

A Three-Dimensional Hydro-Environmental Model of Dublin Bay

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Received: 23 February 2010 / Accepted: 3 February 2011 / Published online: 23 February 2011
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Abstract In this study, the impact of *Escherichia coli* emissions from a sewage treatment plant on the bathing water quality of Dublin Bay (Ireland) is assessed using a three-dimensional hydro-environmental model. Before being discharged, the effluent from the plant is mixed with cooling water from a thermal–electrical power generation plant, creating a warm buoyant sewage plume that can be 7–9°C higher and is less saline than the ambient water in the bay. The ability of the three-dimensional model in representing such a stratified condition is assessed based on a comparison of its results with two-dimensional modelling results. Hydrodynamic simulations of water levels and flow velocities in Dublin Bay were obtained using the TELEMAC-3D model in one case and the depth-averaged TELEMAC-2D model in the other. The results of each model were separately used as inputs to the water quality model SUBIEF-3D to simulate the transport and fate of *E. coli* in the bay and to generate maps of *E. coli* concentrations over the bay. In addition, the necessity for three-dimensional modelling in simulating the effects of wind direction on the dispersion of *E. coli* was demonstrated by comparing the results of three-dimensional and two-dimensional model simulations with a number of different wind directions. The comparison showed that the three-dimensional model performed better

than the depth-averaged model in simulating the hydrodynamics and resulted in better simulation of the water quality processes in the bay. In particular, the three-dimensional model had reasonably simulated the timing of the delivery of *E. coli* to the bay. Moreover, the effect of wind on the movement of the buoyant plume of pollution and on the *E. coli* distribution was found to be more pronounced with the three-dimensional hydrodynamics. The results demonstrate the need for three-dimensional simulations in situations of density differences or significant wind influences.

Keywords Three-dimensional modelling · Cooling water discharges · Environmental impact assessment · Water quality · Bathing water directive · Depth-averaged · Wind effect

1 Introduction

The growing demand for electricity production has caused rapid increases in the volumes of hot water discharged into the marine environment. In Ireland, the largest power-generating plant, the Electricity Supply Board (ESB) Power Plant (Fig. 1), abstracts 2.1 million cubic metre/day from the Liffey Estuary (which discharges into Dublin Bay) for cooling purposes and discharges it back into the estuary at a higher temperatures (7–9°C above ambient). Before being discharged, the cooling water from this plant is mixed with the sewage effluent from Ringsend Treatment Works, creating a warm and less saline pollutant plume that remains buoyant on the water surface. The resulting stratification profoundly affects the assimilation of polluting discharges by preventing the mixing between the warm upper levels and the cooler water underneath. Therefore, the volume, and time, available for the self-purification

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

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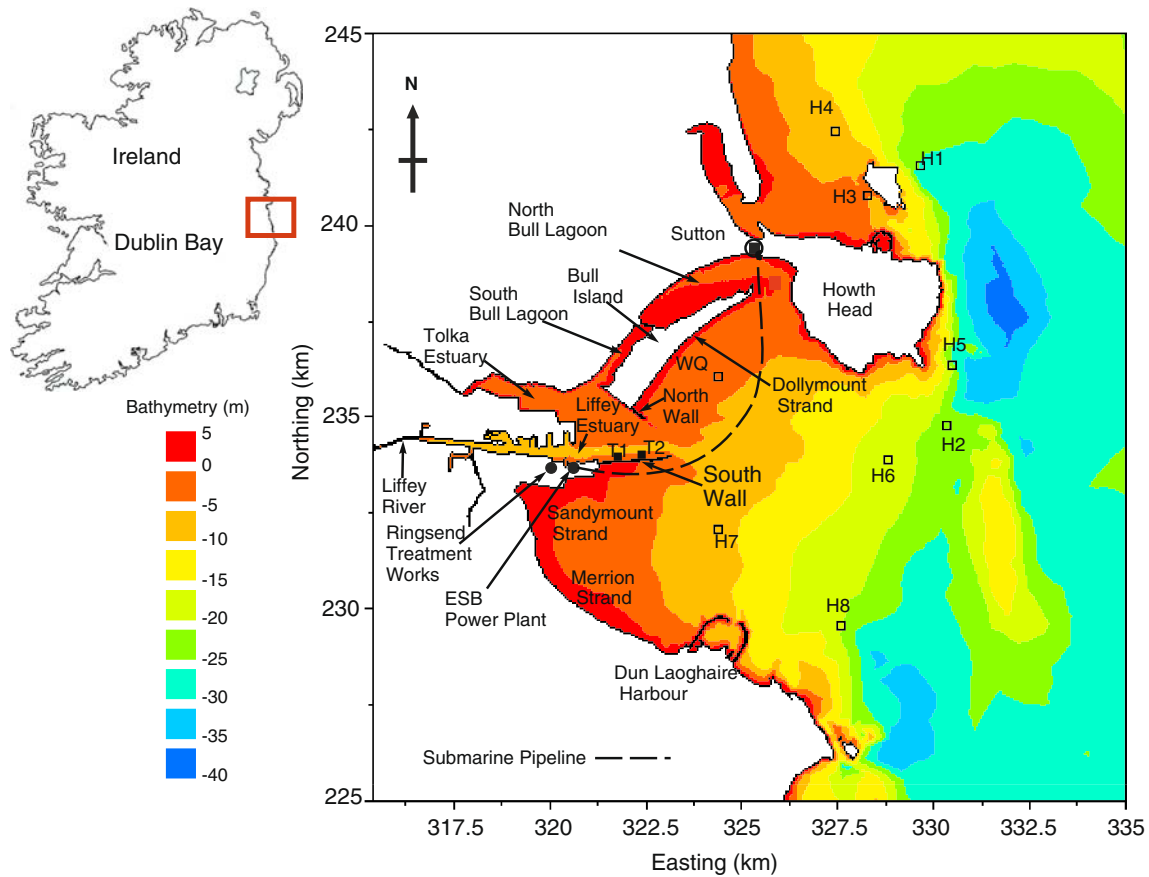


Fig. 1 Liffey Estuary and Dublin Bay. The colours represent the depth of the seabed below means sea level. H1–H8 denote points of current metre measurements. WQ is the water quality measurement location. T1 and T2 are measurement points of temperature and salinity

processes can be substantially limited [1], and this has a direct impact on the water quality of Dublin Bay. The Bay has a rich heritage of ecosystems, including ten wildlife habitats of exceptional scientific and research interest [2]. It is also an important amenity area providing recreational demands for one third of Ireland's population [3] living in Dublin and surrounding areas. Therefore, the water quality of Dublin Bay has been a source of concern to the local authorities with respect to water quality legislations, particularly the EU Bathing Water Directive (2006/7/EC) [4]. This directive sets stringent limits to the concentrations of *Escherichia coli* and intestinal enterococci in recreational waters. In this study, the environmental impact of the Ringsend Treatment Works discharges of these organisms on the water quality of Dublin Bay has been evaluated using a hydro-environmental model which accounts for the stratification due to the mixing of the sewage effluent with cooling water from the ESB plant.

Numerical hydro-environmental models are increasingly used in environmental studies. Typically, they solve a set of governing physically based equations describing the flow and the transport of contaminants. The accuracy of the

solution depends on how adequately these equations reflect the actual physical conditions [5]. In practice, two different types of models have evolved—depth-averaged and three-dimensional—each representing a different compromise between ease of use (depth-averaged models) and better representation of the spatial aspects of the behaviour (three-dimensional models).

Depth-averaged models (e.g. DIVAST [6, 7], MIKE21 [8], TELEMAC-2D [9]) integrate hydrodynamic and/or water quality variables over a vertical water column and thus neglect variations in density and contaminant concentrations over the water column. These models are ideal for waters where vertical mixing is well established, i.e. where there is homogeneity of the transport variables (e.g. salinity, *E. coli*, etc.) within a water column. Depth-averaged hydro-environmental models have a wide range of applications [10–13] due to their reasonable computational cost and the relative ease with which they can be set up. However, in waters that exhibit considerable stratification along the vertical, as in Dublin Bay, a three-dimensional model is required, and this study validates this hypothesis.

Three-dimensional models (e.g. TRIVAST [14], EFDC [15], TIDE3D [16] and TELEMAC-3D [17]) solve the Navier–Stokes set of equations. Most of these models apply the hydrostatic approximation by assuming a negligible vertical acceleration to simplify the Navier–Stokes equations. Also, most three-dimensional models simulate the mass transport of active tracers (i.e. tracers that influence water density such as 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. [18–20]), lakes and estuaries that exhibit thermal stratification (e.g. [21, 22]), and sediment transport studies (e.g. [23, 24, 5]).

In the past, a number of models were applied to simulate the water quality of Dublin Bay (e.g. [25, 26]). However, these studies applied depth-averaged hydrodynamic and water quality models and neglected the effect of thermal discharges from the ESB Plant. This study is a leading study in applying a three-dimensional modelling approach to simulate the density-driven aspects of the flow and *E. coli* fields in Dublin Bay. The TELEMAC modelling suite, developed by the National Laboratory of Hydraulics and Environment of Electricité de France, is at the forefront of three-dimensional modelling and was selected for the study because it provides all features required for the current modelling study. These features include: (1) the use of a finite element unstructured grid which enables the selective refinement of the mesh at key locations in the domain, e.g. discharge outfalls; (2) the robust treatment of tidal flats; (3) the use of boundary fitting (sigma transformation) method for vertical discretisation; (4) density-driven hydrodynamics; (5) heat exchange with the atmosphere; and (6) availability of a range of options for vertical turbulence.

In this paper, Section 2 below 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 setup and the modelling scenarios. The results of the hydrodynamic and water quality simulations are presented in Section 5, and the conclusions drawn from these results are summarised in Section 6.

2 Hydro-environmental Model

2.1 Three-Dimensional Hydrodynamics: TELEMAC-3D

TELEMAC-3D [27] solves the full set of the Navier–Stokes equations for free surface flow environments (e.g. estuaries, coastal waters, seas, streams, rivers and lakes). The current study applies the hydrostatic version of TELEMAC-3D, which reduces the equations to:

2.1.1 Continuity and Momentum Equations

$$\frac{\partial}{\partial x}(u) + \frac{\partial}{\partial y}(v) + \frac{\partial}{\partial z}(w) = 0 \quad (1)$$

$$\begin{aligned} & \frac{\partial}{\partial t}(u) + u \frac{\partial}{\partial x}(u) + v \frac{\partial}{\partial y}(u) + w \frac{\partial}{\partial z}(u) \\ &= -\frac{1}{\rho_o} \frac{\partial}{\partial x}(p) + \frac{\partial}{\partial x} \left(\nu_H \frac{\partial}{\partial x}(u) \right) + \frac{\partial}{\partial y} \left(\nu_H \frac{\partial}{\partial y}(u) \right) \\ & \quad + \frac{\partial}{\partial z} \left(\nu_Z \frac{\partial}{\partial z}(u) \right) + S_x \end{aligned} \quad (2)$$

$$\begin{aligned} & \frac{\partial}{\partial t}(v) + u \frac{\partial}{\partial x}(v) + v \frac{\partial}{\partial y}(v) + w \frac{\partial}{\partial z}(v) \\ &= -\frac{1}{\rho_o} \frac{\partial}{\partial y}(p) + \frac{\partial}{\partial x} \left(\nu_H \frac{\partial}{\partial x}(v) \right) + \frac{\partial}{\partial y} \left(\nu_H \frac{\partial}{\partial y}(v) \right) \\ & \quad + \frac{\partial}{\partial z} \left(\nu_Z \frac{\partial}{\partial z}(v) \right) + S_y \end{aligned} \quad (3)$$

$$p = p_{atm} + \rho_o g(Z - z) + \rho_o g \int_z^Z \frac{\Delta \rho}{\rho_o} dz \quad (4)$$

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 (s); 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}); and ν_H and ν_Z are the velocity diffusion coefficients in the horizontal and vertical direction, respectively ($\text{m}^2 \text{s}^{-1}$).

The two-dimensional continuity and momentum equations of TELEMAC-2D are obtained by depth averaging the Navier–Stokes.

2.1.2 Active Tracers Modelling: Temperature and Salinity

In TELEMAC-3D [27], a general mass balance equation, given below, is used to model the temporal and the spatial variations of concentrations (C) of the active tracers (temperature and salinity herein).

$$\begin{aligned} & \frac{\partial}{\partial t}(C) + u \frac{\partial}{\partial x}(C) + v \frac{\partial}{\partial y}(C) + w \frac{\partial}{\partial z}(C) = \frac{\partial}{\partial x} \left(\nu_{HC} \frac{\partial}{\partial x}(C) \right) \\ & \quad + \frac{\partial}{\partial y} \left(\nu_{HC} \frac{\partial}{\partial y}(C) \right) + \frac{\partial}{\partial z} \left(\nu_{ZC} \frac{\partial}{\partial z}(C) \right) + Q_C \end{aligned} \quad (5)$$

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 ($\text{m}^2 \text{s}^{-1}$).

The resulting values of temperature and salinity at any point in space or time are then used to compute the water density according to state equation [17]:

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

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

2.1.3 Wind Effect

Wind effects are simulated by TELEMAC-3D [27] as a two-dimensional condition at the water surface.

$$\nu_H \frac{\partial \vec{u}_H}{\partial n} = \frac{\rho_a}{\rho} a_w \vec{w} \|\vec{w}\| \quad (7)$$

where \vec{u}_H is the horizontal velocity at the water surface and \vec{w} is the wind velocity 10 m above the water; the wind stress coefficient a_w is computed from a formula suggested by the Institute of Oceanography, United Kingdom [28].

2.2 Water Quality Model: SUBIEF-3D

The three-dimensional water quality model SUBIEF-3D [29] 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 either TELEMAC-3D or TELEMAC-2D. The model can neither compute nor update hydrodynamics; therefore, the quality of the results strongly depends on the quality of the hydrodynamic calculation carried out beforehand.

E. coli concentrations are calculated in the model according to the following mass transport equation:

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

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 (day^{-1}) is as follows:

$$T_{90} = \frac{2.303}{k_d} \quad (9)$$

Falconer and Chen [30] reported that the decay value for *E. coli* was typically in the range of 0.05–4.0 day^{-1} . In the

USA, Bowie et al. [31] reported a range of 0.48–8.0 day^{-1} of coliform decay rates which have been used in modelling studies in estuaries of different conditions of salinity and sunlight. Also, Chapra [32] estimated the base mortality of coliforms and gave a decay rate of 1.4 day^{-1} in saline waters, which is considerably higher than the value of 0.8 day^{-1} for freshwaters. Fujioka et al. (cited in [33]) have reported that for faecal coliform, the decay rate was in a range of 37–110 day^{-1} in seawater and for sunlight conditions.

In practice, the coliform decay rate can be affected by several environmental factors such as light intensity, water temperature, salinity, suspended solids, pH, etc. [32, 10, 11]. With SUBIEF-3D, it is possible to vary the T_{90} value with temperature and salinity only when using TELEMAC-3D hydrodynamics (which computes temperature and salinity variation in a flow field), whereas with two-dimensional hydrodynamics (depth-averaged), a fixed T_{90} value can only be used in SUBIEF-3D. Since the focus of the current article is to compare the effect of three-dimensional and two-dimensional hydrodynamics results on the transport of *E. coli*, the T_{90} value was not allowed to vary in the simulations based on three-dimensional hydrodynamics. 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 (Fig. 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.

Two main structures lie on the south bank of the Liffey Estuary: the Ringsend Wastewater Treatment Works and the 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 (Fig. 1) and a 10.5-km 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 bath-

ing season) to meet the standards of the EU Bathing Water Directive [4] at the designated sampling points in Dublin Bay (e.g. Dollymount Strand, Sandymount Strand and Merrion Strand in Fig. 1).

The ESB plant, which is the largest power plant in Ireland, is powered by gas and oil and has an installed capacity of 1,020 MW. The steam-driven generating equipment requires $24.2 \text{ m}^3 \text{ s}^{-1}$ of once-through seawater to cool the heat exchanger and discharges the heated water into the estuary at a temperature of $7\text{--}9^\circ\text{C}$ above ambient.

Before being discharged (approximately 120 m 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 resulting stratification in the estuary is magnified by freshwater inflow from the Liffey River. In addition, the mixing of the buoyant plume is further complicated by the tidal currents which transport it into and out of the Liffey Estuary. The impact of these density-driven processes on the water quality of Dublin Bay has been studied by a three-dimensional hydro-environmental modelling approach described below.

4 Methodology

4.1 Modelling Approach

Modelling was performed in a two-step procedure that externally links two main models: (1) hydrodynamic model (TELEMAC-3D/-2D) and (2) SUBIEF-3D water quality model. Firstly, the hydrodynamic model was constructed, calibrated and run in order to provide hydrodynamic variables (water surface elevations and velocities) which were used subsequently in the second step 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 three main reasons: (1) 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; (2) in comparison to a spring tide, dispersion of contaminants during the neap tide is significantly less and hence regarded as a worst-case scenario for environmental impact due to water-borne contaminants; and (3) availability of velocity measurements on days which had a tidal range of a mean neap tide (1.9 m).

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 Fig. 1). The data were split into two sets. The first set (time series of water speed and direction at locations H1–H4) was used for model calibration whilst the remaining set (measurements at locations H5–H8) was used to validate the model.

The calibrated three-dimensional hydrodynamic model was then rerun with a number of wind scenarios (used later on for the purpose of comparison) 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 and used as a baseline for comparison with the three-dimensional simulations.

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 [25, 26, 41, 42]
Hydrodynamic data: velocity measurements (for model calibration)	Points H1–H4 (Fig. 1): Measurements spanned full neap and spring tides, taken at five depths in the water column. Points H5–H8 (Fig. 1): 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 (Fig. 1): Depth profile of temperature and salinity taken at different stages of a neap tidal cycle
Water quality data: <i>E. coli</i> measurements on 14 July 2005 (for model comparison)	Point WQ (Fig. 1): Measurements of <i>E. coli</i> concentration spanning a neap tidal cycle, taken at the water surface Ringsend Sewage Treatment Works: Measurements of <i>E. coli</i> concentration and flow (on the sampling day), obtained from local authorities

The SUBIEF-3D water quality model was set up to simulate the conditions of 14 July 2005 on which measurements of *E. coli* at both the Ringsend Treatment Works and location WQ (Fig. 1) were collected. It is worth mentioning that on the data collection day, water samples have been collected from the water surface and stored in 50-ml sterile plastic tubes and preserved in ice-packed containers until they have been analysed for *E. coli*. Microbial enumeration commenced within 24 h of the sample being taken using Colilert and Enterolert reagents produced by IDEXX (<http://www.idexx.com/>).

The SUBIEF-3D model used the hydrodynamics of the calibrated TELEMAC-3D neap tide model as the tidal conditions on 14 July 2005 corresponded to a mean neap tidal cycle of a range of 1.92 m. The water quality simulation was run for nine 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

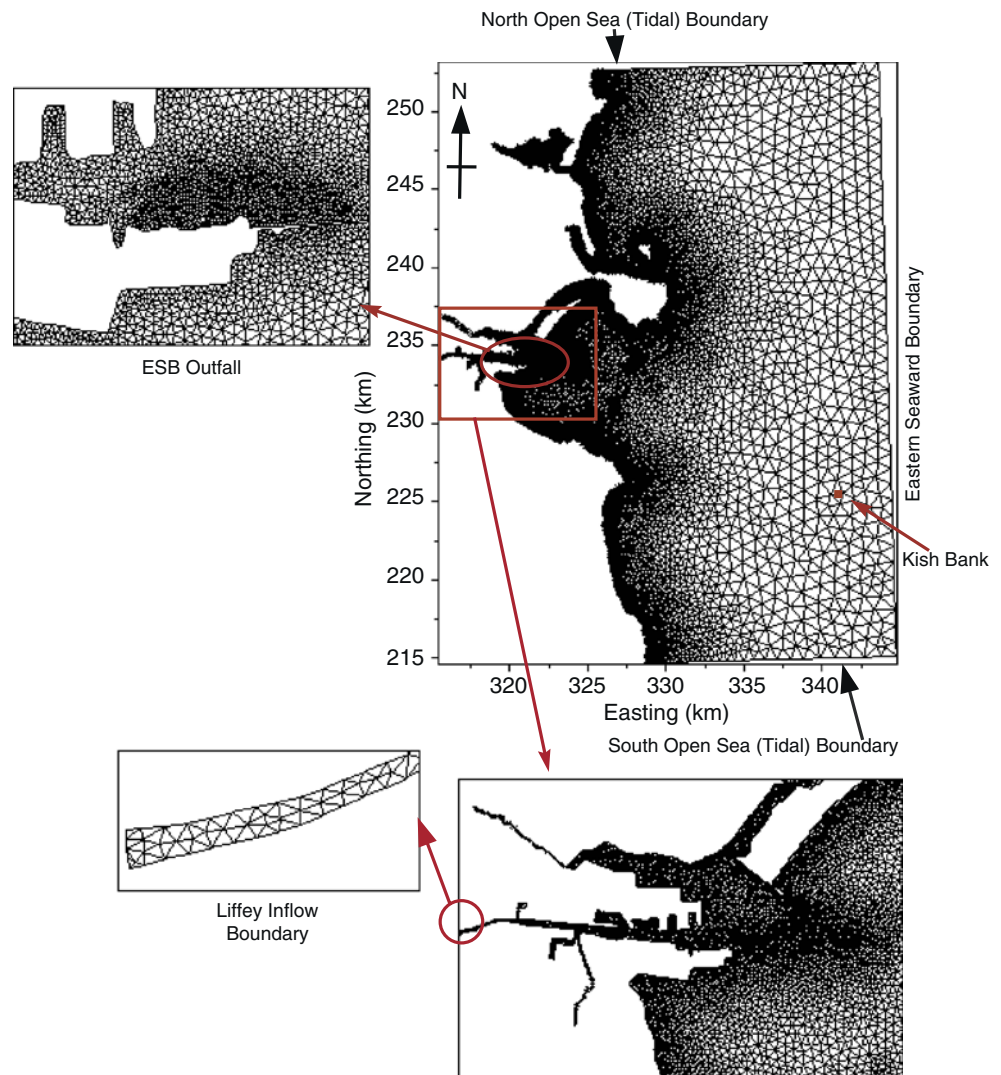
simulate the transport and fate of *E. coli*. This exercise aimed to compare the effect of hydrodynamics on the prediction of *E. coli* in the bay.

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 (Fig. 2). Its finite element mesh was constructed using the mesh generator MATISSE of the TELEMAC modelling system [34]. The mesh has 43,742 elements and 23,503 nodes with a mesh size ranging from 750 m at the open sea boundary to 12.5 m around the discharge outfall.

The vertical grid of the Dublin Bay model was constructed by repeating the horizontal mesh five times over the vertical to produce a five-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.

Fig. 2 Mesh of the Dublin Bay model



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.

4.3 Initial and Boundary Conditions

4.3.1 Hydrodynamic Model

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). For temperature and salinity, background values of 16°C and 34 psu were imposed, respectively.

Five types of boundary conditions were used in the Dublin Bay model (Fig. 2):

1. Open sea boundaries: Time-varying tidal elevations at the north and south boundaries were imposed [25]. Measurements from gauges in Dublin Bay identified M_2 , S_2 , N_2 , K_2 , K_1 , and O_1 as the tidal constituents with the largest amplitudes (amplitudes >10 mm) [35, 25]; therefore, only these constituents were used to drive the open sea boundary conditions of the hydrodynamic model.
2. Coastline: This 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.
3. 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 approximation as previous flow observations and current metre measurements (e.g. [36]) in the outer bay showed that the current direction changes

from 2° (with respect to true North) during the flood tide to 182° during the ebb. The measurements of flow direction (Fig. 3) at Kish Bank (Fig. 2) confirm that the tidal flow pattern in this area is predominantly in the north–south direction. Thus, at this distance from the bay, the flow directions are essentially north–south, parallel to the mirror boundary used in the study.

4. Inflow boundary: Discharges of 12.4 and 24.2 m³ 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.
5. 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).

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 (8 tidal cycles or 4 days) is long enough to diminish the effect of residual *E. coli* at the start of the computation.

The model inflow boundaries are:

1. 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. The measured *E. coli* concentration (Table 1) 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.
2. Liffey River: This river is regulated by an upstream hydroelectric plant and dam, and as a result of this flow regulation, almost no seasonal variation can be observed in the freshwater inflow into the bay. Therefore, an average discharge of 12.4 m³/s was taken as the

Fig. 3 Current metre measurements of velocity speed and direction at Kish Bank

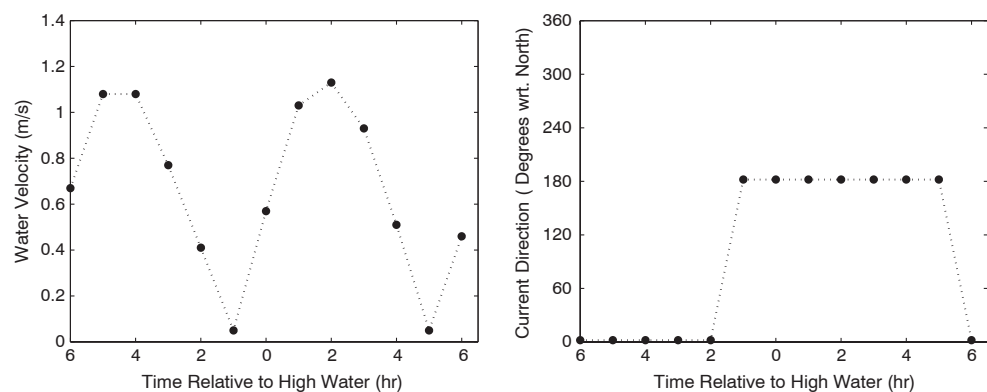


Table 2 Boundary conditions of temperature and salinity (TELEMAC-3D model)

	Discharge ($\text{m}^3 \text{ s}^{-1}$)	Temperature ($^{\circ}\text{C}$)	Salinity (psu)
Ambient conditions		16.0	34.0
Liffey River boundary	12.42	16.0	0.0
ESB outfall	24.2	21.0 ($\Delta T=5.0$) ^a	30.0 ($\Delta S=4.0$) ^a

^a ΔT and ΔS denote the excess in temperature or deficit in salinity relative to ambient conditions in the bay

inflow to the bay. This value may increase during the high flow period when streams and combined sewer overflows contribute to the inflow into the bay. However, in the case under study, dry weather prevailed for the week preceding the sampling day; thus, it was reasonable to ignore the riverine input of *E. coli* which, in such circumstances, contributes <1% of the total *E. coli* load to the Bay [37].

4.4 Wetting and Drying of Coastal Zones

Two options for the treatment of wetting and drying of coastal zones are available in TELEMAC-3D [38]: (1) 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 (2) masking of nodes technique which involves the removal of tidal flats or areas from the computation when a predefined water depth is reached. In the latter, the exposed elements still form part of the mesh but do not contribute to the computation. Bedri [39] 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 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 rewetted zones:

(1) “Force to zero” in which the value of a tracer’s concentrations at a dry node is set to zero upon rewetting and (2) “Value before masked” in which the concentration before masking is retained whilst the node dries and rewets. Here, the second option “Value before masked” was applied to avoid the occurrence of freshwater plumes upon rewetting of the masked area.

SUBIEF-3D [29] 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 as retaining the concentrations of *E. coli* at rewetted elements prevents any discrepancies in the *E. coli* mass balance.

4.5 Modelling Scenarios

A number of hydrodynamic and water quality scenarios have been formulated in order to (1) study the effect of wind direction on the dispersion of *E. coli* and (2) 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 affected by the density-driven flow.

The calibrated neap tide TELEMAC-3D and TELEMAC-2D models have been run using four hydrodynamic scenarios comprising:

1. A baseline scenario in which the effect of wind was neglected
2. 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 was obtained from Rohan [40] and represents the average wind speed over the bathing season for the years 1962–1984. The selected wind directions for the hydrodynamic simulations are the most frequent directions found in the historical records of wind speed and direction at the Dublin Airport Weather station.

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 (southwesterly)	WQ3D-W1
HYD3D-W2	TELEMAC-3D	4.3	SE (southeasterly)	WQ3D-W2
HYD3D-W3	TELEMAC-3D	4.3	NE (northeasterly)	WQ3D-W3
HYD2D-W0	TELEMAC-2D	0	–	WQ2D-W0
HYD2D-W1	TELEMAC-2D	4.3	SW (southwesterly)	WQ2D-W1
HYD2D-W2	TELEMAC-2D	4.3	SE (southeasterly)	WQ2D-W2
HYD2D-W3	TELEMAC-2D	4.3	NE (northeasterly)	WQ2D-W3

For each of the aforementioned four scenarios used in TELEMAC-3D and TELEMAC-2D (Table 3), a corresponding SUBIEF-3D model was set up to simulate neap tide mass transport of *E. coli*, and this results in eight different modelling scenarios. The decay rate of *E. coli* (T_{90}) was kept at a constant value in all eight scenarios, and the resulting distributions of *E. coli* of the eight scenarios were compared.

5 Results

5.1 Hydrodynamic Model Calibration: Velocity

The best match between the modelled and the measured velocities was obtained with a value of 50 for the Chezy coefficient. The results for two representative points (points H2 and H5 in Fig. 1) are shown below.

Station H2 The model satisfactorily replicated the general velocity pattern and fitted well to the second peak (maximum ebb; Fig. 4). The model simulations at 0.3 of the water depth and at the bottom tended to fit well to most of the measurements. Also, the model exhibited a good

match to the period 3 h 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 ms^{-1} , whilst in the model they were zero. At this station, the measured flow direction was approximately 0° (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 southwest.

Station H5 Similar to station H2, the simulated current speed matched measurements on the ebb tide reasonably well (Fig. 4). Based on a comparison with the few measured velocities during the flooding stage, the model was generally able to produce good estimates to the values. In addition, the model captured the residual currents

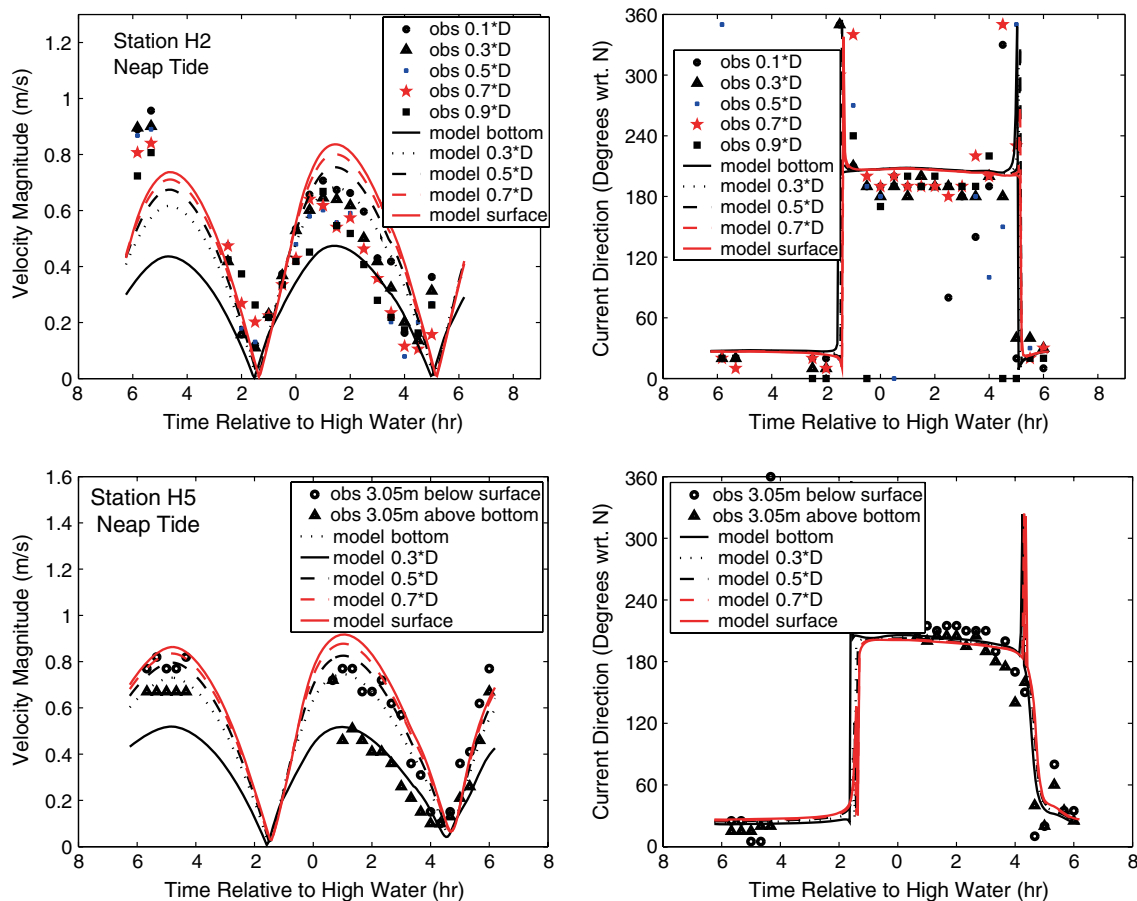


Fig. 4 Comparison of model results and measurements for neap tide at locations H2 and H5

adequately. The modelled velocity at 0.7 of the water depth reasonably matched current measurements at 3.05 m below water surface (water depth is 25.6 m at this location). However, 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 southeast-flowing currents during the ebb tide change direction to northeast 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 Fig. 5) were taken close to the time of low water where

stratification was expected to be at its maximum level. 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 Liffey 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 Fig. 5 as continuous lines. It is obvious that the simulated temperature and salinity represent a good match to the measured profiles at T1 and T2. However, it is noticeable that the simulated temperature and salinity fit the measurements for the lower half of the water column better than the upper half.

5.3 Water Quality Model: Effect of Varying Decay Rate, T_{90}

The governing parameter of the water quality model is the *E. coli* decay rate (T_{90} value) which is a physical quantity that can be determined under controlled laboratory conditions or in situ by tracking a slug of a conservative tracer

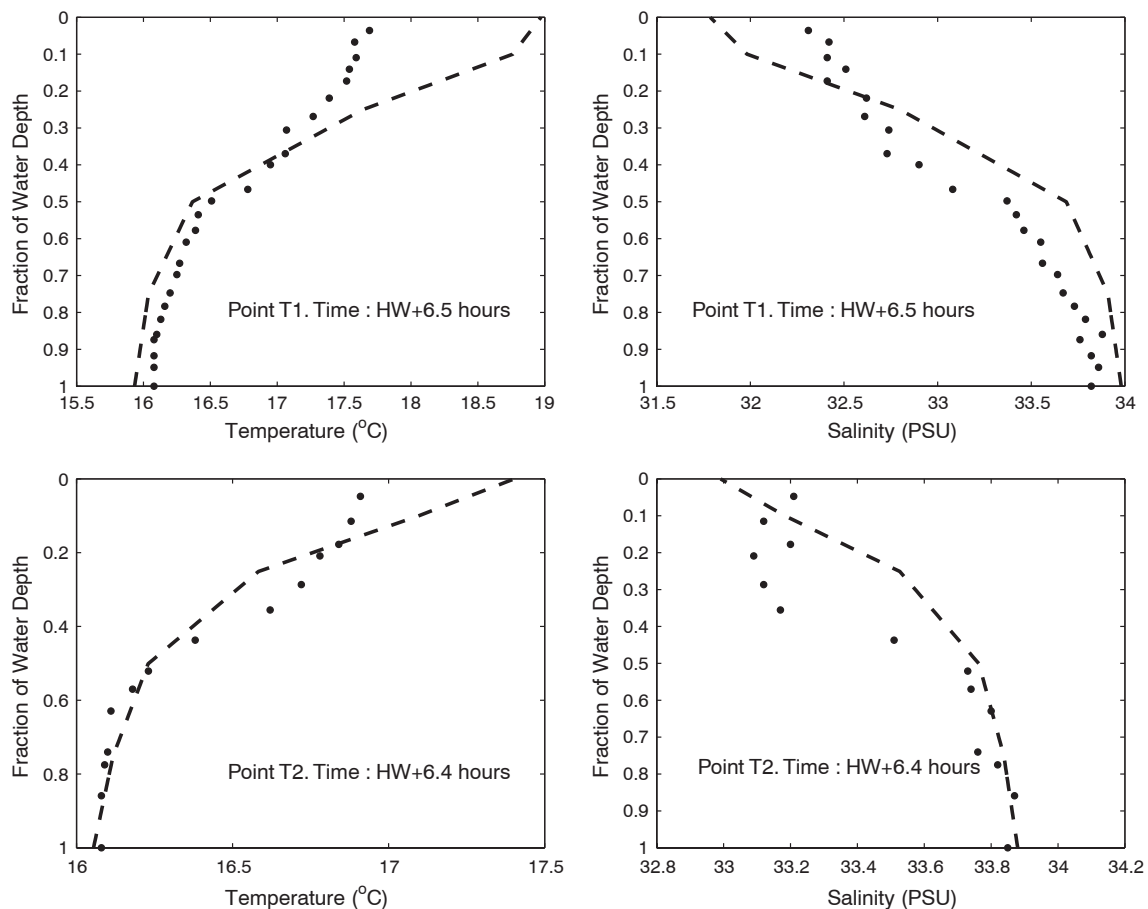


Fig. 5 Vertical profiles of temperature and salinity at points T1 and T2

substance (dye, radioisotope, etc.) added to the discharge (e.g. sewage treatment outfall).

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 ($T_{90}=24$ h) and the SUBIEF-3D model (based on the calibrated TELEMAC-3D hydrodynamics) was run for nine 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 measured *E. coli* concentrations taken at the water surface at location WQ (Fig. 1). Furthermore, the model has been tested with different values of T_{90} : 16, 12, 6 and 3 h (Fig. 6). This allows examining the range of decay rates that match the *E. coli* measurements. During these runs, all numerical and physical parameters except the decay rate were kept constant. Since the decay rate is originally a physical parameter that should be ideally quantified, rigorous calibration of the SUBIEF-3D model was not attempted; instead, a performance envelope resulted from running the model with the different T_{90} values was used to examine the range of decay rates that match the *E. coli* measurements.

The same procedure was also applied for SUBIEF-3D based on the depth-averaged hydrodynamics of TELEMAC-2D (Fig. 6) using a higher range of T_{90} values (18–48 h) in order to match measurements. When SUBIEF-3D computations are based on TELEMAC-2D hydrodynamics, the two-dimensional grid was repeated over the vertical five times to match the number of layers in TELEMAC-3D.

The lower decay rates (or higher values of T_{90}) required in the case of 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.

In general, the range of values used in the above SUBIEF-3D runs ($T_{90}=3$ –48 h corresponding to 1.15–18/day) are well within the range of the published decay rates for saline waters as mentioned earlier in Section 2.2. The current study uses a constant decay rate. This is because using a different day and night decay rate (T_{90}) in the current study will not only require a significant additional computation cost, but is not also expected to result in any significant improvement given (1) the time of the year in which the measurements were taken (July), which witnesses 20 h of daylight and hence the effect of night decay would be insignificant, and (2) the considerable variability in *E. coli* measurements.

5.4 Water Quality Scenarios

In all simulations, the decay rate of *E. coli* (T_{90}) was kept at a constant value of 18 h. The value of T_{90} was chosen based on the simulations in Fig. 6. The chosen value of T_{90} was a compromise based on the simulations shown in Fig. 6. For SUBIEF-3D based on the hydrodynamics of TELEMAC-2D, the lowest T_{90} value to suit the range of data was 18 h. On the other hand, the highest T_{90} to suit the data for the SUBIEF-3D simulations based on the hydrodynamics of TELEMAC-3D was 12 h. Using a T_{90} value of 12 h for all of the comparisons would have resulted in unrealistically low concentrations of *E. coli* in the plots of spatial distribution of SUBIEF-3D based on TELEMAC-2D hydrodynamics.

5.4.1 WQ Based on Three-Dimensional Hydrodynamics

The results of scenarios described in Section 4.5 are displayed in Figs. 7 and 8. These show the distribution of

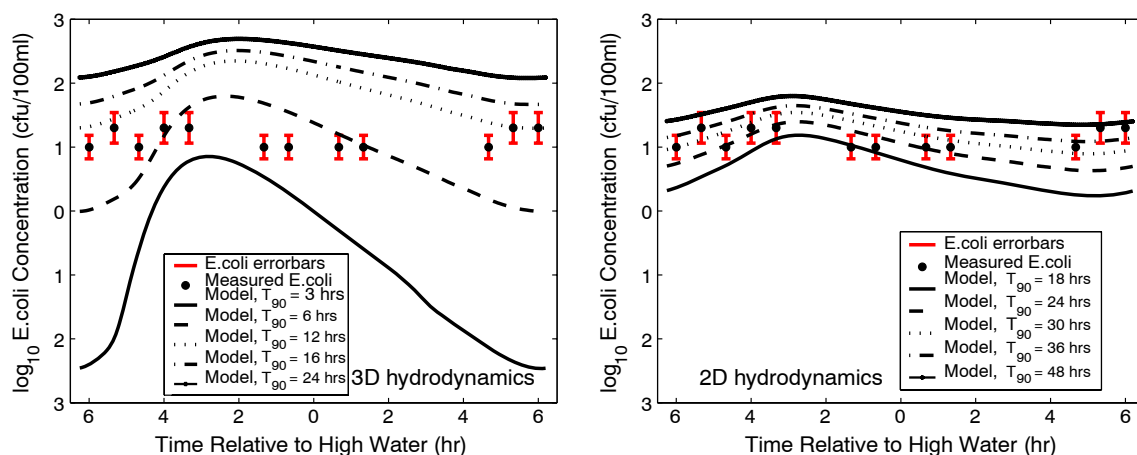


Fig. 6 SUBIEF-3D *E. coli* model: comparison between simulated and measured *E. coli* at location WQ

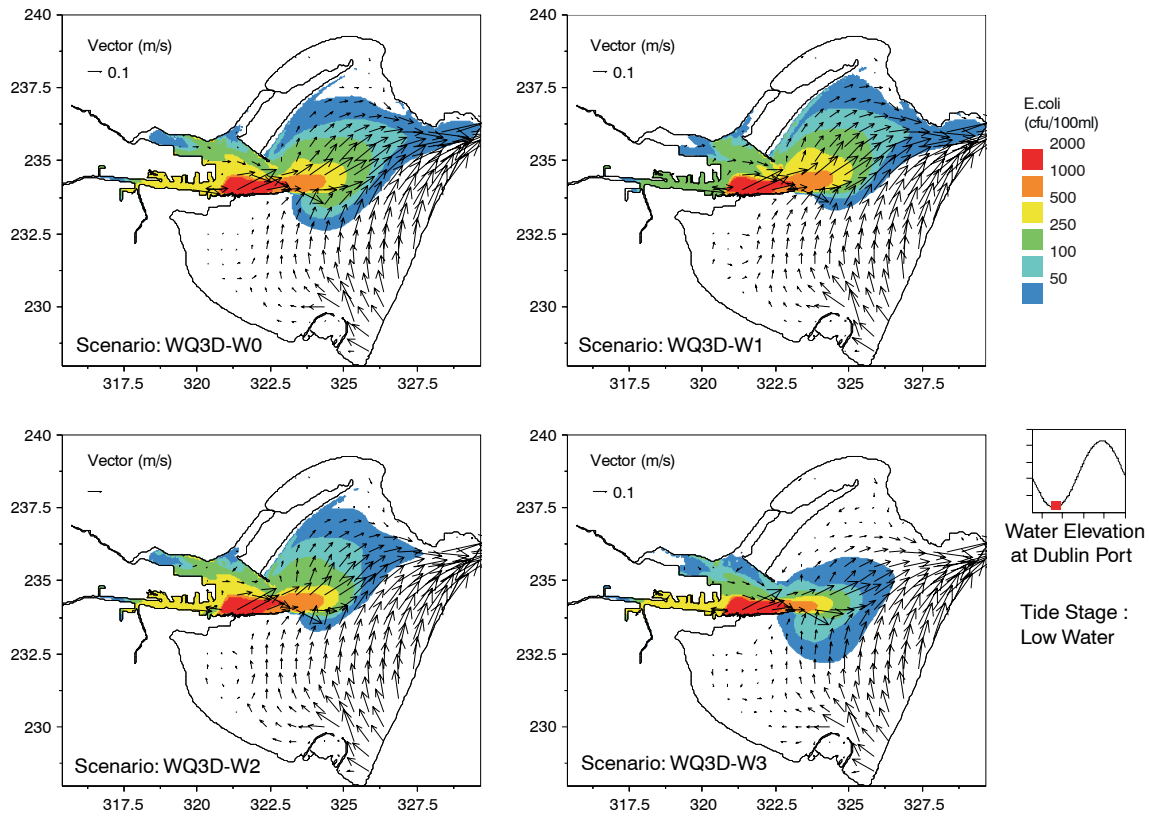


Fig. 7 *E. coli* distribution at the water surface. SUBIEF-3D based on 3D hydrodynamics: low water stage

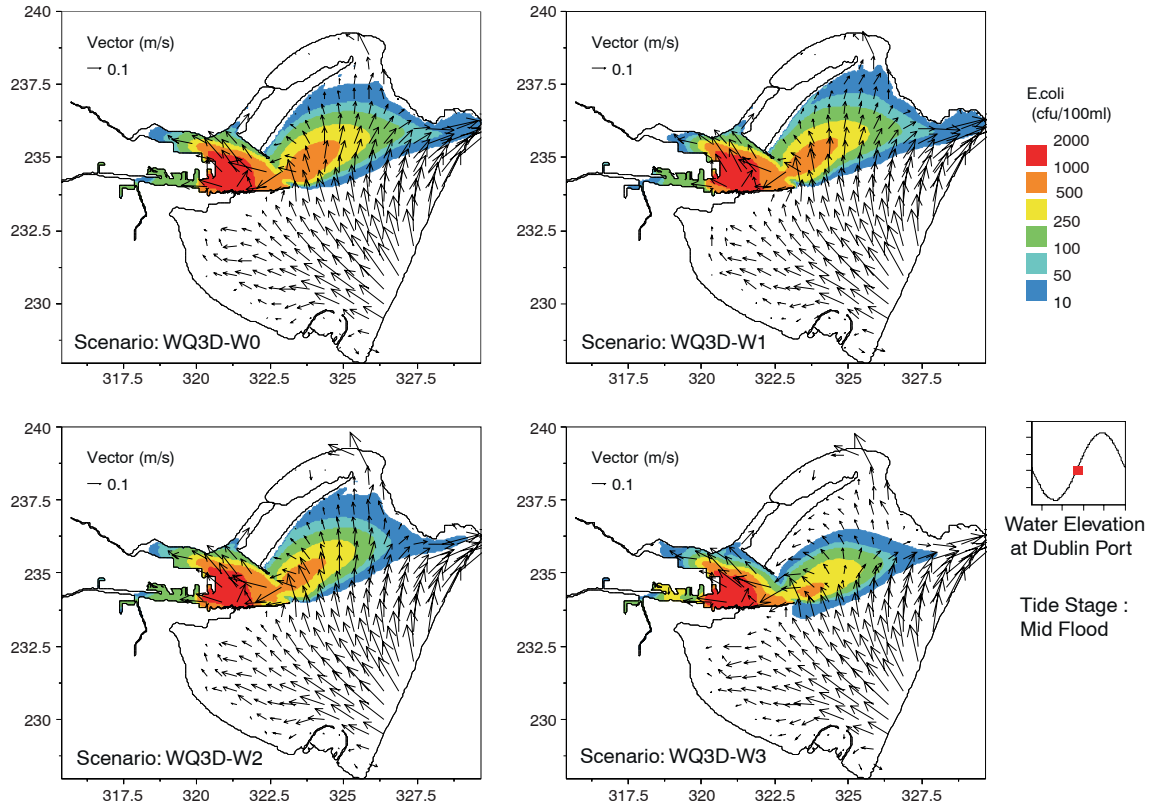


Fig. 8 *E. coli* distribution at the water surface. SUBIEF-3D based on 3D hydrodynamics: mid flood stage

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 and then 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 (Fig. 7), the sewage effluent plume will reach further eastwards into the bay.

During the flood tide (Fig. 8), the incoming water pushes the plume back into the harbour and up the Liffey and Tolka Estuaries whilst 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 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 southwesterly wind has the effect of pushing the sewage plume northwards towards Howth Head, Dollymount

Strand and into the North Bull Lagoon. Southeasterly 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, northeasterly winds prevent the plume from reaching Dollymount Strand and the northern shores of the bay (Figs. 7 and 8).

It was noticed that the wind scenarios have 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; 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.

5.4.2 WQ Based on Depth-Averaged Hydrodynamics

To enable the comparison with the results presented in Section 5.4.1, the *E. coli* distribution at the same two stages of the tide (low water and mid flood) are shown herein (Figs. 9 and 10). The figures demonstrate that the size and concentrations of the *E. coli* plume in the estuary and the bay simulated by the two-dimensional models are very much less than the *E. coli* distribution produced by SUBIEF-3D based on the three-dimensional hydrodynamics.

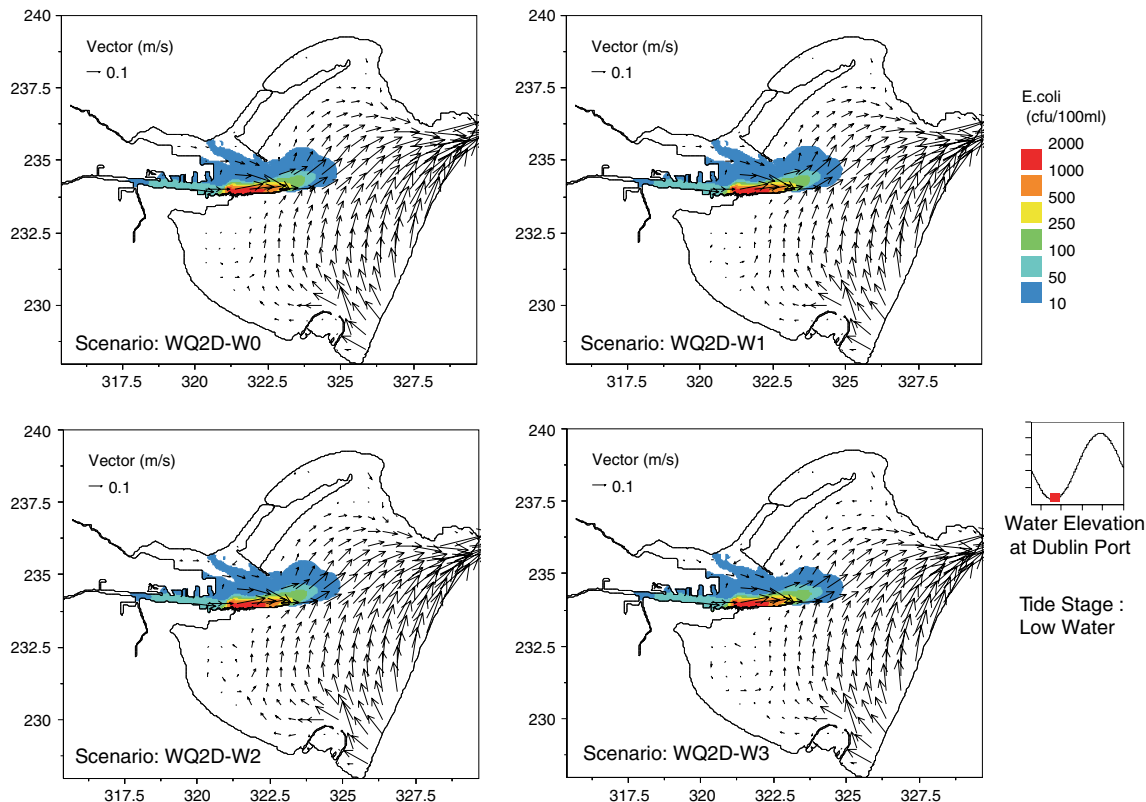


Fig. 9 *E. coli* distribution at the water surface. SUBIEF-3D based on 2D hydrodynamics: low water stage

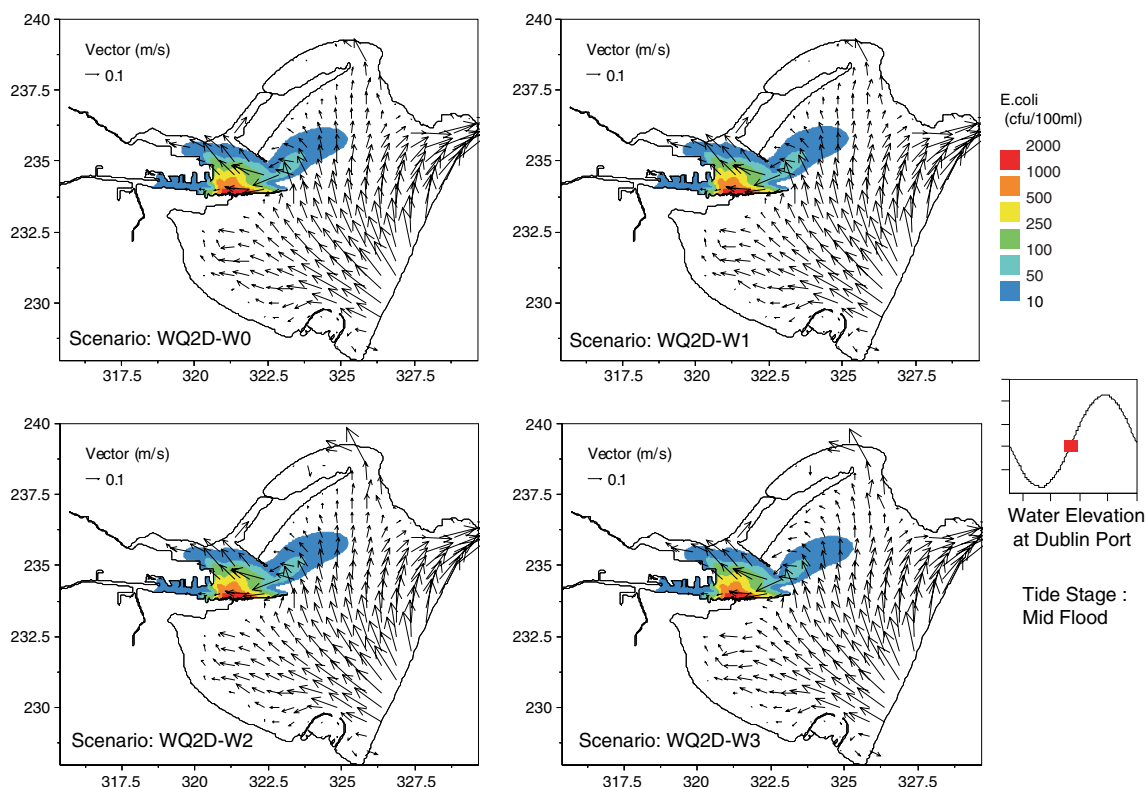


Fig. 10 *E. coli* distribution at the water surface. SUBIEF-3D based on 2D hydrodynamics: mid flood stage

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 9 and 10 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 two-dimensional 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.

5.4.3 Wind Scenarios: Effects on Recreational Waters

Dollymount Strand is one of the most nationally important recreational areas in Dublin; 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 results carried above, southwesterly and southeasterly 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; therefore, they adversely affect the waters at Dollymount Strand.

The worst possible scenario for the waters at Dollymount Strand can be a combination of a westerly wind of a high velocity during an ebb tide that afterwards changes direction to southeasterly.

Although a northeasterly 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 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 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 highlights the need for a three-dimensional modelling approach to study the effect of density differ-

ences and of wind on the distribution of *E. coli* in an estuary and its effects on bathing water quality. This has been demonstrated by comparing the performance of 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 through testing various wind direction scenarios with the SUBIEF-3D water quality model which used the hydrodynamics of the TELEMAC-3D model in one case and of the TELEMAC-2D model in the other.

The three-dimensional modelling results revealed that (1) the three-dimensional model gave an adequate representation of the hydrodynamic processes, (2) the effect of wind on the *E. coli* distribution using three-dimensional hydrodynamics was more pronounced, and (3) the southeasterly and southwesterly winds are more likely to adversely affect the waters at Dollymount Strand.

On the other hand, 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.

Acknowledgements This research has been jointly funded by the INTERREG IIIA (Ireland and Wales) project and the Marine Institute, Ireland.

The TELEMAC package has been developed by Electricité de France and supplied by Hydraulic Research Wallingford, UK under an academic license agreement.

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