007) due to their reasonable computational cost and the relative ease with which they can be set-up. However, in waters that exhibit considerable stratification in the vertical, as in Dublin Bay, a three-dimensional model is required and this study tests this hypothesis.

Three-dimensional models (e.g. TRIVAST (Binliang and Falconer, 1996), EFDC (Hamrick, 1992), TIDE3D (Walters, 1987) and TELEMAC-3D (Hervouet, 2007)) solve the Navier-Stokes set of equations. Most of these models apply the hydrostatic approximation by assume a negligible vertical acceleration to simplify the Navier-Stokes

equations. Moreover, most of three-dimensional models simulate the mass transport of active tracers (i.e. tracers that influence water density e.g. temperature, salinity and sediments) and incorporate their effect on the flow hydrodynamics. This feature favours the use of such models in stratified environments such as waters that receive cooling water discharges (e.g. Kolluru et al., 2003, Hamrick and Mills, 2000, and Marcos et al, 1997), thermal stratification in lakes and estuaries (e.g. Ji et al., 2007 and Kopmann and Markofsky, 2000), and sediment transport (e.g. Chao et al, 2008, Bai et al., 2003, and Falconer and Lin, 1997).

A number of depth-averaged models have been applied to simulate the water quality of Dublin Bay (e.g. Hussey, 1996; Dowley and Qiang, 1992). However these modelling attempts neglected the effect of thermal discharges from the ESB Plant. This study presents the first attempt to simulate the density-driven flow and E.Coli fields using a three-dimensional model in Dublin Bay using the TELEMAC modelling system.

The TELEMAC modelling suite, developed by the National Laboratory of Hydraulics and Environment (LNHE) of Electricité de France (EDF) was selected for the purpose of the study. The system is based on the finite element method and uses an irregular grid that facilitates selective refinement to better represent boundaries, and small scale features such as discharge outfalls, etc. Moreover, the hydrodynamic model TELEMAC-3D model has a number of powerful features: for example, including the effect of density on hydrodynamics, treatment of tidal flats, boundary-fitting (sigma transformation) method for vertical discretisation, all of which may be required in a model of Dublin Bay.

In this paper, Section 2, outlines the main equations of TELEMAC-3D and the water quality model SUBIEF-3D. Section 3 describes the study area, followed by a description of the model set-up for the study area. The results for the hydrodynamic and water quality simulations are presented in Section 5. The conclusions of the work are summarised in Section 6.

2. Hydro-environmental Model

2.1 Three-dimensional Hydrodynamics: TELEMAC-3D

TELEMAC-3D (EDF-DRD, 1997a) solves the full set of the Navier-Stokes equations describing free-surface transient flow and has a wide range of applications e.g. estuaries, streams, lakes, seas, and coastal waters.

2.1.1 Continuity and Momentum Equations

The hydrostatic version of the Navier-Stokes equations is used in the current study (EDF-DRD, 1997a):

$$\frac{\partial}{\partial t}(u) + u\frac{\partial}{\partial x}(u) + v\frac{\partial}{\partial y}(u) + w\frac{\partial}{\partial z}(u)$$

$$= -\frac{1}{\rho_o}\frac{\partial}{\partial x}(p) + \frac{\partial}{\partial x}(\nu_H\frac{\partial}{\partial x}(u)) + \frac{\partial}{\partial y}(\nu_H\frac{\partial}{\partial y}(u)) + \frac{\partial}{\partial z}(\nu_Z\frac{\partial}{\partial z}(u)) + S_x$$

(1)

$$\frac{\partial}{\partial t}(v) + u\frac{\partial}{\partial x}(v) + v\frac{\partial}{\partial y}(v) + w\frac{\partial}{\partial z}(v)$$

$$= -\frac{1}{\rho_o}\frac{\partial}{\partial y}(p) + \frac{\partial}{\partial x}(\nu_H\frac{\partial}{\partial x}(v)) + \frac{\partial}{\partial y}(\nu_H\frac{\partial}{\partial y}(v)) + \frac{\partial}{\partial z}(\nu_Z\frac{\partial}{\partial z}(v)) + S_y \tag{2}$$

$$p = p_{atm} + \rho_o g(Z-z) + \rho_o g\int_z^Z\frac{\Delta\rho}{\rho_o}dz \tag{3}$$

$$\frac{\partial}{\partial x}(u) + \frac{\partial}{\partial y}(v) + \frac{\partial}{\partial z}(w) = 0 \tag{4}$$

Where, x, y, and z are the Cartesian axes, u, v, and w are the velocity components in the x, y, and z directions $(m\ s^{-1})$, t is the time in seconds, Z is the water surface elevation (m), p is the pressure $(N\ m^{-2})$, ρ_o and $\Delta\rho$ are the reference density and variation in density respectively $(kg\ m^{-3})$, g is the gravitational acceleration $(m\ s^{-2})$, S_x and S_y are source or sink terms (wind, Coriolis force, etc.) $(m\ s^{-2})$, ν_H and ν_Z are the velocity diffusion coefficients in the horizontal and vertical direction respectively $(m^2\ s^{-1})$.

The shallow water equations (SWE) are obtained by depth-averaging the basic equations of Navier-Stokes resulting in the two-dimensional horizontal flow equations of TELEMAC-2D.

2.1.2 Mass Balance Equation of Temperature and Salinity

The mass-balance of each tracer, transported by the water, (e.g. temperature and salinity in this study) is represented by the following equation (EDF-DRD, 1997a):

$$\frac{\partial}{\partial t}(C) + u\frac{\partial}{\partial x}(C) + v\frac{\partial}{\partial y}(C) + w\frac{\partial}{\partial z}(C) = \frac{\partial}{\partial x}(\nu_{HC}\frac{\partial}{\partial x}(C))$$

$$+\frac{\partial}{\partial y}(\nu_{HC}\frac{\partial}{\partial y}(C)) + \frac{\partial}{\partial z}(\nu_{ZC}\frac{\partial}{\partial z}(C)) + Q_C \tag{5}$$

where C is the concentration of the tracer (temperature or salinity), Q_C is the tracer source or sink term (tracer unit), and ν_{HC} and ν_{ZC} are the tracer diffusion coefficients in the horizontal and vertical directions respectively $(m^2\ s^{-1})$.

2.1.3 Density as a function of Temperature and Salinity

In this study, the tracer mass balance, equation (5) is used to compute the values of temperature and salinity every timestep at each node of the computational domain. The effect of these tracers on the water density is described by (Hervouet, 2007):

$$\rho = \rho_o[1 - ((T-T_o)^2\times 7 - 750S)10^{-6}] \tag{6}$$

with $\rho_o = 999.972\ kg\ m^{-3}$ (reference density) and $T_o = 4\ ^oC$ (reference temperature). Hence the density variation $(\rho - \rho_o)/\rho_o$ $(\Delta\rho/\rho$ in Equation 3) can be computed.

2.1.4 Wind Effect

Wind effects are simulated by TELEMAC-3D (EDF-DRD, 1997a) as a two-dimensional condition at the water surface.

$$\upsilon H \frac{\partial \overrightarrow{u_H}}{\partial n} = \frac{\rho_a}{\rho} a_w \vec{w} \|\vec{w}\| \tag{7}$$

where $\overrightarrow{u_H}$ is the horizontal velocity at the water surface and $\vec{w}$ is the wind velocity *10m* above the water, the wind stress coefficient a_w is computed from a formula suggested by the Institute of Oceanography, United Kingdom (Flather, 1976)

2.4 Water Quality Model: SUBIEF-3D

The three-dimensional water quality model SUBIEF-3D (Luck and Guesmia, 2002) was used to simulate the transport and fate of E.Coli discharged from the Ringsend Treatment Works.
SUBIEF-3D takes as an input the hydrodynamics of TELEMAC-3D or TELEMAC-2D. The model can neither compute nor update hydrodynamics and therefore the quality of the results strongly depends on the quality of the hydrodynamic calculation carried out beforehand.
SUBIEF-3D calculates the concentrations of E. Coli according to the following mass transport equation:

$$\frac{\partial(EC)}{\partial t} + \vec{u}.\nabla(EC) = div(\vec{K}.\nabla(EC)) - k_d EC \tag{8}$$

where *EC* is the concentration of E.Coli (*cfu/l*), $\vec{K}$ is the dispersion coefficient (m^2/s), $\vec{u}$ is the flow velocity (*m/s*), and k_d is the E.Coli decay rate (s^{-1}).
The decay of E.Coli is generally defined in terms of T_{90} which is the time during which the original E.Coli population would be reduced by *90%*. The relationship between T_{90} (days) and k_d (d^{-1}) is as follows:

$$T_{90} = \frac{2.303}{k_d} \tag{9}$$

Falconer and Chen (1996) reported that the decay value for E.Coli was typically in the range of 0.05 - $4.0d^{-1}$. Bowie et al. (1985) gives values of coliform decay rates used in modelling studies in the US. A range of $0.48 - 8.0d^{-1}$ was used in estuaries. Also Chapra (1997) estimate the base mortality of coliforms and give the decay rate in saline waters as $1.4d^{-1}$, considerably higher than their value for freshwaters $0.8d^{-1}$. Fujioka *et al* (cited in Thomann and Mueller (1987)) have reported that for faecal coliform the decay rate was in a range of 37-110 day^{-1} in seawater and for sun-light conditions. The faecal coliform decay rate was also reported in the range of 0.0 to 6.1 day^{-1} for different conditions of salinity and sunlight.

In practice, the coliform decay rate can be affected by several environmental factors such as light intensity, water temperature, salinity, suspended solids, pH, etc. (Chapra, 1997; Kashefipour et al., 2006, 2002). However, since the aim of the current study is to study the difference in modelling the transport of E.Coli when using three-dimensional or depth-averaged hydrodynamics, a constant decay rate of E.Coli was employed in all models to ensure comparable E-Coli decay conditions for both the 2D and 3D models. This removes the decay rate as a confounding variable in the model comparisons.

3 Study area

Dublin Bay, located on the east coast of Ireland (Figure 1), is bounded by the rocky headlands of Howth Head and Dalkey. It is about 10 km wide at its mouth and has an area of about 100 km^2. The bed of the bay slopes gently seawards from low water to a depth of about 12 m, thereafter it slopes more steeply to reach 20-25 m approximately on the line between the headlands. Both the north and south sides of the bay have rocky shores, but Howth Head extends slightly further seaward. The mouth of the bay is effectively aligned at 20° to the principal line of the east coast of Ireland and hence to the tidal currents.

Dublin Bay has a high recreational and heritage value. Adjacent to Dublin, the largest and the most populated (approximately 1,190,000) city in Ireland, it is one of the largest amenity areas in Ireland with a variety of water- and land-based activities (swimming, sunbathing, sailing, canoeing, rowing, walking, etc.). It is the focus of the coastal-oriented recreational demands arising from almost one third of Ireland's population (Central Statistics Office, 2006) and is within easier reach of more people than is any other aquatic amenity in Ireland.
The Bay also has a rich heritage of ecosystems within it and around its margins, including ten wildlife habitats within the bay of exceptional scientific and research interest (Environmental Research Unit, 1992).

Figure (1) here

Two main structures lie on the south bank of the Liffey Estuary; the Ringsend Wastewater Treatment Works and the Electricity Supply Board (ESB) power generating plant at Poolbeg. Until 2003, the wastewater at Ringsend Treatment Plant received preliminary (grit separation) and primary (primary sedimentation) treatment before being discharged into Dublin Bay. During the Dublin Bay Project in 2003, a pumping station was built at Sutton (Figure 1) and a 10.5km submarine pipe was laid under the Bay to bring wastewater from the north Dublin pumping station to Ringsend. In addition, the treatment plant was expanded to cater for a population equivalent of 1.7 million. It was upgraded to include secondary and tertiary treatment (ultraviolet disinfection during the bathing season), to meet the standards of the EU Bathing Water Directive (EC, 2006).

The ESB plant which is the largest power plant in Ireland is powered by gas and oil and has an installed capacity of 1020 MW. The steam-driven generating equipment requires 24.2 $m^3\ s^{-1}$ of once-through seawater to cool the heat exchanger and discharges the heated water into the estuary at a temperature of 7-9 ^oC above ambient.
Before being discharged (approximately 120m upstream the discharge weir), the cooling water from this plant is mixed with the sewage effluent from Ringsend Treatment Works. This results in a pollutant plume that is warm and less saline than seawater and remains buoyant on its surface. The stratification in the Estuary is increased by fresh water inflow from the Liffey River. The mixing of the plume is further complicated by the tidal currents which transport the plume into and out of the Liffey Estuary. These density-

driven processes have a direct impact on the water quality of Dublin Bay and hence a three-dimensional hydro-environmental model was set-up.

4 Methodology

4.1 Modelling Approach

The modelling is done in a two-step procedure with the two main components, hydrodynamics and water quality, externally-linked. First the hydrodynamic model is constructed and run to provide hydrodynamic variables (water surface elevations and velocities) that are then used to calculate the transport and decay of E.Coli in a water quality simulation.

The three-dimensional hydrodynamic model TELEMAC-3D was set-up and calibrated for Dublin Bay using a mean neap tidal cycle for a number of reasons: (i) given the long computation time necessary for a three-dimensional model, it was impractical to simulate a full lunar cycle to facilitate a complete comparison, in comparison to a mean spring tide, the neap tide is regarded as a worst case scenario for environmental impact due to water-borne contaminants, and (iii) availability of velocity measurements on days which had a tidal range of a mean neap tide (1.9m).
        The hydrodynamic model was calibrated by varying the bottom friction (Chezy formula). A value for the bottom friction was applied to the bottom and the model was run for a warm-up period of three mean neap tidal cycles after which the model demonstrated a quasi-steady state. It was then run for a fourth cycle to produce outputs for comparison with velocity measurements. The model outputs were water velocity at the five planes of the three-dimensional model.
        The available data for comparison with the model (Table 1) consisted of neap tide velocity measurements (taken on days which had a tidal range of approximately $1.9 \ m$) at eight locations (locations H1-H8 in Figure 1). The data was split into two sets. The first set (time series of water speed and direction at locations H1 -H4) was used for model calibration while the remaining set (measurements at locations H5-H8) was used to validate the model.
The calibrated three-dimensional hydrodynamic model was then re-run to simulate a number of wind scenarios in order to investigate the effect of wind forcing on the transport of E.Coli in coastal waters. Depth-averaged hydrodynamics of TELEMAC-2D were produced for the same wind scenarios in order to facilitate the comparison with the three-dimensional simulations.

The SUBIEF-3D model was set-up to simulate the conditions on July 14th, 2005 on which measurements of E.Coli at the both Ringsend Treatment Works and location WQ (Figure 1) were available. Tidal conditions on July 14th, 2005 corresponded to a mean neap tidal cycle of a range of 1.92m, therefore the SUBIEF-3D model used the hydrodynamics of the calibrated TELEMAC-3D model.
The water quality simulation was run for 9 neap tidal cycles to ensure that any influences of the initial conditions were eliminated.

For each of the wind scenarios simulated by TELEMAC-3D (and TELEMAC-2D), a SUBIEF-3D model was set-up to simulate the transport and fate of E.Coli. This exercise aimed to compare the effect of hydrodynamics on the prediction of E.Coli in the Bay.

4.2 Model domain and mesh

The Dublin Bay model extends a distance of 29.5 *km* in the east-west direction and a distance of 38.5 *km* in the north-south direction (Figure 2).
Its finite element mesh was constructed using the mesh generator MATISSE of the TELEMAC modelling system (EDF-DRD, 1998).
The mesh has 43,742 elements and 23,503 nodes with a mesh size ranging from *750m* at the open sea boundary to *12.5m* around the discharge outfall.
The vertical grid of the Dublin Bay model was constructed by repeating the horizontal mesh 5 times over the vertical to produce a 5-layer model. The horizontal planes of the mesh were positioned at bottom and the water surface and at depths 0.1, 0.3, 0.5, and 0.7 times the water depth.
When water quality simulations are based on the hydrodynamics of TELEMAC-3D, SUBIEF-3D applies the exact level of discretisation as TELEMAC-3D (i.e. it uses the same vertical and horizontal mesh). When the calculations are based on depth-averaged hydrodynamics of TELEMAC-2D, the two-dimensional grid is repeated over the vertical to form the number of required layers.

The SUBIEF-3D model was run on the same domain and mesh of TELEMAC-3D.

Figure 2 about here

Table 1: Data Sources

Data Type	Source Details
Bathymetric data (for the development of the mesh in MATISSE)	Previous studies Bathymetric surveys, and Admiralty Charts (Hussey, 1996; Dowley and Qiang, 1992; UKHO, 1999a, 1999b).
Hydrodynamic data: velocity measurements (for model calibration)	* Points H1-H4 (Figure 1): Measurements spanned full neap and spring tides, taken at five depths in the water column. * Points H5-H8 (Figure1): Measurements spanned full neap and spring tides, taken at two depths in the water column.
Temperature and Salinity data (for hydrodynamic model)	* Points T1 and T2 in the Estuary (Figure1): Depth profile of temperature and salinity taken at different stages of a neap tidal cycle.
Water Quality data: E.Coli measurements	* Point WQ (Figure 1):

on July 14th, 2005 (for model comparison)	Measurements of E.Coli concentration spanning a neap tidal cycle, taken at the water surface. * Ringsend Sewage Treatment Works: Measurements of E.Coli concentration and flow (on the sampling day), obtained from local authorities.

4.3 Initial and Boundary Conditions

4.3.1 Hydrodynamic Model

Initial conditions describe the state of the model at the start of the simulation. The Dublin Bay model was initiated using a fixed initial condition or "cold start" in which water in the model domain was assumed to be initially at rest (i.e. zero flow component values and a still water surface elevation at mean sea level). Background values of $16^{o}C$ and 34PSU were imposed for temperature and salinity respectively.

Five types of boundary conditions were used in the Dublin Bay Model (Figure 2):
(i) Open sea boundaries: Time varying tidal elevations at the north and south boundaries were imposed (Hussey, 1996).
Measurements from gauges in Dublin Bay identified M_2, S_2, N_2, K_2, K_1, and O_1 as the constituents with the largest amplitudes (amplitudes greater than 10 *mm*) (Mansfield, 1992; Hussey, 1996) so these constituents were used to drive the open sea boundary conditions of the hydrodynamic model.

(ii) Coastline: is where the water level intersects the bathymetry. No flow is allowed across this type of boundaries and friction governs the relation between velocity and its gradient along the boundary wall.

(iii) Eastern seaward boundary: This is treated as a mirror-type boundary where water is allowed to flow along/parallel to the boundary but not across it.
This was a reasonable assumption as flow observations and current-meter measurements close to the boundary confirm that the tidal flow pattern in this area is predominantly in the north-south direction.

(iv) Inflow boundary: Discharges of 12.4 and 24.2 $m^3 s^{-1}$ were imposed at the boundary of the Liffey River and ESB outfall at Poolbeg respectively. The values of temperature and salinity at the inflow boundaries are presented in Table 2 below.

(v) Bottom: No flow is allowed through this boundary. A uniform Chezy friction coefficient is imposed at the bottom, the value of which was determined by model calibration (Section 5.1).

Table 2: Boundary Conditions of Temperature and Salinity (TELEMAC-3D model)

	Discharge $(m^3\ s^{-1})$	Temperature (^oC)	Salinity (*PSU*)
Ambient Conditions		16.0	34.0
Liffey River Boundary	12.42	16.0	0.0
ESB Outfall	24.2	21.0 ($\Delta T^* = 5.0$)	30.0 ($\Delta S^* = 4.0$)

*ΔT and ΔS denote the excess in temperature or deficit in salinity relative to ambient conditions in the Bay.

4.3.2 Water Quality Model

As an initial condition, it was assumed that there was no E.Coli in the model domain. This was reasonable assumption since the duration of the "run up" simulation is long enough (9 tidal cycles i.e.4.5 days) to diminish the effect of residual E.Coli at the start of the computation.

The model inflow boundaries are:

(i)   Ringsend Treatment Works: The effluent from Ringsend STW is the major source of coliforms to Dublin Bay. A boundary concentration of E.Coli was prescribed at the ESB outfall (Table 1). The measured E.Coli concentration was diluted by a factor of 8 to account for the dilution of the effluent by the cooling water from the ESB power generation plant at Poolbeg.

(ii)   Liffey River: Water quality measurements (Wilson, 2003) have shown that the Liffey catchment contribution of E.Coli (at average flow conditions) is less than 1% of the total load to the Bay. Furthermore, no storms or heavy rainfall events have preceded the week of the sampling day (the conditions of which are simulated herein) and thus it was reasonable to ignore the riverine input of E.Coli into the Bay for this comparison.

4.4 Wetting and Drying of Coastal Zones

Two options for the treatment of wetting and drying are available in TELEMAC-3D (EDF-DRD, 1997b): (i) correction of water surface gradient in which, the identification of tidal flats and correction of the terms rendered inappropriate by the absence of water (e.g. the gradient of the free surface) is carried out, and (ii) masking of nodes technique which involves the removal of tidal flats or areas from the computation when a pre-defined water depth is reached. The exposed elements still form part of the mesh but do not contribute to the computation. Bedri (2007) compared these two options for the Dublin Bay model and showed that the masking of dry nodes provided a better mass-balance and representation of tidal flats and than the correction of gradients method. Therefore the masking technique was adopted in the current study.

TELEMAC-3D offers two options for the treatment of tracers (temperature and salinity here) on re-wetted zones: (i) "Force to zero" in which the value of a tracer's concentrations at a dry node is set to zero upon re-wetting, and (ii) "Value before Masked" in which the concentration before masking is retained while the node dries and re-wets. Here the second option "value before masked" was applied to avoid the occurrence of freshwater plumes upon re-wetting of the masked area.

SUBIEF-3D (Luck and Guesmia, 2002) also offers the same two options as TELEMAC-3D "Force to Zero" and "Value before Masked".

The second option "Value before Masked" was also applied to the E.Coli at the tidal zones in the model. By retaining the concentrations of E.Coli at re-wetted elements, discrepancies in the E.Coli mass-balance are avoided.

4.6 Modelling Scenarios

Hydrodynamic and water quality scenarios were simulated to (i) study the effect of wind direction on the dispersion of E.Coli, and (ii) demonstrate the variance in simulated E.Coli distribution when using depth-averaged or three-dimensional hydrodynamics in the case of Dublin Bay where conveyance of contaminants are effected by the density-driven flow.

The calibrated neap tide TELEMAC-3D and TELEMAC-2D models were used to simulate four hydrodynamic scenarios:
- (i)   a baseline scenario in which the effect of wind was neglected and
- (ii)   three scenarios in which the wind had the same magnitude but varying directions (Table 3). These scenarios intended to demonstrate the effect of wind direction on the water velocities at the water surface.

The wind magnitude, obtained from (ref), was that of the average wind speed over the bathing season for the years 19??. The selected wind directions for the hydrodynamic simulations are the most frequent directions according to historical records of wind speed and direction at the Dublin Airport Weather station.

To facilitate the comparison with depth-averaged hydrodynamics, the four scenarios were repeated using TELEMAC-2D.

Table 3: Hydrodynamic and Water Quality model scenarios

Hydrodynamic Scenario	Hydrodynamic Model	Wind Speed (m/s)	Wind Direction	Corresponding Water Quality Scenario
HYD3D-W0	TELEMAC-3D	0	---	WQ3D-W0
HYD3D-W1	TELEMAC-3D	4.3	SW (south westerly)	WQ3D-W1
HYD3D-W2	TELEMAC-3D	4.3	SE (south easterly)	WQ3D-W2
HYD3D-W3	TELEMAC-3D	4.3	NE (north easterly)	WQ3D-W3
HYD2D-W0	TELEMAC-2D	0	---	WQ2D-W0
HYD2D-W1	TELEMAC-2D	4.3	SW (south westerly)	WQ2D-W1
HYD2D-W2	TELEMAC-2D	4.3	SE (south easterly)	WQ2D-W2
HYD2D-W3	TELEMAC-2D	4.3	NE (north easterly)	WQ2D-W3

For each of the simulated wind scenarios by TELEMAC-3D and TELEMAC-2D (Table 3), a corresponding SUBIEF-3D model was set-up to simulate neap tide mass transport of E.Coli. The distribution of E.Coli of the eight runs was compared. In all simulations the decay rate of E.Coli (T_{90}) was constant in all runs (at a value of 18 hours).

5 Results

5.1 Hydrodynamic Model Calibration: velocity

Model calibration indicated that a Chezy coefficient of 50 produced the best match between model velocities and measurements.
The results for two representative points (Points H2 and H5 in Figure 1) are shown below.

Station No. H2 The model satisfactorily replicated the general velocity pattern and fitted well to the second peak (maximum ebb) (Figure 3a). The model simulations at 0.3 of the water depth and at the bottom tended to fit well to most of the measurements. The model is reasonably a good match to the period three hours before high water. The absence of velocity measurements for most of the flooding tide made it impossible to assess the model performance at the flooding stage. The model failed to capture the residual currents (these are random velocities of small values that occur close to the time of turn of the tide, i.e. time of high water and low water, caused by the nonlinear interactions of tidal currents and irregular bathymetry) of the neap tide, the residual currents reached approximately 0.1 $m\ s^{-1}$ while in the model they were zero. At this station the measured flow direction was approximately zero degrees (to the North) on the flood tide, and 180° on the ebb (to the South). The model showed identical current direction for all five water depths and they all replicated reasonably well the flow direction at this location in spite of a deviation in the direction during the ebb tide from south to south-west.

Figure 3 here

Station No. H5 The simulated current speed matched measurements on the ebb tide reasonably well (Figure 3b). There were insufficient measurements on the flooding stage, however the model estimates of the maximum flood were reasonably good. Residual currents were also adequately captured. The modelled velocity at 0.7* water depth presented a fair match to current measurements at 3.05 m below water surface (water depth is 25.6 m at this location). The modelled velocity at 0.3 of the water depth considerably overestimated the near bottom measurements of the flood peak. Comparisons of the ebb stage showed that the modelled velocity at 0.5 of the water depth fit reasonably well to measurements taken at 3.05 m (??) below the surface and that the near-bottom simulation of velocity presented a good fit to measurements taken at 3.05 m (??) above the bottom.

The simulated flow direction also presented a reasonably good fit to measurements of flow direction during the ebb tide (where measurements were sufficient to carry out the comparison). The south east flowing currents during the ebb tide change direction to north east then north approximately at the time of low water. This flow pattern was well simulated by the model.

5.2 Hydrodynamic Model: Temperature and Salinity

Measurements of temperature and salinity (shown as points in Figure 4) were taken close to the time of low water where stratification was expected to be greatest. The measured temperature close to the water surface was higher at Point T1 than at Point T2. This

indicates that the surface temperature decreases with distance towards the mouth of the estuary. Similarly, the salinity at the water surface at T1 was less than that at T2 suggesting that a greater degree of stratification is exhibited at T1 compared to T2.

The simulated TELEMAC-3D profiles of temperature and salinity are shown in Figure 4 as continuous lines. It can be observed that the simulated temperature and salinity give a satisfactory match to the measured profiles at T1 and T2. However, note that the simulated temperature and salinity fit the measurements for the lower half of the water column better than the upper half.

Figure 4 here

5.3 Water Quality Model: Effect of varying decay rate T90

The calibration parameter of the water quality model is the E.Coli decay rate (T_{90} value). The effect of varying T_{90} on the simulated E.Coli concentrations was investigated. First a value for the decay rate T_{90} was selected (T_{90}=24 hours) and the SUBIEF-3D model (based on the calibrated TELEMAC-3D hydrodynamics) was run for 9 tidal cycles to ensure that a quasi-steady state was reached. The model outputs of the last tidal cycle were then used for comparison with the measurements which consisted of concentrations of E.Coli taken at the water surface at location WQ. In search of a decay rate that matches the range of available measurements, the run was repeated while changing the value of T_{90} to 16, 12, 6, and 3 hours (Figure 5a) . In all runs all numerical and physical parameters except the decay rate were kept constant.

The above runs were repeated with SUBIEF-3D based on the depth-averaged hydrodynamics of TELEMAC-2D (Figure 5b) using a higher range of T_{90} values (18- 48 hours) in order to match measurements. When SUBIEF-3D computations are based on TELEMAC-2D hydrodynamics, the two-dimensional grid was repeated over the vertical 5 times to match the number of layers in TELEMAC-3D.

Figure 5 here

The lower decay rates (or higher values of T_{90}) required by SUBIEF based on TELEMAC-2D can be explained by the fact that depth-averaged hydrodynamics cannot account for the higher velocities of the stratified layer at the water surface and therefore significantly underestimates the E.Coli delivery rate to the Bay. Hence the decay rate of E.Coli was decreased in order to match the range of observed E.Coli data. In contrast, three-dimensional hydrodynamics account for the buoyancy effects of the discharge from the ESB power plant at Poolbeg, the surface layers of the water travel at a faster speed than in the depth-averaged hydrodynamics resulting in a quicker delivery of pollutants to the Bay.

Note that the range of values used in the above runs (T_{90} =3 – 48 hours corresponding to 1.15/day to 18/day) are well within the range of decay rates for saline waters reported in the literature (see Section 2.4).

5.3 Water quality Scenarios

5.3.1 WQ based on three-dimensional hydrodynamics

The results of scenarios described in Section 4.6 are displayed in Figures 6 -7. These show the distribution of E.Coli at the water surface at two stages of the tidal cycle; low water and mid flood.

During the ebb stage, the tide pushes the sewage effluent plume eastwards out of Dublin Harbour and into Dublin Bay draining water out of the Tolka Estuary and South Bull Lagoon. Once in the Bay, the plume initially flows southwards, then it is deflected northwards towards Dollymount Strand (a recreational bathing area of high national importance) and then eastwards towards Howth Head. By the time of low water (Figure 6) the sewage effluent plume will reach further eastwards into the Bay.

During the flood tide (Figure 7), the incoming water pushes the plume back into the harbour and up the Liffey and Tolka Estuaries while in the inner Bay and in the vicinity of the harbour mouth, the flood tide sweeps the discharge plume northwards towards Dollymount Strand. This stage gives the highest bacteriological counts at Dollymount Strand (particularly at the end nearest to the estuary). This is because at this stage its waters are directly connected to the estuary over the North Bull wall, which is inundated at half tide.
At high water, the plume retreats from the Tolka Estuary, back into the Liffey Estuary and is pushed westwards. The North Bull Lagoon is refilled and the plume is generally contained in the Estuary during this stage of the tide.

The south-westerly wind has the effect of pushing the sewage plume northwards towards Howth Head, Dollymount Strand and into the North Bull Lagoon. South-easterly winds tend to restrict the movement of the plume into the Bay by pushing it towards Dollymount Strand but away from Howth Head. On the other hand, north-easterly winds prevent the plume from reaching Dollymount Strand and the northern shores of the Bay (Figures 6 and 7).
It was noticed that there wind scenarios has reproduced very similar distributions of E.Coli in the sheltered areas (Liffey and Tolka Estuaries) i.e. the wind has less profound effect on the surface water velocities at the Liffey Estuary. This is because the wind force was relatively small in comparison to the tidal force in the Estuary and therefore its effect was less significant. However, in cases where the wind speed is high, a more significant influence of the water velocity in the Estuary is expected.

Figures 6 and 7 about here

5.3.2 WQ based on depth-averaged hydrodynamics

To enable the comparison with the results presented in Section 5.3.1, the E.Coli distribution at the same two stages of the tide (low water and mid flood) are shown herein (Figures 8 and 9). The figures demonstrate that the size and concentrations of the E.Coli plume in the Estuary and Bay simulated by the 2D models is very much less than the

E.Coli distribution produced by SUBIEF-3D based on the three-dimensional hydrodynamics. This is because the depth-averaged hydrodynamics underestimates the surface water velocity and hence results in a delay in the delivery of pollutants to the Bay and a slower movement of E.Coli in the Liffey and Tolka Estuaries. Furthermore, SUBIEF-3D based on depth-averaged hydrodynamics averages the concentration of E.Coli over the water column resulting in a low concentration of E.Coli at the water surface.

Figures 8 and 9 demonstrate negligible differences in E.Coli distribution at the water surface between the simulated wind scenarios suggesting that varying the wind direction has little effect on the 2D hydrodynamics and hence on the transport of E.Coli. This can be explained by the fact that the two-dimensional model averages the wind term in the momentum equation over the water column and hence underestimates the effect of wind. On the other hand the wind force in the three-dimensional models acts at the water surface and therefore presents a more accurate representation of the wind effect.

Figures 8 and 9 about here

5.3.3 Wind Scenarios: Effects on recreational waters

Dollymount Strand is one of the most nationally important recreational areas in Dublin and therefore it is essential to understand and observe the environmental conditions that are likely to negatively impact the quality of its waters.
Based on the modelling exercise carried above, south westerly and south easterly winds push the pollutant plume from the Ringsend sewage treatment works towards Dollymount Strand during both the ebb and flood stages of the tidal cycle and therefore they adversely affect the waters at Dollymount Strand.
The worst possible scenario for the waters at Dollymount Strand can be a westerly wind of a high velocity during an ebb tide that afterwards changes direction to south-easterly.

Although a north easterly wind tends to push the E.Coli plume towards the beaches on the south side of Dublin Bay (e.g. Sandmount Strand and Merrion Strand), these are expected to be less impaired than Dollymount Strand because the South Great Wall which was originally constructed to reduce channel siltation, extends a long distance eastwards into the bay separating the waters of the beaches on the south side of the Bay from the flow exiting the Estuary during an ebb tide. Hence the South Great Wall prevents the E.Coli plume from flowing directly southwards, at least initially, to the beaches of Sandymount and Merrion Strand.

6 Conclusions

This paper compares a two-dimensional model with a three-dimensional hydro-environmental model of the Liffey Estuary and Dublin Bay both subjected to a mixed discharge of sewage effluent from a wastewater treatment works and cooling water discharges from a thermal generation plant. These result in warm buoyant sewage plume that had hindered mixing and dilution within the receiving water. The effect of wind

direction on the dispersion of E.Coli was also studied by performing three model scenarios in which the study area was subject to wind from various directions.

In addition, the simulations were repeated using a 2-dimensional depth-averaged model in order to highlight the differences in the simulated E.Coli distributions when using either two-dimensional depth-averaged or three-dimensional hydrodynamics.

The results demonstrated that the three-dimensional model gives an adequate representation to the hydrodynamic processes and the distribution of E.Coli and gives a better fit to the measured data. Also the effect of wind on the E.Coli distribution using three-dimensional hydrodynamics was more pronounced. The simulations demonstrated that south-easterly and south-westerly winds are more likely to adversely affect the waters at Dollymount Strand.

The use of depth-averaged hydrodynamics in the stratified environment of the Liffey Estuary and Dublin Bay has significantly underestimated the E.Coli delivery rate to the Bay. Also the two-dimensional simulations were less sensitive to the effect of wind due to the depth-averaging of the hydrodynamics.

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