3.2

Greater Dublin Drainage Project

SID Application

ABP Case file 312131

Appendix Reference:

32

Appendix Description:

Ringsend WWTP DHI
EIAR Modelling Services

SJK 2 of 3

Sabrina Joyce-Kemper submission 7th June 2024

(3.2)

Ringsend WwTP - EIAR modelling services

Water Quality Modelling

Ringsend Wastewater Treatment Plant Upgrade Project

Report

May 2018

The expert in **WATER ENVIRONMENTS**

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Ringsend WwTP - EIAR modelling services

Water Quality Modelling

Prepared for	Ringsend Wastewater Treatment Plant Upgrade Project
Represented by	Mr. Gordon Barry, JB Barry & Partners

Ringsend WwTP outfall on the Lower Liffey Estuary at Poolbeg

Project manager	Nicholas Elderfield
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Project number	26800565
Approval date	23rd May 2018

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1 Introduction

DHI were commissioned by **J.B. Barry and Partners Ltd.**, working on behalf of Irish Water to perform services relating to water quality modelling for the **Ringsend Wastewater Treatment Works** in Dublin, Republic of Ireland.

Water quality modelling services are required to support the assessment of appropriate final effluent discharge standards associated with the Ringsend Wastewater Treatment Plant Upgrade Project. In addition, the modelling work will also be used in the assessment of the environmental impacts for the purposes of Environmental Impact Assessment and Appropriate Assessment, to be carried out as part of the planning application for the project. In the first instance the outcome of the modelling work will be included in the Environmental Impact Assessment Report (EIAR) for the project.

DHI has previously conducted numerous studies on hydrodynamics and water quality in the Lower Liffey Estuary and in Dublin Bay (Ref. /1-2/). This included the development of a three-dimensional (3D) hydrodynamic water quality model to predict effluent dispersion and plume trajectories. The results of the simulation were part of the previous Environmental Impact Statement for the proposed long sea outfall to relieve the existing Waste Water Treatment Plant at Ringsend (Ref. /1/).

DHI used the existing hydrodynamic model of Dublin Bay and redeveloped it for the objective of performing water quality modelling for the revised Ringsend WwTP. As part of this, DHI recalibrated the model against newly surveyed ADCP and CTD data specific to this investigation. This re-calibrated model was informed by previous DHI studies within the Liffey Estuary and Dublin Bay.

This report details the setup of the data available to the study, the modelling approach and the results of the modelling assessment.

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2 Scope of Work

The objective of the water quality modelling is to assess the fate of a set of key indicators (pollutants) within the Lower Liffey Estuary, Tolka Estuary and Dublin Bay. Pollutants may enter the system via the various rivers, canals, or outfalls (including the treated effluent from the Ringsend WwTP) that discharge into these receiving waters.

The following biological and chemical substances have been assessed:

- Faecal coliforms (Escherichia coli, *E. coli*);
- DIN (dissolved inorganic nitrogen);
- Ammonia;
- MRP (Molybdate reactive phosphorus);
- BOD (biochemical oxygen demand); and
- Total suspended solids (TSS).

To permit the continued discharge of treated effluent at its current location, Irish Water are seeking to include nutrient removal at the Ringsend WwTP. To assess the impacts/effects of this modification, it is necessary to establish the water quality environment for the existing ("baseline") situation. It was proposed for this study that the baseline conditions were established for a typical summer and typical winter periods, based on 3-year average conditions (2013 – 2015, inclusive). Background flows and pollutant concentrations (from rivers, canals and outfalls) were included in addition to the effluent discharge from the Ringsend WwTP.

Following the establishment of the baseline situation, the water quality environment following the construction of the proposed alteration to the Ringsend WwTP can be predicted using information on estimated future emissions.

The change in the water quality environment between the baseline and future emissions scenarios can then be used to inform the environmental impact statement for the project.

The key stages for this study therefore include:

1. Examine water quality monitoring data from within Dublin Bay;
2. Setup and calibrate a 3-dimensional hydrodynamic model;
3. Setup and run water quality models for typical summer and typical winter conditions;
4. Validate the water quality model for summer and winter conditions;
5. Setup and perform baseline modelling scenarios; and
6. Undertake the future "with scheme" modelling scenarios.

These stages are outlined in the following sections of the report.

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3 Study Area

3.1 Geographic Setting

The area of interest for the present study was the estuaries of the Liffey, the Tolka, Dublin Bay and the immediate environs of the Irish Sea, as shown in Figure 3.1.

Dublin Bay is an inlet of the Irish Sea on the east coast of the Republic of Ireland. The Bay can be defined as the area of water enclosed by Howth Head in the north to Dalkey Head in the south - approximately 10 kilometres. The Bay is relatively shallow with water depths generally less than 10m. There exist large intertidal areas with exposed sand and mud flats at low water.

Dublin Port is situated at the mouth of the River Liffey and within the innermost part of Dublin Bay. The Ringsend WwTP is located on the south bank of the River Liffey, from where the Great South Wall extends over 4 kilometres into Dublin Bay. The WwTP discharges into the Bay receiving waters of the Liffey on the north side of the Great South Wall.

To the north of the Port, the River Tolka also discharges into Dublin Bay at Clontarf. The Tolka Estuary is separated from the Irish Sea by the North Bull Wall, which extends 3 kilometres into Dublin Bay. Bull Island is located on the seaward side of the North Bull Wall and extends toward Howth Head to the north-east of Dublin Bay. Bull Island has formed as a long-term consequence of changes to siltation since the construction of the North Bull Wall in the early nineteenth century. The River Santry discharges in the lagoon behind Bull Island and exits through the outlet to the North of the causeway connecting Bull Island to the mainland.

The southern part of Dublin Bay, i.e. south of the Great South Wall, is characterised by an area of mud flats and beaches. Several of these beaches are designated bathing waters.

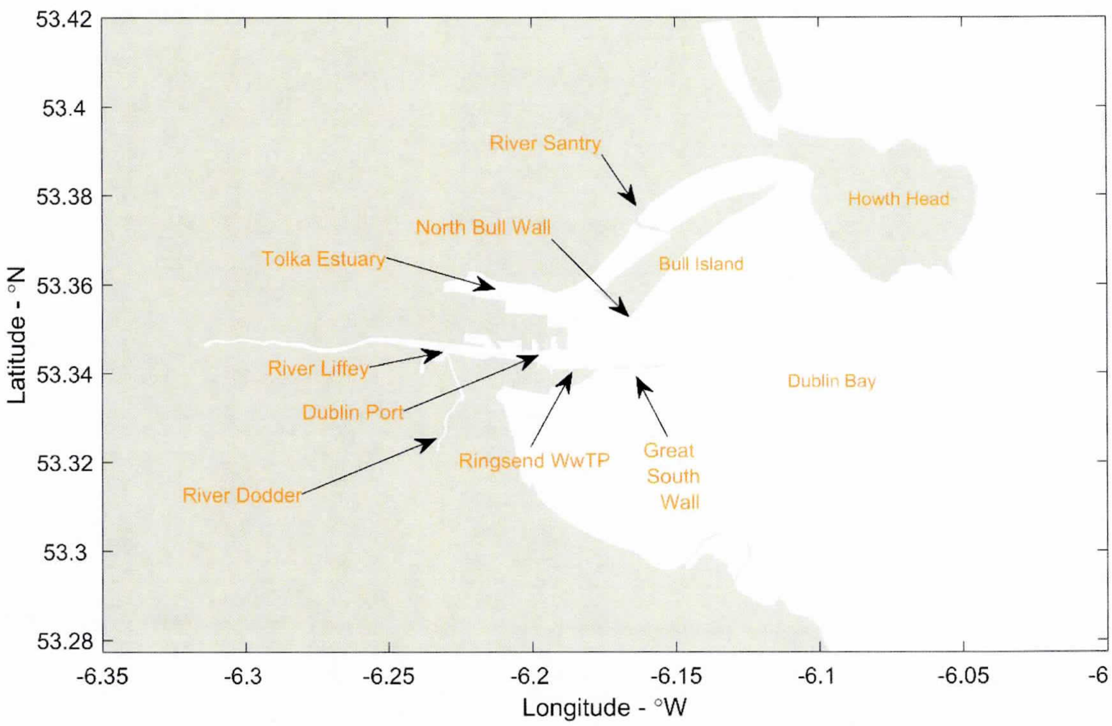

Figure 3.1 Map of Dublin Bay showing key locations as referred to in the text.

3.2 Hydraulic Setting

3.2.1 Tide

The tide in Dublin Bay is semi-diurnal in nature with an average tidal range of approximately 3.4 m during spring tide and 1.9 m during neap tide. The astronomical tidal states for Dublin Port are given in Table 3.1 with reference to Chart Datum and Ordnance Datum (OD) Malin, which is approximately 0.1m below mean-sea-level (MSL).

Table 3.1 Astronomical tidal conditions for Dublin Port (Ref. Dublin Port Tide Tables 2016). N.B. Dublin Port tide tables note that LAT is ~0.1m below Chart Datum.

Tidal state	Levels to Chart Datum	Levels to OD Malin
HAT	+4.50m	+1.99m
MHWS	+4.10m	+1.59m
MHWN	+3.40m	+0.89m
MSL	+2.40m	-0.11m
MLWN	+1.50m	-1.01m
MLWS	+0.70m	-1.81m
LAT	-0.1m	-2.61m
Chart Datum	0.00m	-2.51m

3.2.2 Rivers

There are three major rivers (namely the Rivers Liffey, Dodder and Tolka) plus a number of smaller rivers and canals that discharge into Dublin Bay. Together with the tide, the discharge from these sources sets the flow and determines the vertical distribution of temperature and salinity and the horizontal position of this in the estuary.

River Liffey

The River Liffey is the largest river to enter Dublin. The catchment area (1,370km^2) is divided into three parts according to Ref. /3/:

The upper catchment area (308 km^2) is very mountainous and responds quickly to heavy rainfall. The Pollaphuca Dam is located at the end of the upper catchment area, with the Golden Falls Dam situated 2km further downstream. The inflow to the Golden Falls reservoir is equal to the outflow of the Pollaphuca reservoir. The Pollaphuca reservoir acts as a flood relief reservoir subject to ESB (Electricity Supply Board) operating guideline restrictions intended to avoid overtopping. In addition, a minimum compensation flow of 1.5 m^3/s applies at Pollaphuca, which arises under the Liffey Reservoir Act 1936.

The middle catchment area (534 km^2) is characterised by a rather flat landscape with the Leixlip Dam at the downstream end.

The lower catchment area (528 km^2) is flat and discharges through Dublin into Dublin Bay and the Irish Sea. There are four important tributaries between the Leixlip Dam and the Irish Sea (over a distance of 20 km): Rye Water (215km^2), Griffeen (50 km^2), Cammock (84 km^2) and Dodder (113 km^2). The Dodder enters the Liffey just upstream of the Ringsend WwTP at Dublin Port and has, as such, little influence on the flows of the Liffey through the city.

Apart from the above-mentioned rivers, the Liffey is also fed along the route by an unknown number of small outfalls, contributing urban runoff and local drainage flows. The tidal limit (and the proposed limit of the present modelling study) is Islandbridge Weir on the River Liffey.

River Dodder

From Ref. /3/, it is known that the Dodder is the smallest river (in catchment area) of the three principal rivers (the Tolka, Liffey and Dodder) entering Dublin city. It is, however, the second largest in terms of discharge. The Dodder has a long history of flooding, more than any other river in Dublin. The total catchment area is 113 km^2 with a steep mountainous (1/20) and a fast reacting upper and middle catchment area and a flat lower (Dublin) catchment area. In the upper area, there are two reservoirs (Upper and Lower Bohernabreena Reservoir), but they collect runoff water from only 28 km^2 (25%) of the total catchment area. Some important tributaries such as the Owendoher and Little Dargle are contributing downstream of the dams.

River Tolka

From Ref. /3/, it is known that the River Tolka is the second largest river in terms of catchment area to enter Dublin. It is, however, the smallest in terms of discharge. The River Tolka has a catchment area of 141 km^2. In the upper catchment, the river is just a stream with small meanders and low banks with a relatively flat bed gradient of about 0.4%. The river is 2.5 m to 5m wide. Occasional flooding causes a flood plain extending up to 400 m wide.

Entering urban environments, the profile of the river changes noticeably. Through the Tolka Valley Park, Botanic Gardens and Griffith Park, it becomes somewhat wider and straighter, with generally higher and more defined grass banks. In its latter reaches through Glasnevin, Drumcondra and Marino, the river becomes increasingly canalised. In this section, the riverbank varies from natural riverbank to an ad hoc arrangement of walls of varying height. Downstream of Drumcondra, the river is also subject to tidal influence, and the channel is wider with more formal riverside walls in the lower section.

Minor Rivers and Streams

The Santry is a small river of approximately 7 km length with a catchment area of ~16 km^2. The river flows through predominantly urbanised and industrial areas on the north side of Dublin and enters Dublin Bay via a culvert behind Bull Island. The Bull Island causeway forms a barrier to flow and hence the Santry discharges to the north and has no direct connection with the Tolka.

The Elm Park Stream and the Trimleston Stream are small urban watercourses in the south of Dublin. These streams are not large, but likely receives urban runoff due to surface water drainage. Both discharge into the south of Dublin Bay near designated bathing water beaches.

3.3 Environment

3.3.1 Designated Areas

Within Dublin Bay there are two Special Areas of Conservation (SACs) designated under the EU habitats Directive.

- South Dublin Bay SAC: located to the south of the Great South Wall and primarily designated for presence of extensive Tidal Mudflats and Sandflats.
- North Dublin Bay SAC: the area behind Bull Island is selected for a range of habitats species including Tidal Mudflats, Sandflats, and Fixed Dunes.

Within the inner part of Dublin Bay and its estuaries, there are two designated Special Protection Areas (SPAs) under the terms of the EU Birds Directive (2009/147/EC):

- South Dublin Bay and River Tolka Estuary SPA: includes a substantial part of Dublin Bay and the estuary to the Rover Tolka to the north of the River Liffey.
- Bull Island SPA: covers the Inner Part of North Dublin Bay extending from Bull Island to Howth Head.

In addition to these designations, the Liffey and Tolka estuaries are designated as nutrient sensitive under the Urban Waste Water Treatment Directive.

3.3.2 Water Quality

The qualitative and quantitative status of the water quality environment of Dublin Bay and its estuaries is governed by the EU water framework directive (WFD). Table 3.2 summarises the relevant standards that must be achieved to meet the environmental objectives specified in the WFD for surface waters (Ref. /4/). The standards are defined according to two relevant categories, transitional waters (estuaries) and coastal waters. Figure 3.2 shows the definition of these areas in relation to Dublin Bay.

The most recent published status (2010-2015) of the transitional waterbodies are:

- Liffey Estuary Upper – Moderate Status
- Liffey Estuary Lower – Moderate Status
- Tolka Estuary – Moderate Status

The most recent published status (2010-2015) of the coastal waterbody are:

- Dublin Bay – Good status.

The WFD risk score shows that all sites (transitional and coastal) are at risk of not achieving good status.

Table 3.3 summarises the relevant status for bathing water quality (Ref. /5/). There are three designated bathing water areas within Dublin Bay (Figure 3.2). The most recent status of these bathing water areas is:

- Dollymount Strand – good
- Sandymount Strand – sufficient
- Merrion Strand – poor

Table 3.2 Environmental quality standards as specified in the European Communities Environmental Objectives Surface Waters 2009 (Ref. /4/).

Parameter	Description	Transitional water body	Coastal water body
Biochemical Oxygen Demand (BOD)	European communities environmental objectives (surface waters) regulations 2009	95 %ile concentration: ≤ 4 mg/l	N.A.
Dissolved Inorganic Nitrogen (DIN)		N.A.	Median concentration: ≤ 0.17 mg/l (High status) ≤ 0.25 mg/l (Good status)
Moly date Reactive Phosphorus (MRP)		Median concentration: ≤ 0.04 mg/l	N.A.

Table 3.3 Environmental quality standards for bathing waters as specified in the European Communities Environmental Objectives Bathing Waters 2008 (Ref. /5/).

Parameter	Description	Concentration (No./100ml)		
		Excellent quality	Good quality	Sufficient quality
Escherichia coli (E. coli)	European communities bathing water quality regulations 2008	250*	500*	500**

* Based on 95% of samples or more, ** Based on 90% of samples or more

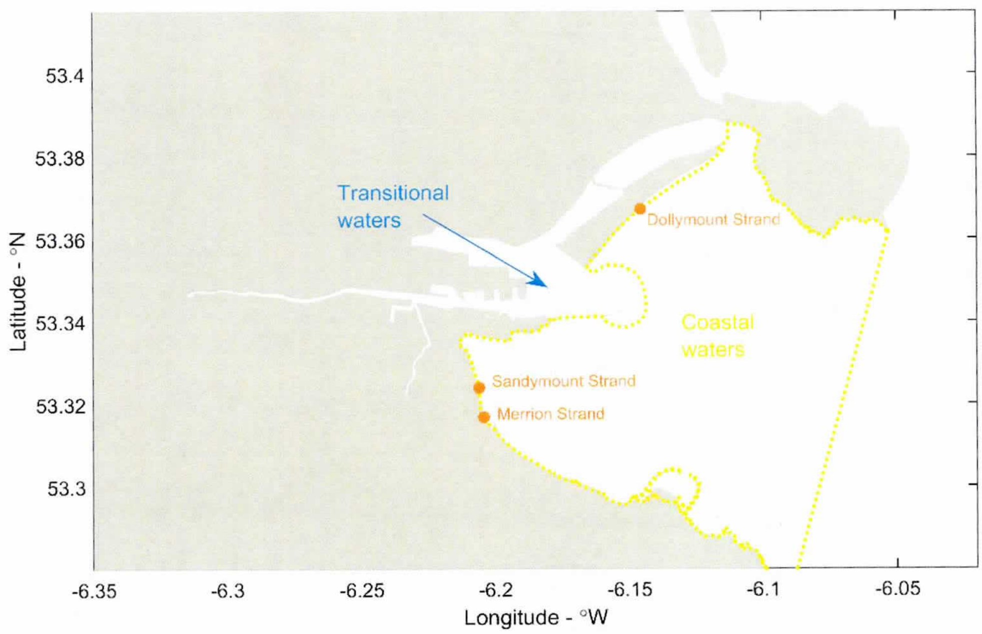

Figure 3.2 Map of Dublin Bay showing definitions of transitional waters and coastal waters and locations of bathing water beaches (orange).

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4 Available Data

4.1 Hydrometric/Hydrodynamic Data

Hydrometric data includes information on river flow rates and water levels, whilst hydrodynamic data refers to current speeds, temperatures and salinities in Dublin Bay and its estuaries.

These data were obtained from several sources and were analysed in order to ensure consistency and reliability for use in the study. The data were used for the following purposes:

1. To provide background conditions used as inputs to the hydrodynamic model (e.g. flow rates from rivers); and
2. As calibration and validation data for the hydrodynamic model (water levels, current speeds, temperature, and salinity)

4.1.1 River Flow Rates

There are three major rivers (namely the River Liffey, Dodder and Tolka) plus several smaller rivers and canals that discharge into the study area.

Flow data for the relevant rivers and tributaries was obtained, where available, from the Environmental Protection Agency's (EPA) HydroNet site.

Flow data for the River Liffey at Leixlip Power Station was provided by the Electricity Supply Board (ESB) and was available for the year 2015 only.

Figure 4.1 shows the locations of the gauging stations used.

The gauging station for the Liffey at the Leixlip Power Station was located at a position some distance upstream from the tidal limit at Islandbridge Weir. These values were therefore scaled based on the size of the catchment between Leixlip and Islandbridge Weir (see Section 5.4.5). For other rivers, no allowances have been made for any additional run-off between the gauging stations and the receiving waters.

Many additional ungauged flows representing smaller rivers, streams and canals were also included in the hydrodynamic model. The specification of all freshwater sources in the model is described in Section 6.3.1.

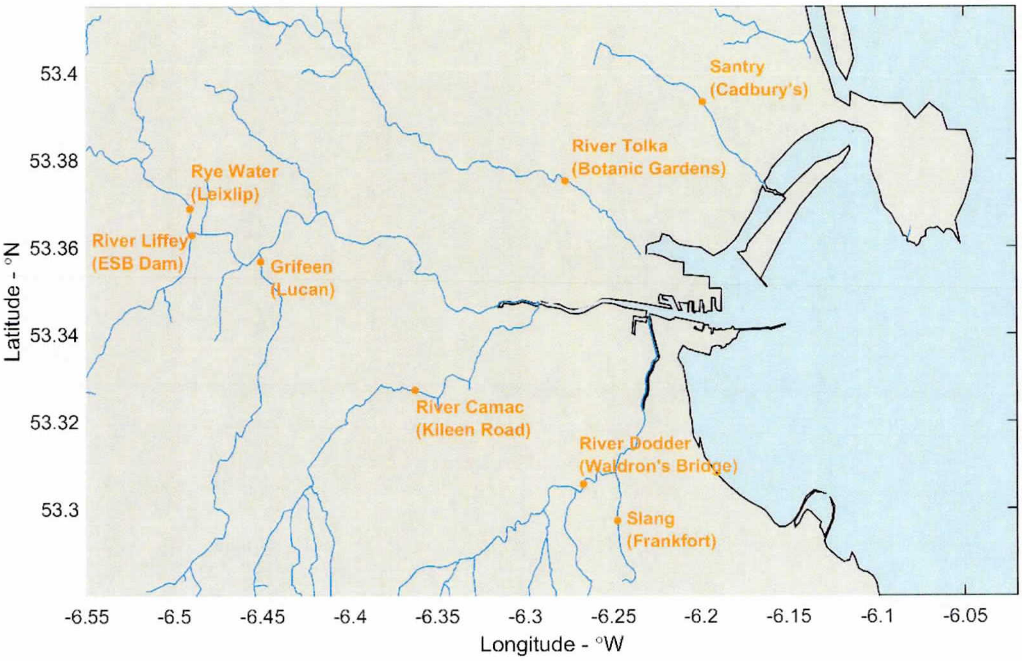

Figure 4.1 Map showing location of rivers and their gauging stations in and around Dublin.

4.1.2 Water Levels

Information on Water level in the Lower Liffey Estuary were available from two tide gauges:

- Dublin Port Tide Gauge (obtained from the Marine Institute, data.marine.ie); and
- Ringsend Tide gauge (provided by Dublin City Council).

Figure 4.2 shows the location of the two tide gauges and Figure 4.3 shows a time-series of water levels from the two gauges during the model calibration period (September – October 2015).

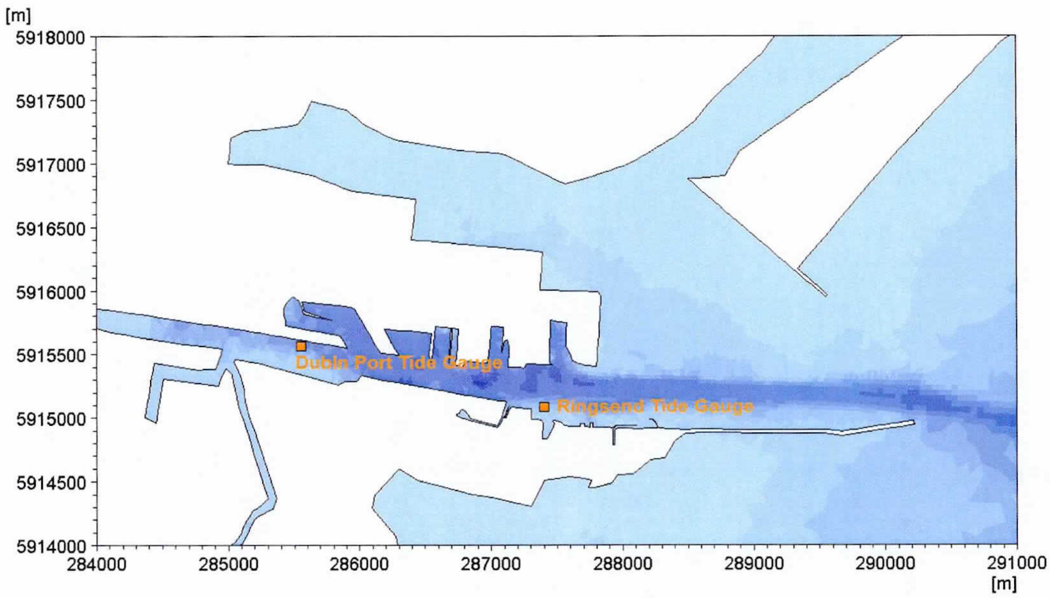

Figure 4.2 Map showing location of tide gauges on Lower Liffey Estuary.

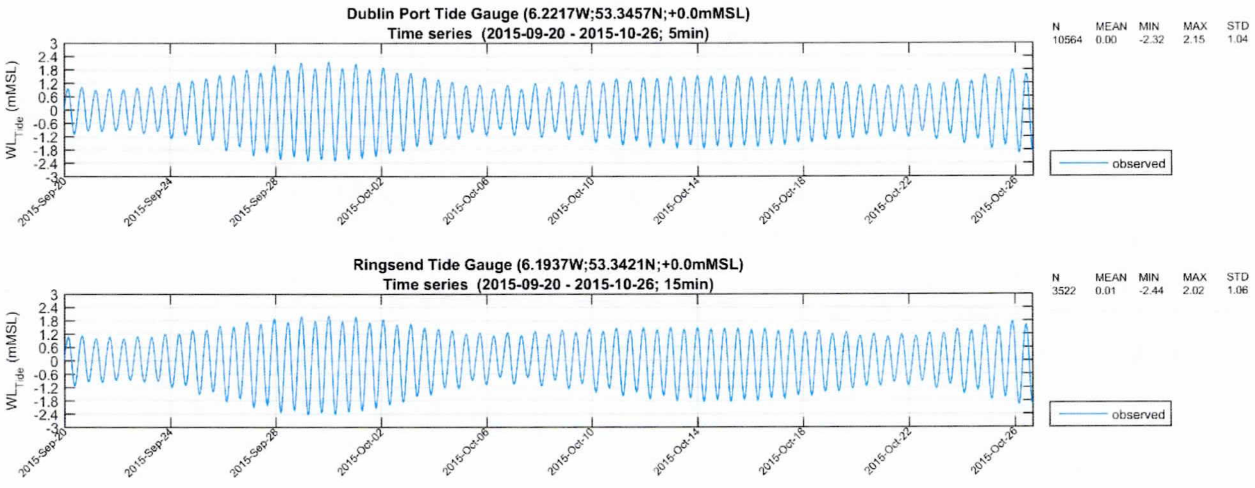

Figure 4.3 Time series of water levels recorded at gauges on Lower Liffey Estuary during September – October 2015.

4.1.3 Currents

Acoustic Doppler current profilers (ADCPs) are a commonly used instrument for measuring water current velocities. An ADCP emits pulses of sound which are scattered off particles suspended in the water column. The current velocity is then estimated using the principle of the Doppler Effect of echoed sound waves.

ADCPs are typically deployed on the bottom of a river or on the seafloor and measure the flow speed and direction at regular intervals (bins) through the water column. Alternatively, an ADCP can be mounted off the side of a moving vessel to measure the spatial variation in current speed along the vessel route. However, it should be noted that data for bins adjacent to the free surface (for a bottom-mounted ADCP) or the seafloor/river bed (for a vessel mounted ADCP) must be discarded as these data are contaminated by reflection off that surface, so called side-lobe interference.

Information from several ADCP surveys were used for the present study to ensure a suitable calibration of the model and to develop the conceptual understanding of flow in the estuary and the bay. These included:

- 2015 seabed mounted survey of Liffey Estuary, Tolka Estuary and Dublin Bay performed by Aquafact International Surveys (Ref. /6/);
- 2013 seabed mounted survey of Dublin Port (provided by RPS, from the Alexandra Basin EIS study, see Ref. /7/);
- 2010 seabed mounted survey at Burford Bank in Outer Dublin Bay performed by DHI (Ref. /8/); and
- 2009 vessel-mounted survey of Dublin Bay performed by DHI (Ref. /9/).

2015 survey of estuaries and Dublin Bay

Information on current speeds and directions through the water column within Dublin Bay, the Lower Liffey Estuary and the Tolka Estuary were recorded for this study during the Autumn of 2015. These data were collected by means of seabed mounted acoustic current speed profilers and Conductivity Temperature Depth (CTD) dips for currents and other relevant water parameters (see Section 4.1.4).

Directional data from the CTD current profilers was constrained to surface measurements due to problems with the compass at depth.

The ADCP current profilers were set to record the current speeds through the water column from 1m above the seabed to a point below the sea surface (approximately 1-2m below the surface) at intervals of 1m. A sensor on the device also recorded the water temperature 1m above the seabed. For more information on the survey methodology, see Ref. /6/.

Table 4.1 and Figure 4.4 show the location of the three devices deployed as part of the survey. It was noted that there was uncertainty on the location of these devices following the completion of the survey. The approximate locations provided were based on the surveyors best estimate of the position. All surveys covered at least one full spring-neap tidal cycle.

It was noted that the survey data contained high-frequency variations in current speeds and particularly current direction at temporal scales that cannot be resolved by hydrodynamic models. To improve their usability the survey data were therefore smoothed by applying a 30-minute moving average filter to enable comparison to the model predictions.

There were also noted issues with respect to the direction of the current which may have been associated with the positioning of the devices within the estuary - which was performed in consultation with the harbour master to ensure safety of navigation. As such, DHI have concerns regarding the suitability of these data for quantitative assessment of model performance, particularly for direction. However, as there are limited studies on the actual three-dimensional circulation of water in the harbour in the public domain these data provide the most up-to-date recordings and shall be used for a qualitative model assessment.

Figure 4.5 shows time-series' of the depth-averaged current speed at the three ADCP locations. The fastest current speeds were at the shallowest location, ADCP 3 (Clontarf), where mean depth-averaged currents were 0.21 m/s. The current speeds at ADCP 1 and ADCP 2 were low with mean values of 0.11 m/s and 0.12 m/s, respectively. These relatively low current speeds suggest that the overall circulation of water in the estuary can also be impacted by factors other than the tide.

The distribution of current speeds and directions at the three ADCP locations is shown in Figure 4.6. The strongest and most frequent currents at ADCP 1 (Liffey) flow towards the southeast, which suggests that the Tolka has an impact on flow at this location. The asymmetry also suggests that something other than tide controls the currents in this location.

Figure 4.7 shows the breakdown for the Liffey into total, tidal and residual components of the current speed and direction. This confirms the concept that there is more than tide alone controlling the flows. In addition, Figure 4.8 shows that there is a significant difference in speed between the near surface layers and the near seabed layers. The changed distribution of the currents over the vertical is likely to be the results of a density stratification. Runoff from the rivers will be focussed near the surface, while the denser saline water in the sea penetrates along the bed.

The currents at ADCP 3 (Clontarf) show more bi-directional flows which suggests a tidal influence on the flow regime within the Tolka Estuary (Figure 4.6). The flow direction at ADCP3 is also likely to be dictated by its location near a sharp (near 90 degree) corner of the Dublin Port as can be seen from Figure 4.4.

In Dublin Bay, the measurements from ADCP 2 measurements suggest a net flow to the north for the period surveyed (Figure 4.6).

Vertical profiles of the measured current speeds and directions are shown for the complex Liffey (ADCP 1) location in Figure 4.10.

Noticeable in both current speed and current direction at ADCP 1 (Liffey) is the vertical variability (Figure 4.9 and Figure 4.10). Higher current speeds are also seen to occur at the same time as strong wind speeds. For example, on the 23rd September 2015 and the 20th October 2015, strong westerly winds appear to lead to easterly flow at the surface. This data also shows that the pattern of current direction at the surface is one of dominant easterly flow with some reversals during easterly winds. At depth the current direction is a more bi-directional. This is critical to the understanding of the overall circulation in the estuary.

Figure 4.11 shows the variability between the surface measurements from the CTD dips and the near-surface ADCP1 (Liffey), which are in near proximity. The current speeds are shown to be generally comparable between these two measurements. Directions show less strong correlation, particularly during low tide periods. Whilst the ADCP shows a large fluctuation in directions, the CTD maintains a dominant outward surface flow. It is considered that this is likely to be related to the fine balance between the two driving mechanisms as well as the relative difference in the measuring devices.

Of note in the snapshot presented in Figure 4.11, is that the large variability in the directions measured by the ADCP occur during the flood tide period. During ebb tide, the ADCP measurements show a more invariant direction. It is likely that this is caused by the rapid spatial variability in current directions and also the fine balance between the tidal forces at depth and the surface waters, which in addition to the tide have wind forcing, freshwater flow and the effect of maritime traffic.

The water temperature near the seabed measured by the ADCP during the survey deployments is shown in Figure 4.12.

Table 4.1 Location of seabed mounted acoustic profilers.

Location	Easting [m UTM30]	Northing [m UTM30]	Max depth [m]	Survey Dates
ADCP1 – Liffey	288425*	5915085*	8.5	23rd September – 27th October, 2015
ADCP2 – Dublin Bay	291951♦	5915736♦	9.5	23rd September – 22nd October, 2015
ADCP3 - Clontarf	287887♦	5915965♦	4.7	07th October – 22nd October, 2015

*Estimated position provided by surveyors at completion of survey.

♦Position provided by surveyors following deployment.

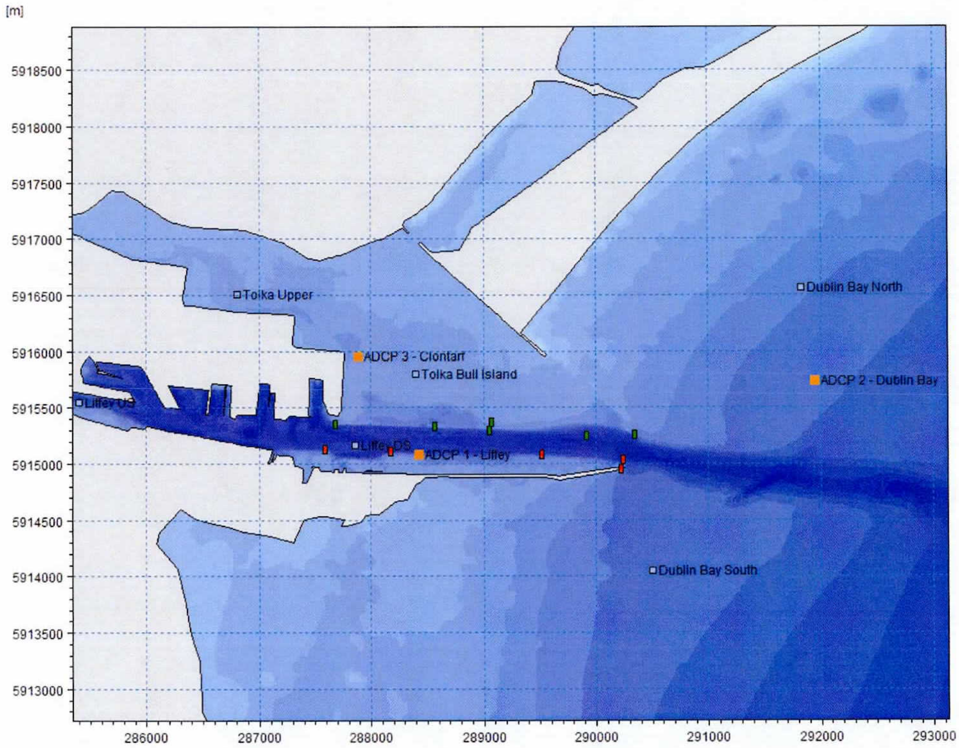

Figure 4.4 Map showing location of seabed mounted acoustic profilers (orange markers) and CTD locations (blue markers) as described in Section 4.1.4 with selected channel marker buoys.

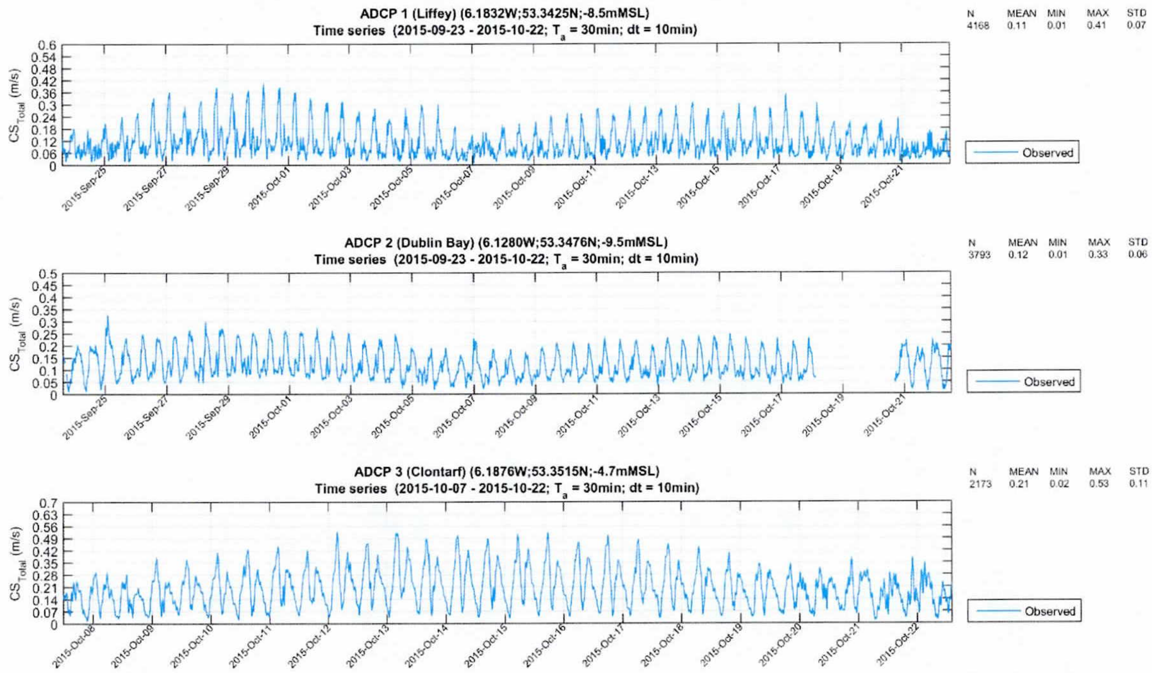

Figure 4.5 Time series of depth-averaged current speed at ADCP 1 – Liffey (top panel), ADCP 2 – Dublin Bay (central panel), and ADCP 3 – Clontarf (lower panel).

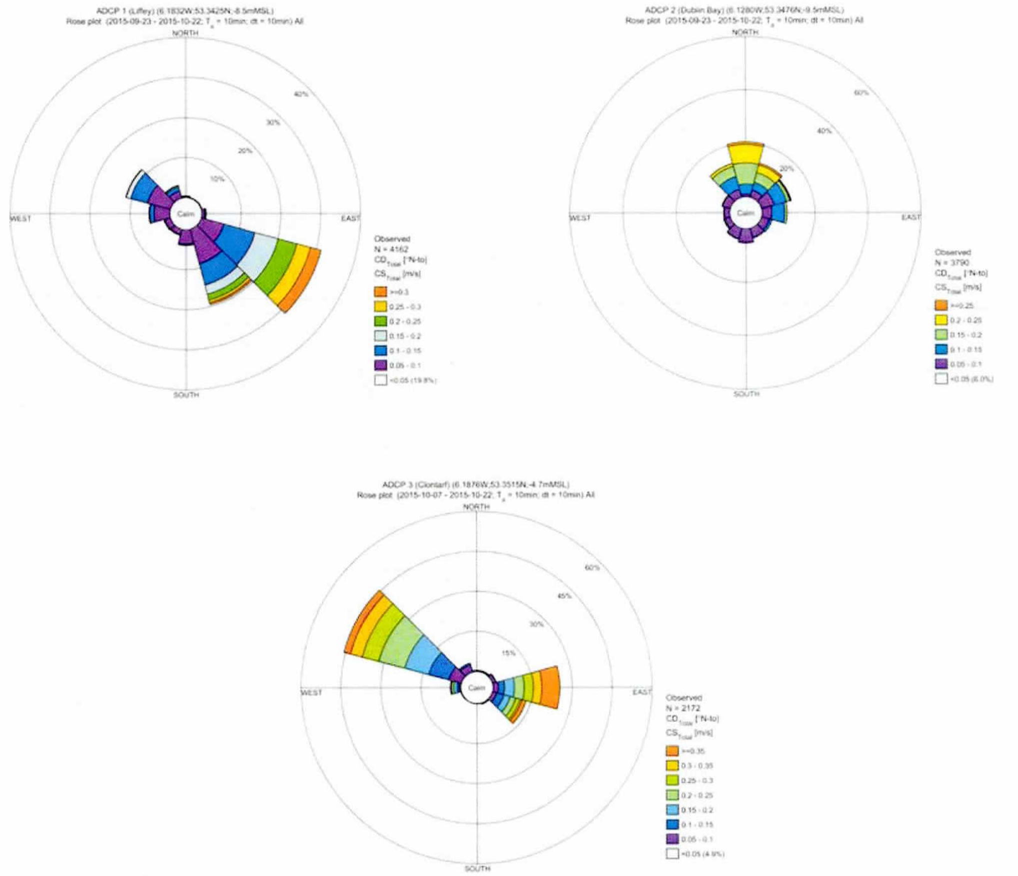

Figure 4.6 Depth averaged current speed rose plot at ADCP 1 – Liffey (top, left), ADCP 2 – Dublin Bay (top, right), and ADCP 3 – Clontarf (bottom). The sectors show the direction towards which the current is flowing.

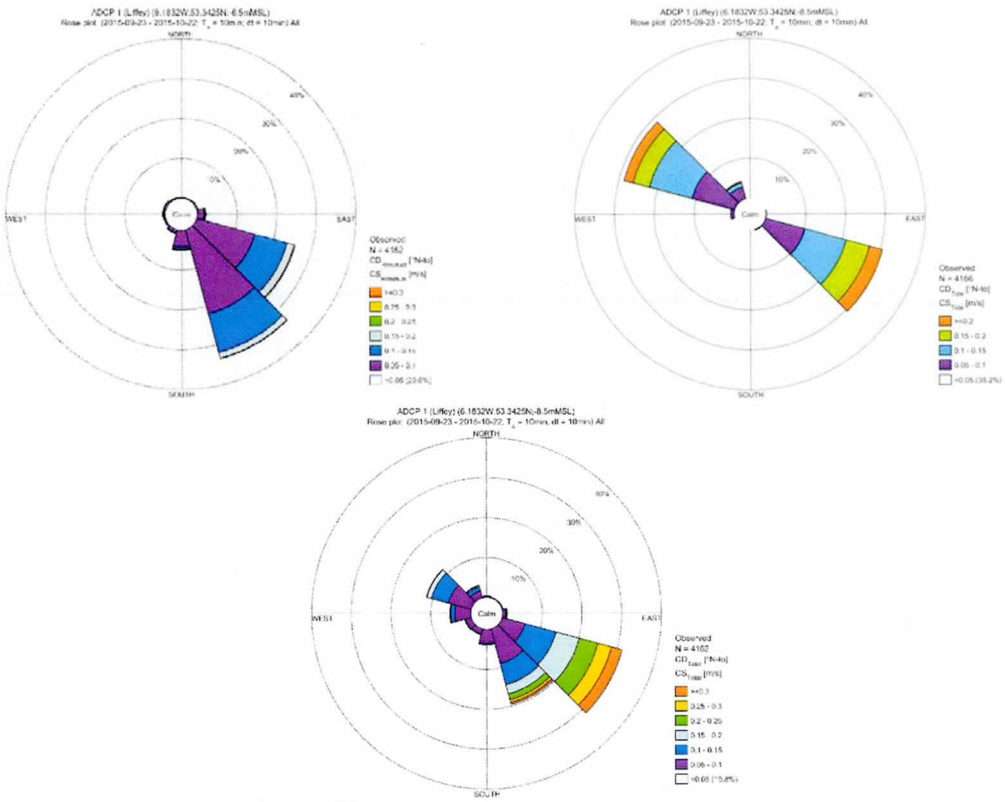

Figure 4.7 Detailed split of depth averaged current speed rose plot at ADCP 1 – Liffey broken down into component parts – Residual (top left), Tide only (top right) and Total (bottom). The sectors show the direction towards which the current is flowing.

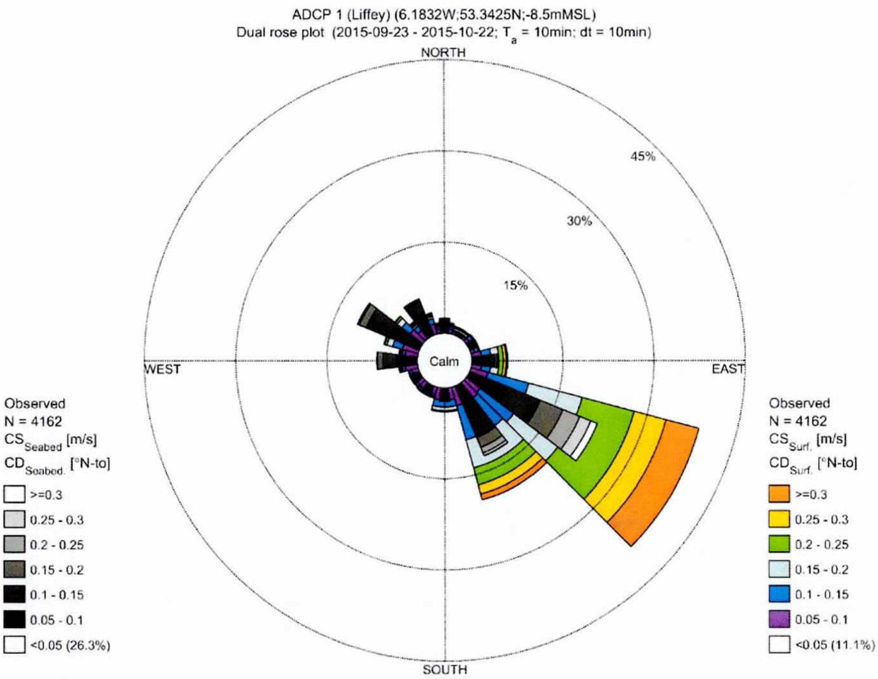

Figure 4.8 Near surface and near seabed current speed and direction at ADCP 1 – Liffey.

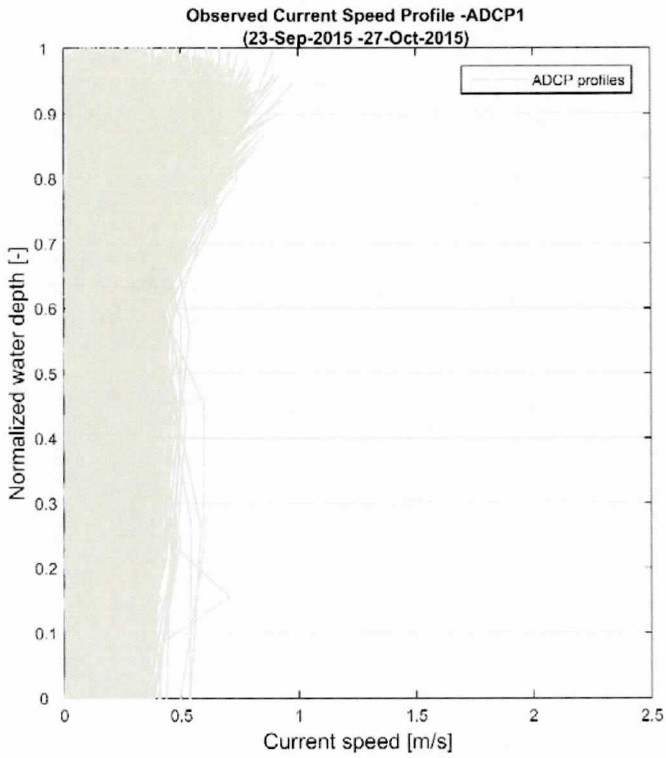

Figure 4.9 Measured current speed profiles at ADCP 1 – Liffey for the entire deployment.

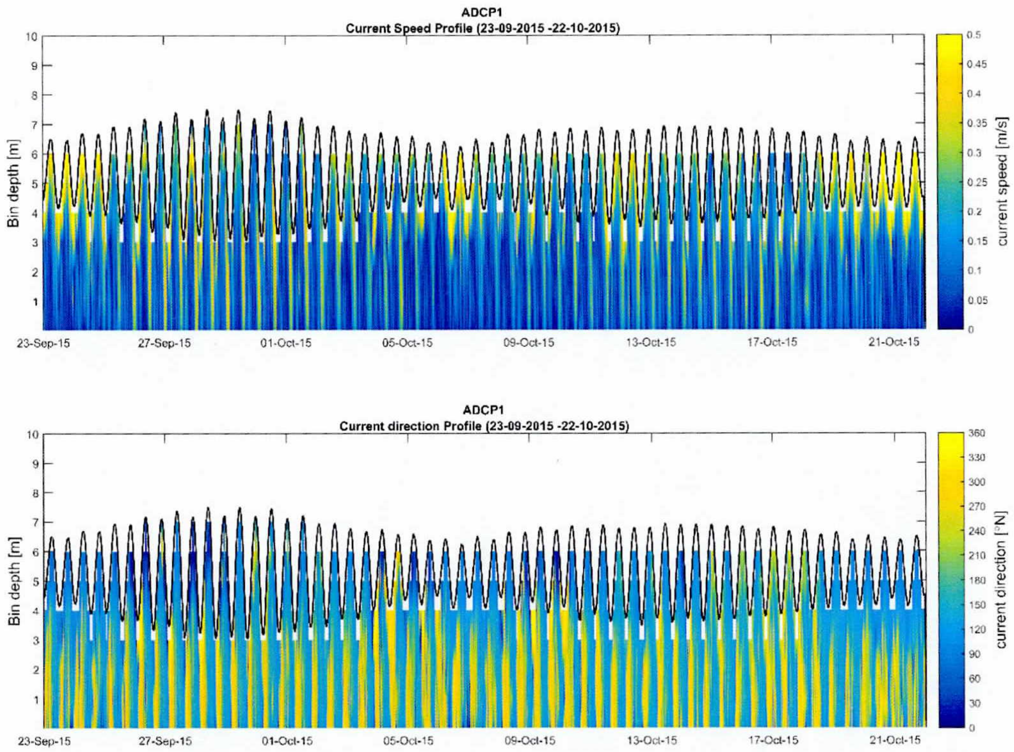

Figure 4.10 Profiles of current speed (top) and current direction (bottom) at ADCP 1 – Liffey. Note blanking of surface layer in measurements.

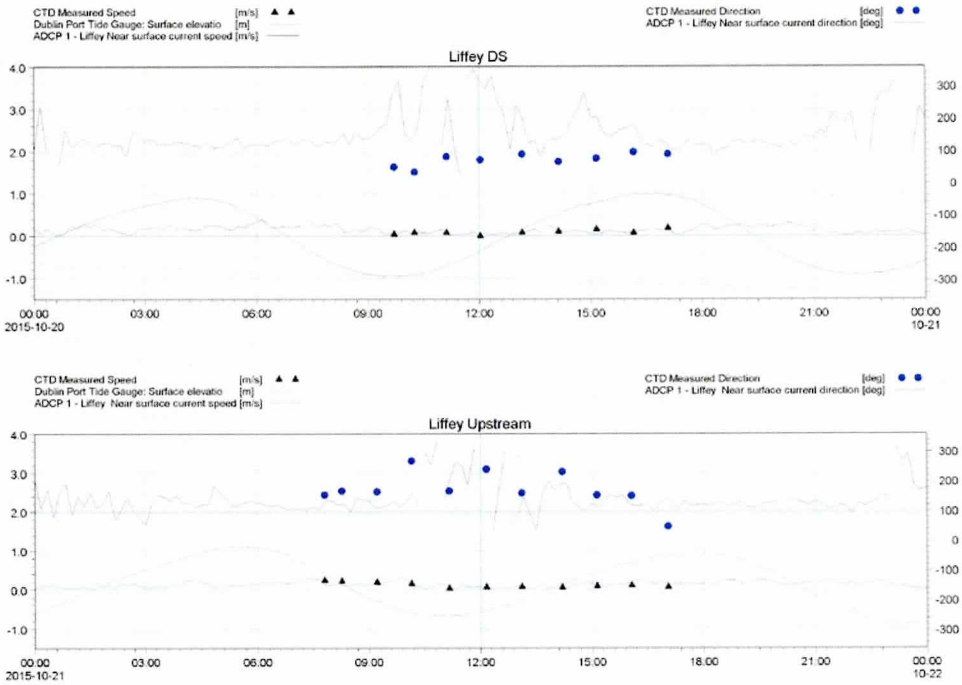

Figure 4.11 Comparison of ADCP 1 –Liffey with CTD measurement data for the surface layer at Liffey Downstream and Liffey Upstream CTD locations.

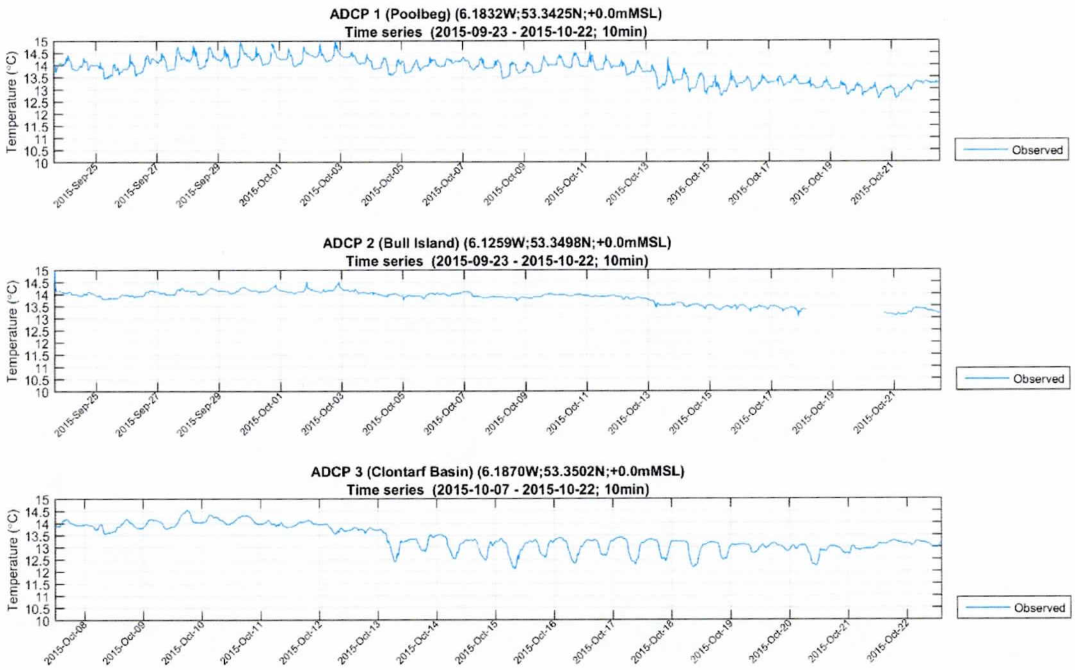

Figure 4.12 Time series of seabed temperature at ADCP 1 – Liffey (top panel), ADCP 2 – Dublin Bay (central panel), and ADCP 3 – Clontarf (lower panel).

2013 survey of Dublin Port

Data was provided from the Alexandra Basin redevelopment project from Dublin Port, via RPS see Ref. /7/. This data point was located upstream of the Ringsend outfall as shown by position 3 in Figure 4.13.

The Dublin Port data was provided for layers through the water column, which were also processed to depth averaged values as per the 2015 survey data to enable comparisons.

Figure 4.14 shows the distribution of current speed and direction for the near surface and near seabed layers for the Dublin Port ADCP.

It was noticeable that the surface flow is more concentrated in an easterly direction than the 2015 survey for ADCP 1 (Liffey) - even though they are separated by only a few hundred metres. This suggests that the influence of the Tolka, on the flows in this lower part of the estuary and that the balance of flow is again in an easterly direction at the surface, is limiting surface flow into the Liffey.

At the seabed, the flows are more balanced with respect to duration of flow in each direction, however ebb current speeds are higher than flood speeds.

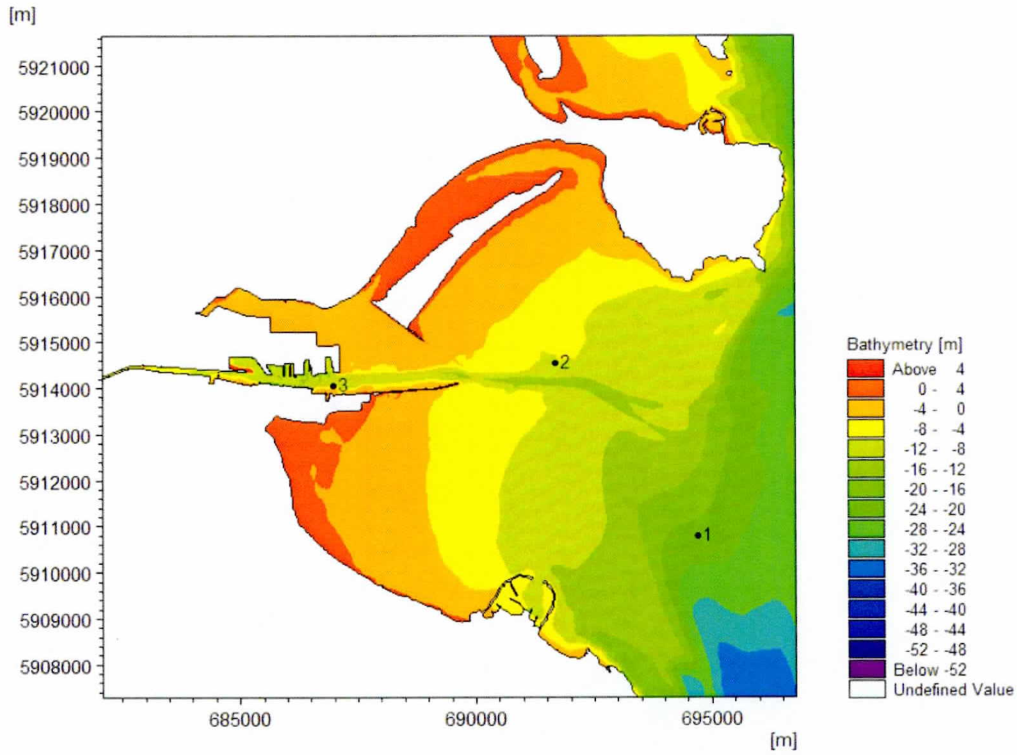

Figure 4.13 ADCP current measurement locations for the Dublin Port Alexandra Basin Redevelopment Project EIS. Location 1 is the DHI ADCP1 at Burford Bank detailed in the following sections. Location 3 is located on the Lower Liffey, upstream of the Ringsend WwTP outfall. Location 2 data was not made available to this project.

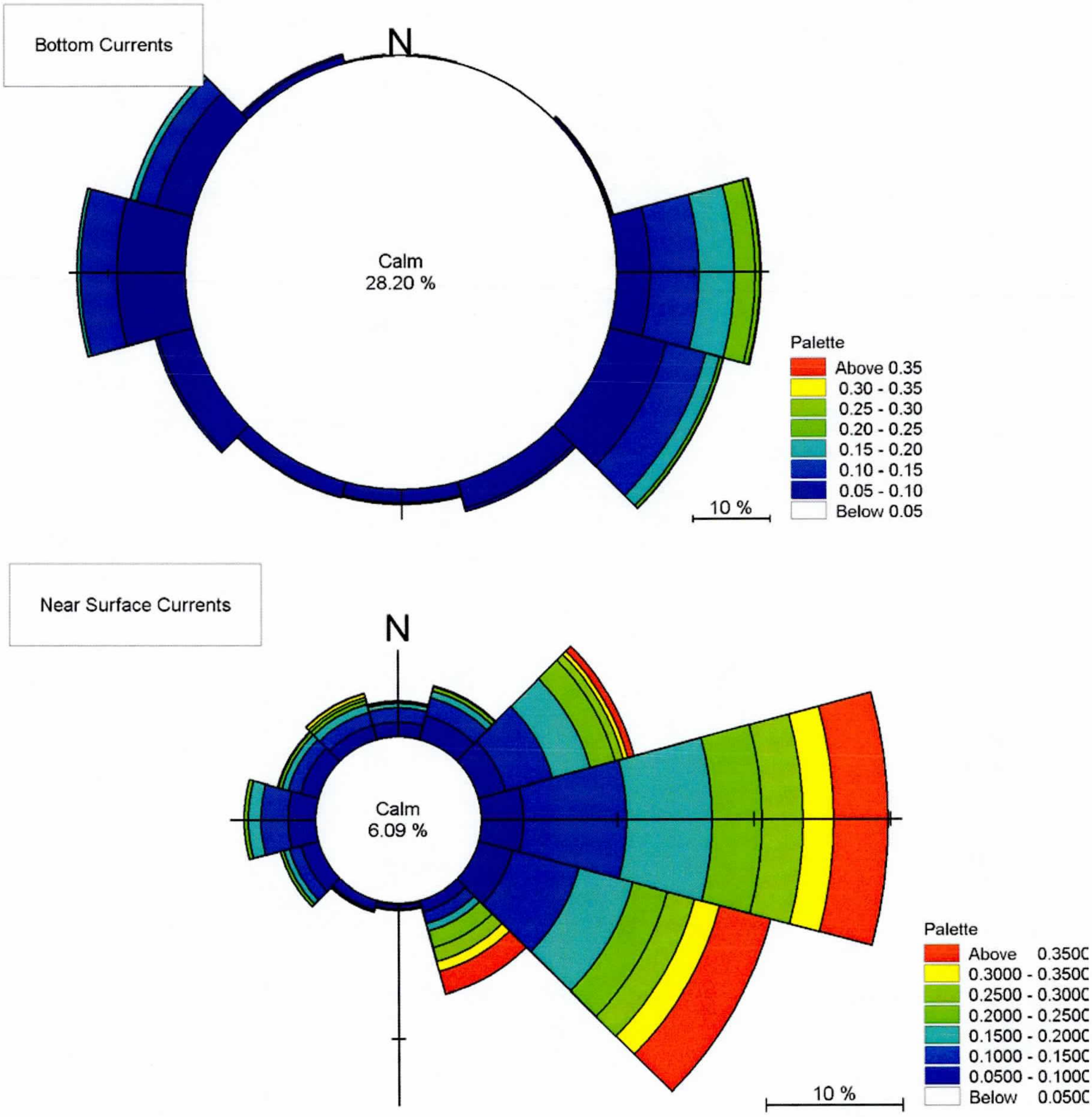

Figure 4.14 Bottom and surface current speed rose plot at RPS ADCP location from 2013. The sectors show the direction towards which the current is flowing (current speed m/s).

2010 survey of Burford Bank and Outer Dublin Bay

DHI previously conducted current speed measurement campaigns in Outer Dublin Bay to support the calibration of numerical models for the Ringsend Long Sea Outfall project (Ref. /8/). This included the deployment of two bottom-mounted ADCP's during April and May 2010. One ADCP was deployed either side of Burford Bank, located at the outer limit of Dublin Bay (Figure 4.15).

Both stations were deployed for 30-days and recorded data that was judged to be of excellent quality. Depth-averaged current speeds at both ADCP locations were available to the project team for the present study. These data were used to validate the hydrodynamic model in the outer part of Dublin Bay.

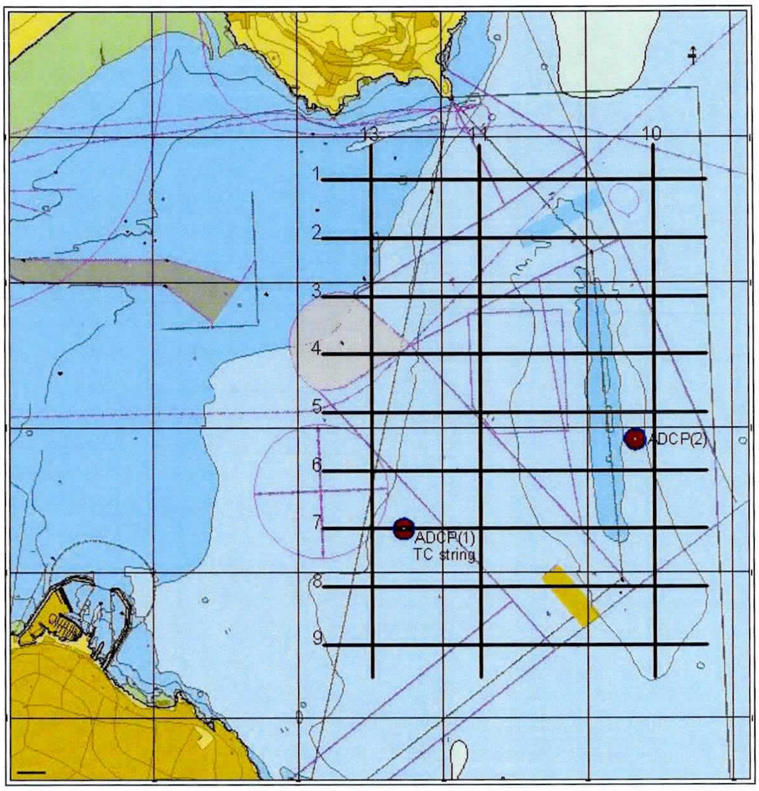

Figure 4.15 Map of Dublin Bay showing location of two DHI ADCP's deployed as part of the Ringsend Long Sea Outfall Study during April-May 2010 (after Ref. /8/).

2009 vessel-mounted survey of Dublin Bay

DHI performed a series of moving vessel ADCP surveys within Dublin Bay during the period of the 8th-10th of July 2009 (Ref. /9/). Figure 4.16 shows the route of the individual tracks which included sections across the entrance to Dublin Port as well as locations further offshore over Burford Bank. Depth-averaged current speeds were available from these surveys and were used to validate the hydrodynamic model in Dublin Bay.

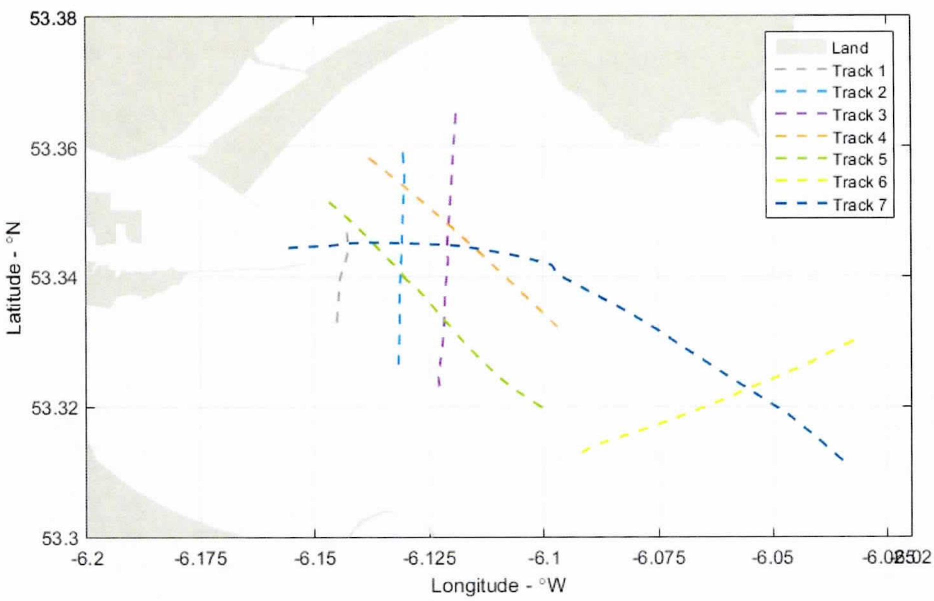

Figure 4.16 ADCP transect surveys routes in Dublin Bay recorded between the 8th and 10th July 2009.

4.1.4 Temperature and Salinity Data

2015 CTD surveys of estuaries and Dublin Bay

Measurements of Conductivity, Temperature and Depth (CTD) were performed at 6 locations within Dublin Bay and the Liffey and Tolka estuaries (see Ref. /6/). The surveys were performed between the 19th and 23rd October 2015, coinciding with the deployment of the current profilers (see Section 4.1.3).

Table 4.2 and Figure 4.26 show the location of the CTD survey stations.

It is noted (see Ref. /6/) that no data were available for Dublin Bay North due to the onset of adverse weather conditions during the survey.

At each location, conductivity (salinity) and temperature were recorded every hour during a complete semi-diurnal tidal cycle.

Observations were recorded at three depths (near-surface, mid-water, and near-seabed). However, due to shallow water at the Upper Tolka location, only near-surface and near-seabed were available.

The salinity observations for Liffey Upstream (US), Liffey Downstream (DS), Dublin Bay South and Tolka Bull Island are shown in Figure 4.18. It was noted that some of the salinity observations in the Liffey DS site were lower than expected, with Practical Salinity Units (PSU) lower than 20. This was especially true for observations near the seabed, where the influence of freshwater is not expected to be significant. These observations were considered outliers and were treated with caution during the model calibration.

The temperature observations for Liffey US, Liffey DS, Dublin Bay South and Tolka Bull Island are shown in Figure 4.19. The recorded temperatures were lower than expected. Comparing the data against the temperature recorded by the current profilers (Figure 4.12), the CTD temperatures were, on average, 3 °C lower. The source of this discrepancy is not known; however it is possible that the CTD reading were taken before stable temperature conditions could be achieved. In the Aquafact report (Ref. /6/) it is noted that there were two outliers in the data for this location. The temperature readings from the CTD were therefore excluded from the model calibration exercise.

The data in Figure 4.18 and Figure 4.19 show that the Dublin Bay South and Liffey DS, and Tolka Bull Island locations are well mixed. However, for the Liffey US site, there exists evidence of stratified flow with salinities at the surface lower than at the seabed due to the influence of freshwater.

Table 4.2 Location of CTD observation stations.

Location	Easting [m UTM30]	Northing [m UTM30]	Max depth [m]
Liffey US	285400	5915550	5.7
Liffey DS	287860	5915165	10.5
Dublin Bay North	291830	5916570	-
Dublin Bay South	290500	5914050	7.5
Tolka Upper	286805	5916510	2
Tolka Bull Island	288400	5915800	3.3

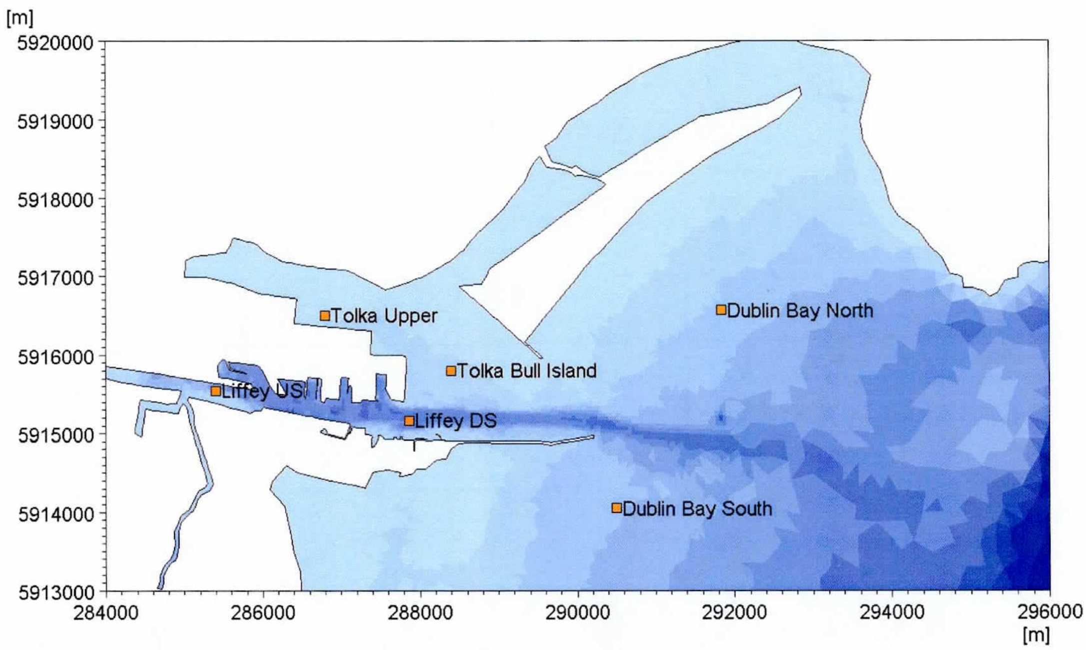

Figure 4.17 Map showing location of CTD observation stations.

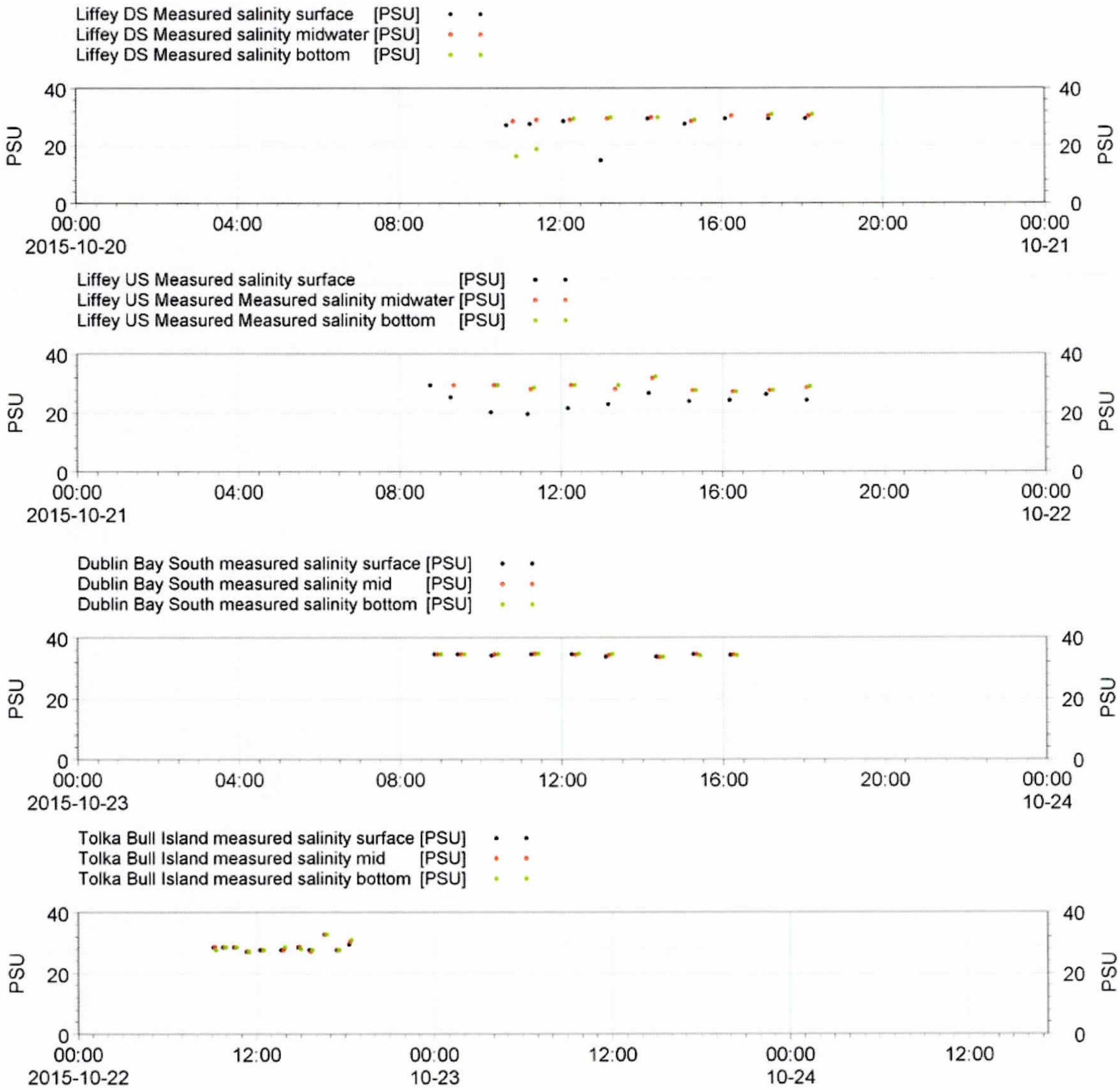

Figure 4.18 Salinity observations from CTD surveys in Dublin Bay and estuaries.

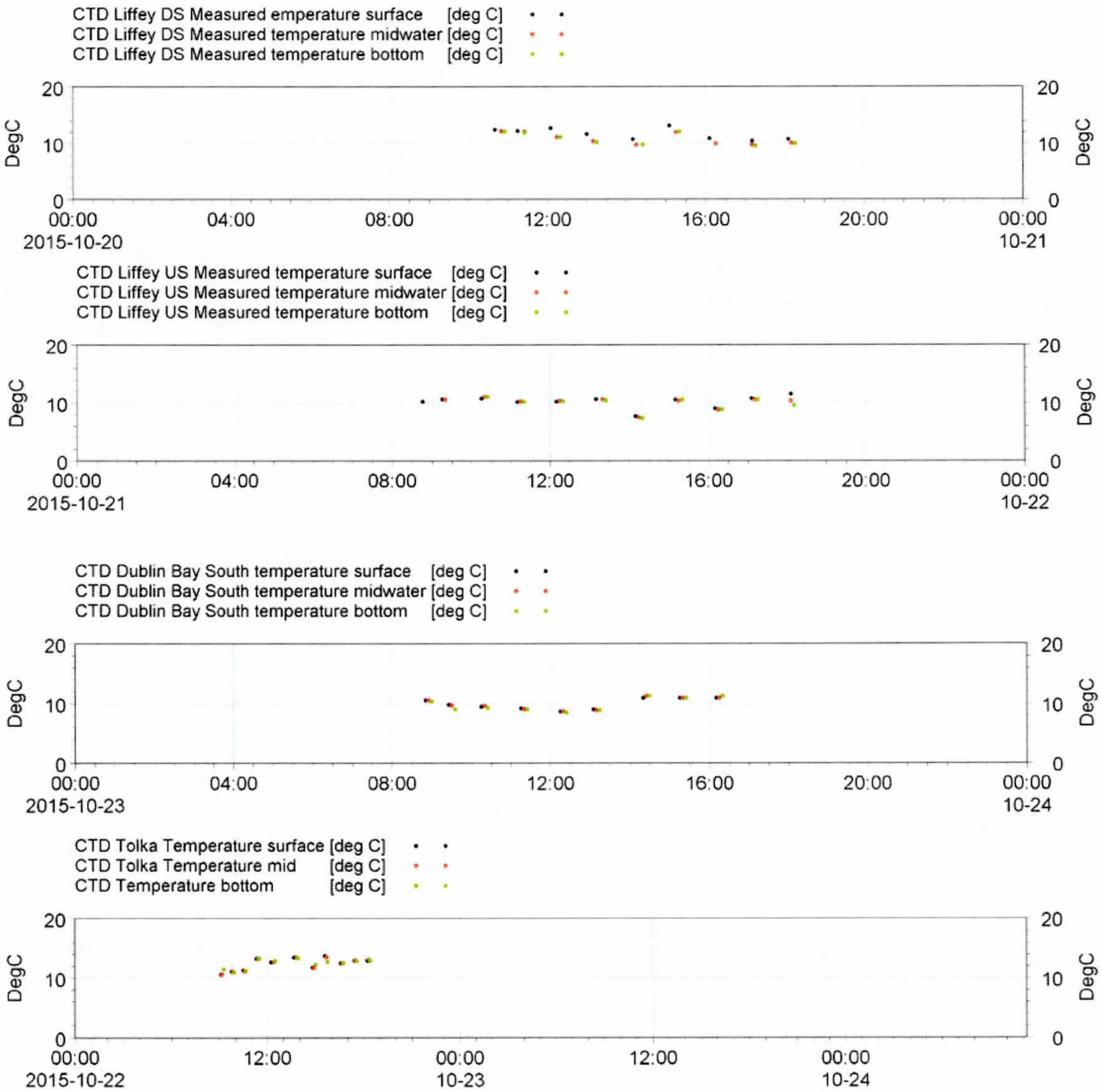

Figure 4.19 Temperature observations from CTD surveys in Dublin Bay and estuaries.

4.2 Other Data Sources

In addition to the main hydrodynamic and hydrology data, there was a requirement in the modelling for data on the variability of wind, air temperature, relative humidity and clearness on the model domain. These data were obtained from Dublin Airport. However, for the 2015 calibration period these were provided from existing regional climate models. Examples of these data are shown below.

4.2.1 Meteorology - Air Temperature, Relative Humidity and Clearness

Spatially varying conditions were used for the modelling of air temperature, relative humidity and clearness. The spatial grid that this have been provided on is shown below in relation to the coastline and as time series. The data was provided from existing StormGeo regional climate models. Data was available at 3-hour timesteps for all model run periods.

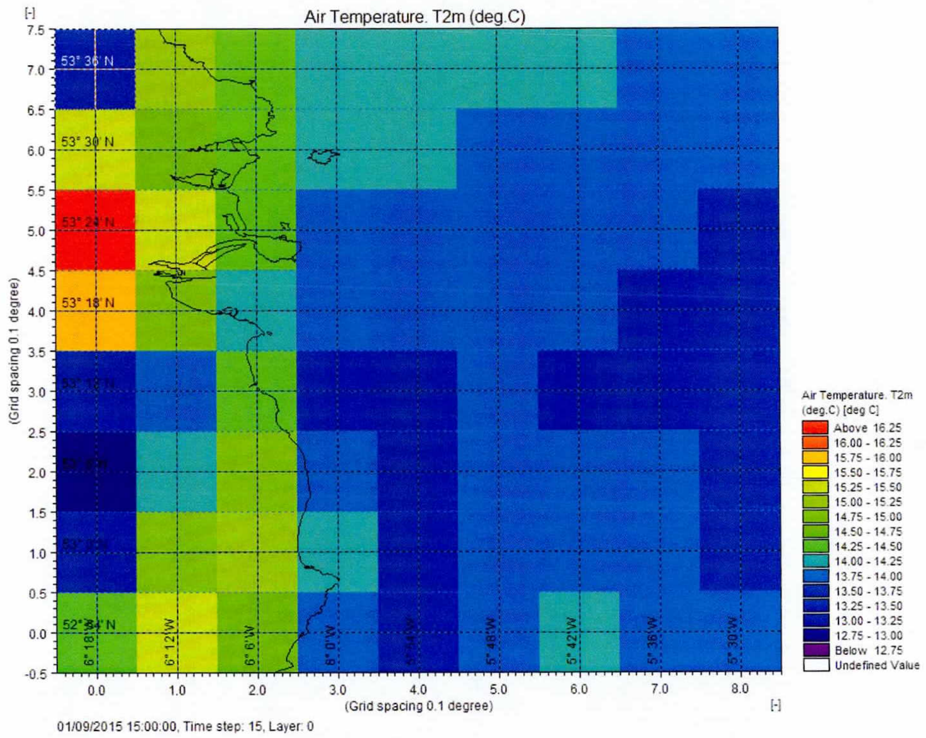

Figure 4.20 Example of meteorological data sets used in this study with the coastline of Ireland shown for reference.

4.2.2 Wind Data

Wind data was available from a range of sources including the StormGeo model, Dublin Airport (measured) and the Dublin Bay Smart Buoy (measured). Both measured data sets provided sub-hourly data for the period of calibration. Smart Buoy data was not available for the periods pre- October 2013. The location of the data is shown below.

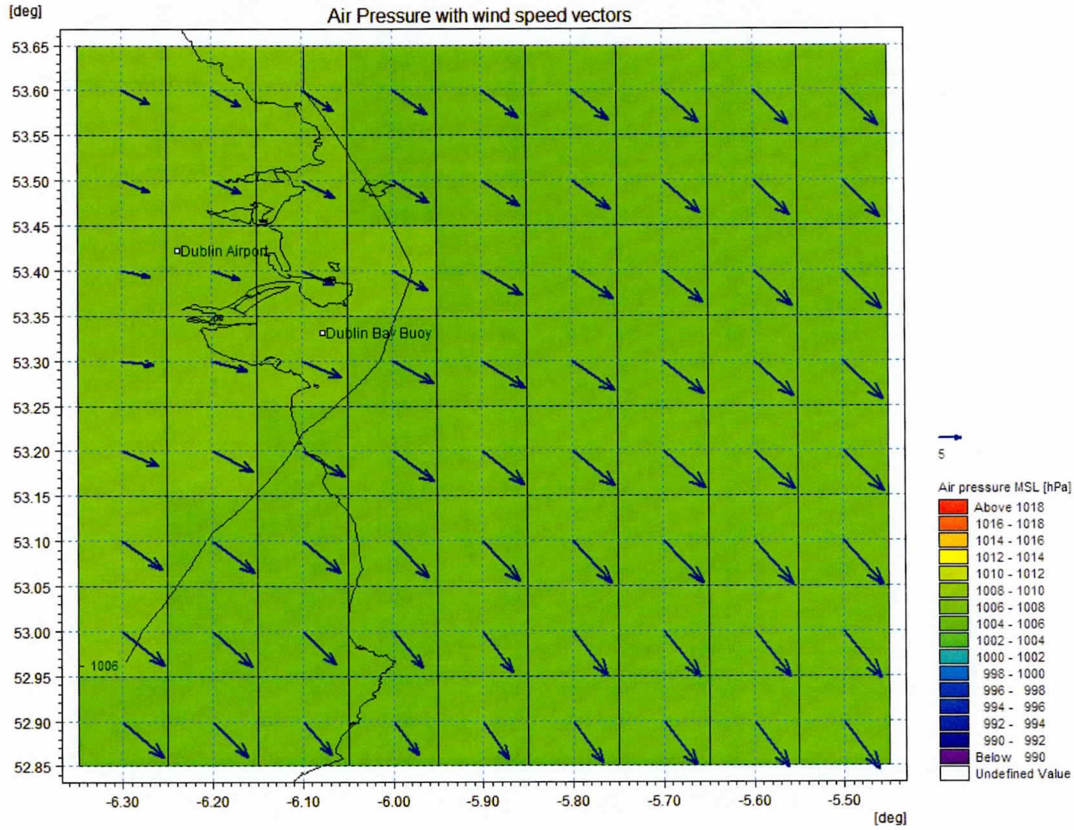

Figure 4.21 Example of wind model data (gridded air pressure and vectors wind magnitude) and measurement locations applied across the domain (points) with the coastline of Ireland shown for reference.

4.3 Water Quality Monitoring

Information on water quality monitoring was provided by Dublin City Council from their ongoing WFD monitoring regimes, at locations previously agreed with the Environmental Protection Agency (EPA).

Discrete water quality sampling has historically been performed in the Liffey and Tolka estuaries and within Dublin Bay. The frequency of this sampling varies, but are typical performed 4-6 times per year, and more commonly in the bathing water season (June - August).

These data were analysed to establish typical concentrations of pollutants over the baseline periods (2013 – 2015, inclusive). The purpose of this assessment was twofold.

1. To provide background loads/concentrations that enter the system via the rivers, streams, and canals; and
2. The observed data were used to validate the concentrations predicted by the water quality model at various locations in the harbour and in Dublin Bay.

4.3.1 Estuarine and Coastal Water Monitoring

Figure 4.22 shows the locations of estuarine and coastal water monitoring sampling sites. Information from these surveys included the concentrations of Ammonia, Biochemical oxygen demand (BOD), dissolved inorganic nitrogen (DIN), and molybdate reactive phosphorus (MRP) in addition to water temperature and salinity.

For the reported BOD information, there were many values that were equal to or below the lower limit of detection (LOD), typically 1 mg/L. Where concentrations fell below the detection limit, nominal values of half the detection limit were used.

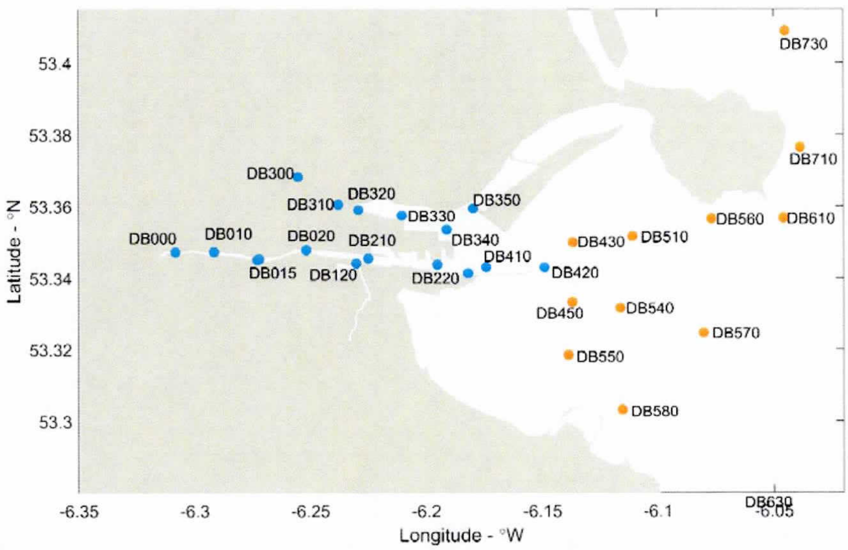

Figure 4.22 Map of Dublin Harbour and Dublin Bay showing locations of water quality monitoring stations. Blue dots show the monitoring points in the transitional waters. Orange dots show the monitoring locations in coastal waters.

MRP (transitional waters)

Figure 4.23 shows the variation in MRP at the surface in transitional waters during summer (2013 – 2015). This shows concentrations in the Liffey from DB010 to DB220 are fairly consistent with median values below 0.06 mg/L. At point DB410, downstream of the Ringsend WwTP outfall (SW1), the concentration of MRP show somewhat larger variations and larger median values of around 0.14 mg/L. The concentrations at the entrance to Dublin Harbour, point DB420, shows that concentrations are reduced as the pollutant disperses and mixes downstream of the WwTP.

In the Tolka Estuary, from DB300 – DB340, the concentrations of MRP are likely to be influenced by the dispersion from the WwTP and by riverine input from the Tolka. As a result, MRP concentrations in the Tolka Estuary appear higher than those in the Liffey from this monitoring period.

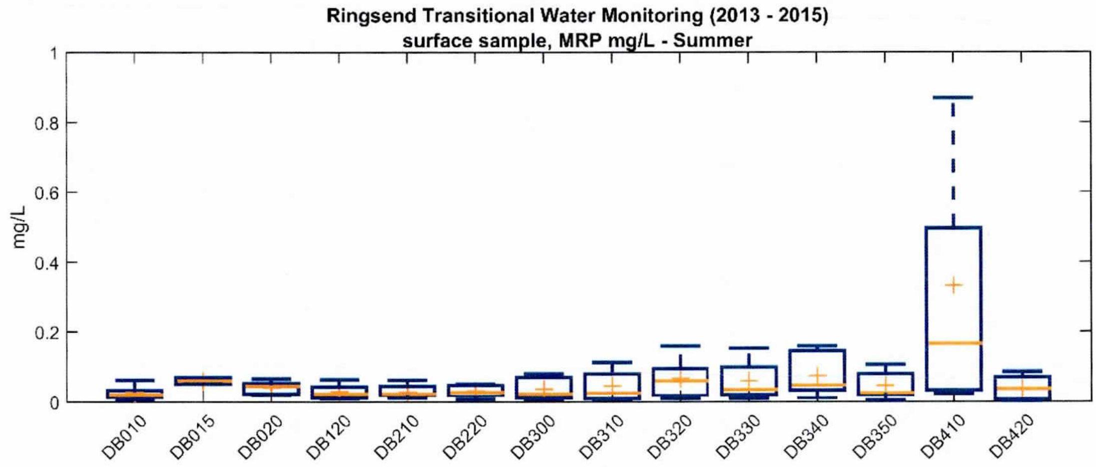

Figure 4.23 Concentration of MRP in the transitional waters (surface sample) during summer (2013 – 2015). Orange crosses shows the mean concentration and horizontal orange line shows the median concentration. The blue box shows the range of the range of the 25-75% quantile and whiskers show the range of the 10-90% quantile.

BOD (transitional water body)

Figure 4.24 shows the variation in BOD concentration in transitional waters during summer conditions (2013 – 2015). The concentrations of BOD at DB310 and DB320 show a larger variation which is likely due to riverine input from the Tolka.

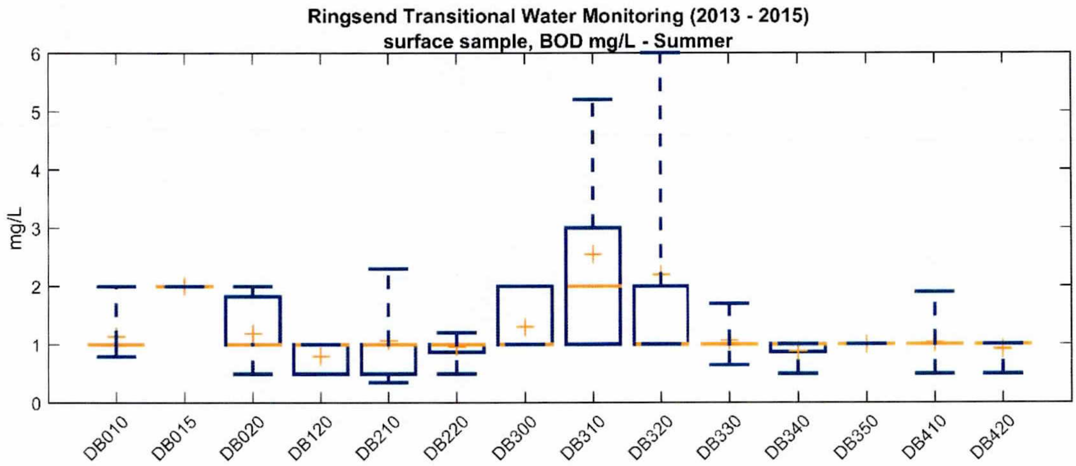

Figure 4.24 Concentration of BOD in the transitional waters (surface sample) during summer (2013 – 2015). Orange crosses shows the mean concentration and horizontal orange line shows the median concentration. The blue box shows the range of the range of the 25-75% quantile and whiskers show the range of the 10-90% quantile.

DIN (coastal water body)

Figure 4.25 shows the variation in DIN in the coastal waters of Dublin during summer from 2013 – 2015. The concentrations are consistent with median values generally lower than 0.05 mg/L. Median concentrations are larger at DB430, which is most likely due their proximity to the entrance of the harbour and hence plume emanating from the Liffey Estuary.

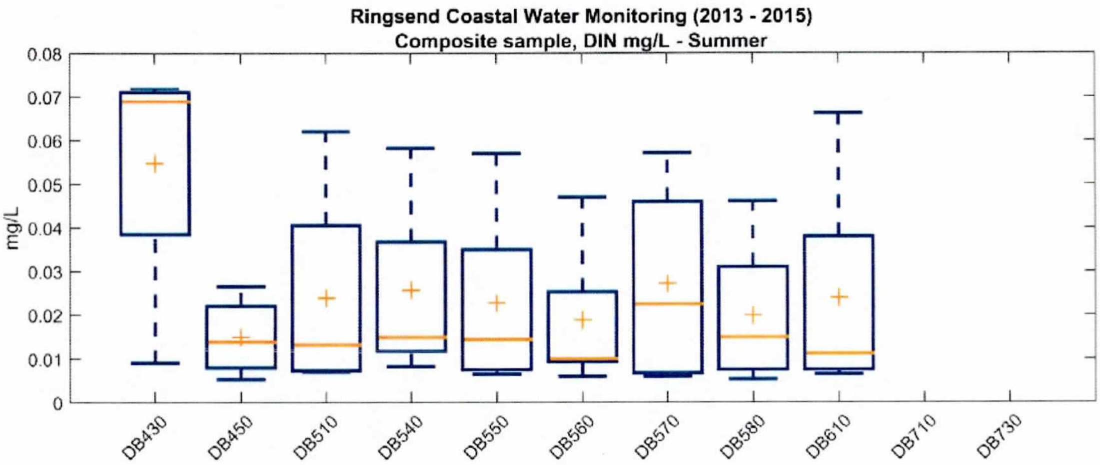

Figure 4.25 Concentration of DIN in coastal waters (composite sample) during summer. Orange crosses shows the mean concentration and horizontal orange line shows the median concentration. The blue box shows the range of the range of the 25-75% quantile and whiskers show the range of the 10-90% quantile.

4.3.2 E. coli (Rivers Monitoring)

River water sampling data for concentrations of E. coli were also available for the period 2013 – 2015. Discrete water sampling was generally performed once a month during the year. Figure 4.26 shows the location of the available sampling data and Figure 4.27 shows the range in the returned data.

E. coli concentrations were largest in the River Cammock and in the River Liffey at the location of the Cammock outfall near Heuston Station. However, at the location on the Liffey further downstream (point 40457, Toll Bridge) the concentration of E. coli were lower, on average.

E. coli concentrations were fairly consistent in the River Dodder, whereas the Tolka samples showed greater variability and larger mean concentrations. There was also a fairly large average E. coli concentration from the Trimleston stream, which discharges directly into Dublin Bay to the South of the harbour.

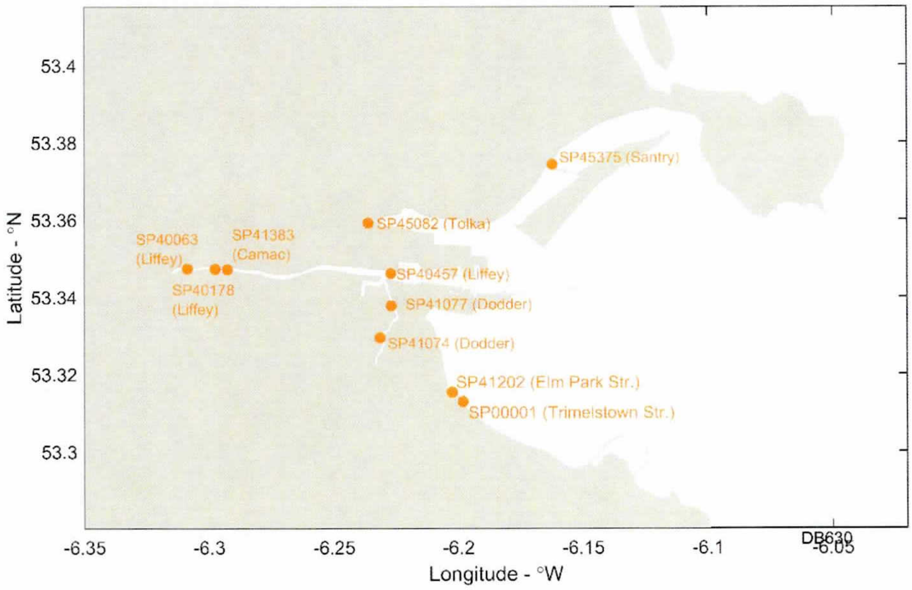

Figure 4.26 Map of Dublin Harbour and Dublin Bay showing locations of river water monitoring stations for E. coli.

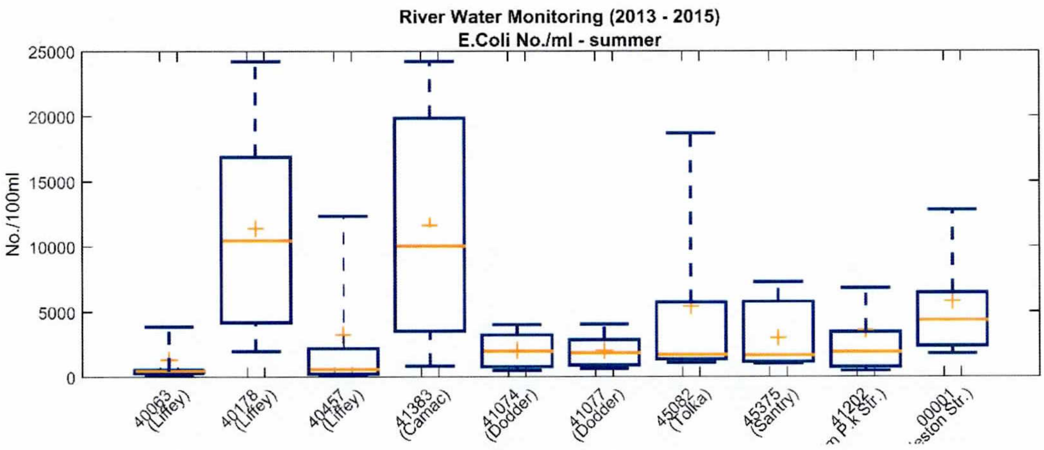

Figure 4.27 Concentration of E. coli at river water sampling sites during summer. Orange crosses shows the mean concentration and horizontal orange line shows the median concentration. The blue box shows the range of the range of the 25-75% quantile and whiskers show the range of the 10-90% quantile.

4.3.3 E. coli (Bathing Water Monitoring)

Coastal water sampling is performed to assess bathing water quality at 8 locations throughout the Dublin Bay (Figure 4.28) as part of the WFD Bathing Water assessment. Discrete water sampling is typically performed at least once per week (and sometimes more frequently) during the summer bathing season (June – September).

Information from the bathing water monitoring that is relevant to the present study is the concentration of Escherichia Coli (E. coli). These data were analysed to establish typical concentrations over the baseline period (2013 – 2015). Note that site ASW15 has been omitted from the analysis presented here, as this location near the Poolbeg outfall is not a designated bathing water site.

There is a high degree of variability in the concentration of E. coli at each site during the bathing water season. Figure 4.29 shows the range in these concentrations and that the mean value is often skewed. The highest concentrations were found at ASW18 (Merrion Strand) on the Southern side of Dublin Bay. It is thought that the high concentrations are due to discharge from a local water source discharging in the proximity of ASW18. Relatively high concentrations of E. coli were also found at ASW14 (Bull Wall Wood Causeway). This site is located within the harbour walls and is therefore more likely to be influenced by dispersion from the WwTP and riverine inputs. For all other sites, the median concentration of E. coli was less than 50 per 100 millilitres (Figure 4.29).

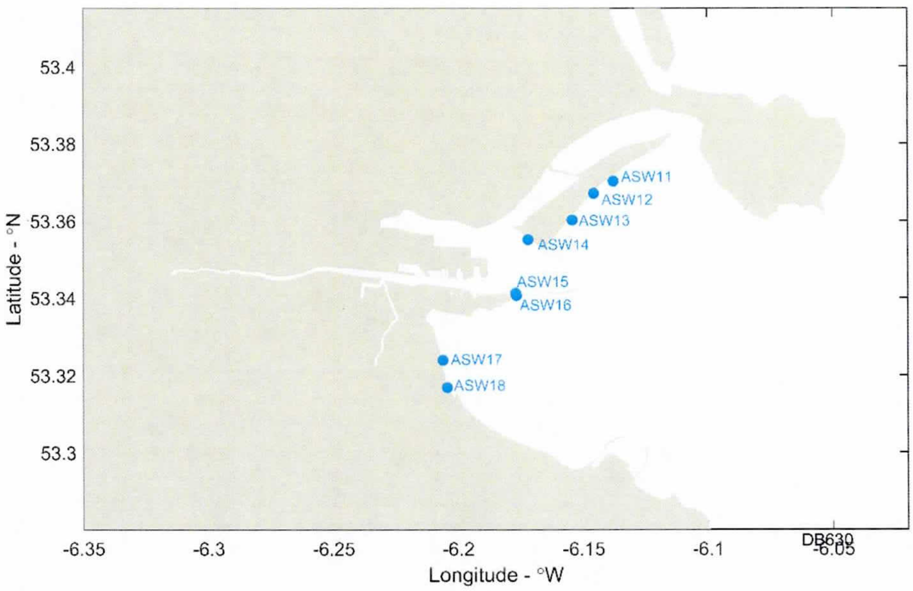

Figure 4.28 Map of Dublin Harbour and Dublin Bay showing locations of bathing water monitoring stations. Note that ASW15 and ASW16 are located on opposite sides of the Poolbeg Wall, with ASW15 on the inside of Dublin Harbour, downstream of the Ringsend WwTP.

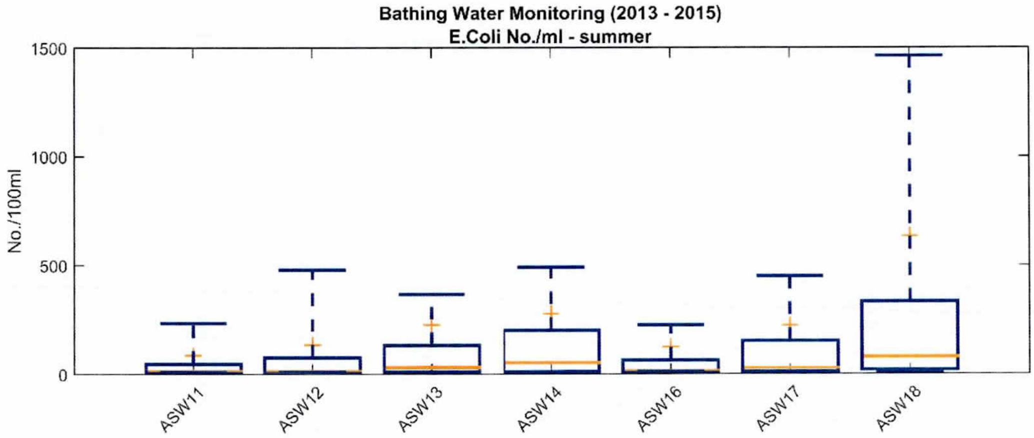

Figure 4.29 Concentration of E. coli at bathing water sites during summer. Orange crosses shows the mean concentration and horizontal orange line shows the median concentration. The blue box shows the range of the range of the 25-75% quantile and whiskers show the range of the 10-90% quantile.

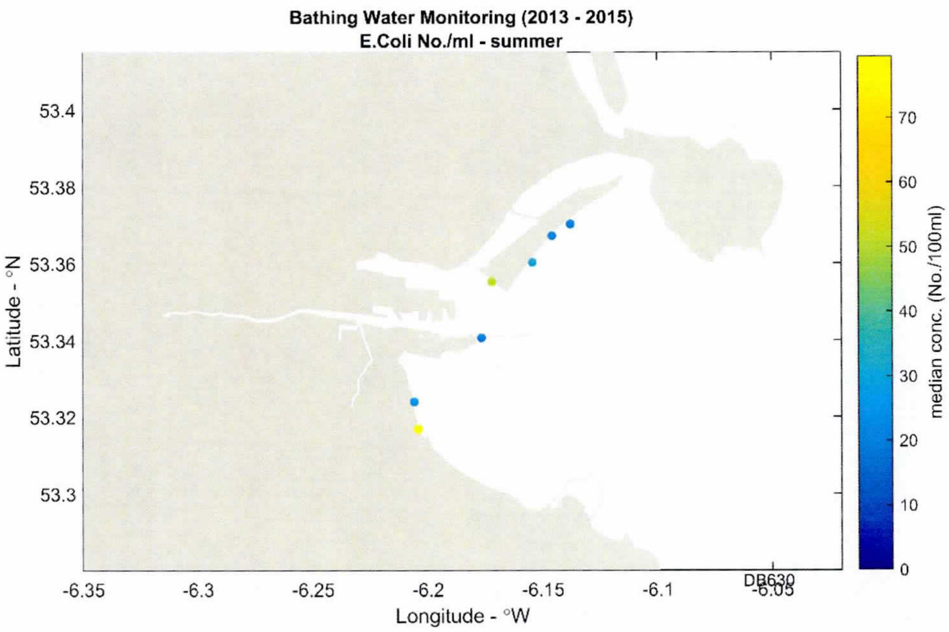

Figure 4.30 Map of Dublin Harbour and Dublin Bay showing median concentration of E. coli at bathing water sites during summer (2013 – 2015).

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5 Hydrodynamic Model

5.1 Methodology

The observational data described in Section 0 shows that the hydrodynamics of the Lower Liffey Estuary, Tolka Estuary and Dublin Bay exhibit a distinct vertical structure, with the balance between freshwater flows, tidal energy and other meteorological forcing (principally wind) controlling the flow. Of fundamental importance to creating a suitable representation of the hydrodynamic environment is to utilise a 3-dimensional model capable of calculating the buoyancy effects due to temperature and salinity stratification.

DHI's 3D model system MIKE 3 FM is applicable for analysing free-surface flow hydrodynamics and heat dispersion in coastal areas and seas. The MIKE 3 FM flow model is a 3D model based on an unstructured flexible mesh and uses a finite volume solution technique. The meshes are based on linear triangular and quadrangular elements. This approach allows for a variation of the horizontal resolution of the model mesh within the model area to allow for a finer resolution of selected sub-areas (see Appendix A for further information).

It was ensured that the computational mesh was sufficiently resolved in order that detailed geometries and complex flow patterns in the river and bay were appropriately captured. For example, around the intake and outfall structures on the Lower Liffey the triangles that defined the computational grid had spatial length scales of only a few metres.

The vertical model resolution was based on a discretisation in layers of varying thicknesses, known as sigma layers. The number of layers was invariant over the model area and independent of variations in water depth and water level. The principle of resolving the vertical part of the computational model grid by using sigma layers can be understood by example in Figure 5.1. The number of layers included in this study (8) was selected to adequately resolve the vertical gradients in temperature and salinity.

A hydrodynamic and thermal model using MIKE 3 Flexible Mesh (FM), was first set up for the Lower Liffey Estuary during the "Dublin Waste to Energy" (WtE) project (Ref. /2/). The geographical coverage of the model included the outer parts of the Lower Liffey Estuary, the Tolka Estuary and the Dublin Bay area to ensure a correct prediction of the circulation in the area. The model was later extended north and south during the Dublin ocean outfall study to ensure correct oceanographic representation further offshore (Ref. /1/).

The model constructed for these two previous studies formed the basis for the hydrodynamic model for the present investigation for the Ringsend WwTP. The setup and calibration of the updated hydrodynamic and thermal model are described in the following sections.

The model domain is first described (Section 5.2). The model mesh and bathymetry were updated to reflect more detailed and up-to-date information gathered in recent years (Section 5.3). To ensure the hydrodynamic model accurately describes the important physical processes within the estuary and bay, a model calibration exercise was performed. The boundaries and sources specified for the model calibration period were established (Section 5.4). The model was then compared against observed data on water levels, current speed, temperature and salinity within (Section 5.5).

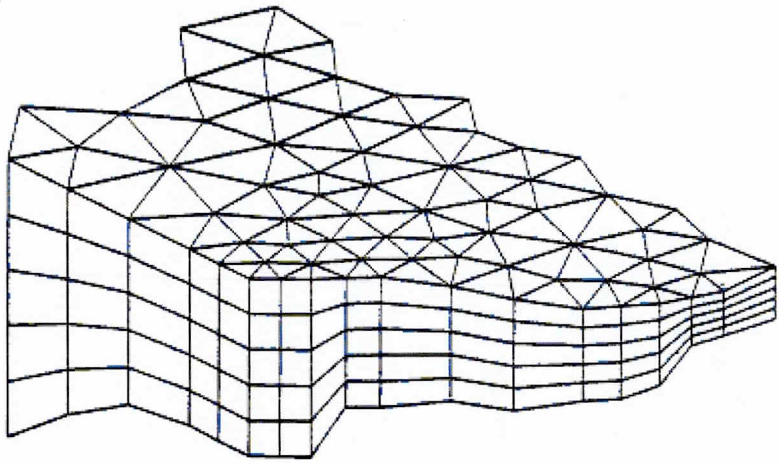


Figure 5.1 Example of schematisation of a 3D model mesh with 5 vertical sigma layers (note that the model developed in this study has 8 vertical layers).

5.2 Model Domain

The geographical coverage of the model included the study area of the Lower Liffey Estuary, the Tolka Estuary and Dublin Bay to ensure a correct prediction of the circulation in the area (Figure 5.2).

The offshore boundaries were positioned sufficiently far away from the from the study area to ensure that any boundary effects do not influence the model solution within the Bay. The open boundary extended more than 20 km to the North, South, and East of Dublin Bay to ensure correct oceanographic representation further offshore.

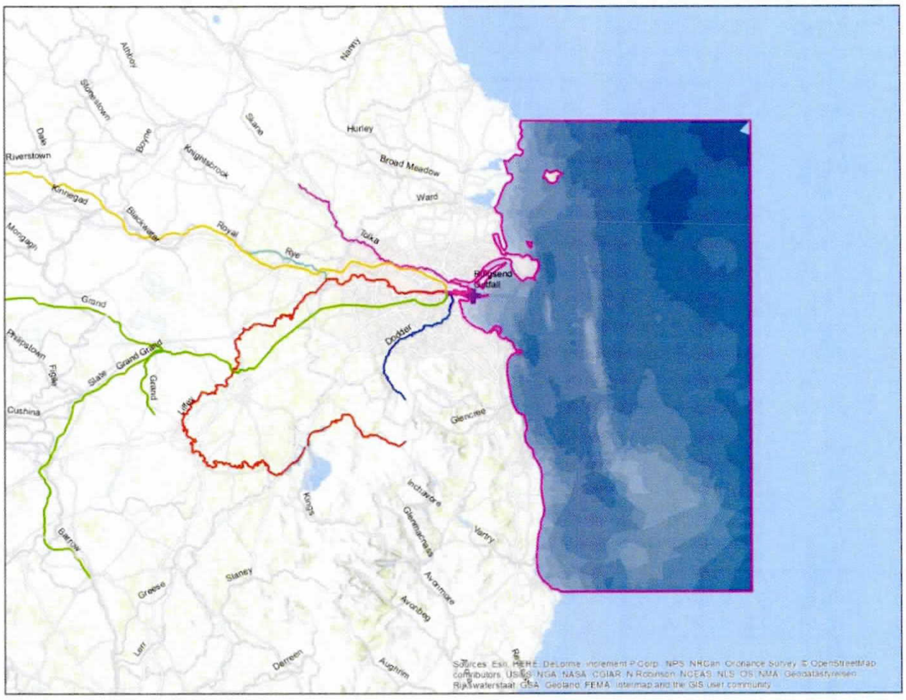


Figure 5.2 Geographical coverage of the Dublin model (pink outline), showing shaded bathymetric data of the Dublin Bay area.

5.3 Mesh and Bathymetry

Bathymetric scatter data were available from several data sources, including:

- EMODNet Bathymetry data for the Irish Sea;
- Data from a survey that DHI conducted in 2005 (see Ref. /2/);
- Lidar of part of the Tolka and particularly Bull Island (Source: OPW, 2012);
- Soundings of the Clontarf Basin/Estuary (Source: OPW, 2012);
- Charted soundings for the approach channel and basins (Dublin Port, August – September 2015)
- Soundings of the Liffey Estuary 2003. This area was surveyed as part of Irish National Seabed Survey (rebranded INFOMAR); and
- Soundings of the Dodder Estuary (2006).

All bathymetry data were converted to a common vertical datum representing mean-sea-level (MSL), which is approximately 0.1m above Ordnance Datum (OD) Malin.

The computational mesh was generated to provide adequate resolution within the rivers, estuaries and Dublin Bay. It was ensured that the mixing zone around the Ringsend WwTP outfall was suitably resolved in order to capture the dispersion of the effluent into the estuary and its discharge into the Bay.

Figure 5.3 to Figure 5.5 show the details of the hydrodynamic model mesh. The minimum spatial resolution was 15 – 20 m in the area around the Ringsend WwTP outfall. In the Liffey and Tolka estuaries, the resolution was typically 100m and within Dublin Bay, was set between 200 – 400 m. The mesh resolution increased with distance offshore up to a maximum value of around 3000 m at the offshore boundaries.

The vertical model resolution was set such that 8 layers distributed equidistant across the water depth.

The bathymetric scatter data was interpolated onto the computational mesh to create a single model bathymetry surface, vertically referenced to MSL (Figure 5.6).

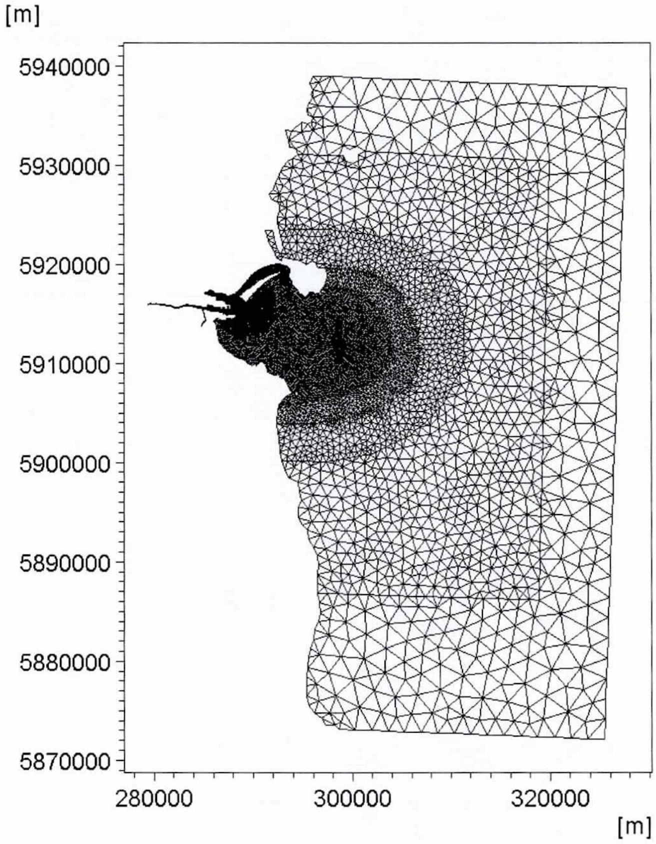


Figure 5.3 Overview of the domain and horizontal mesh for the Ringsend hydrodynamic model.

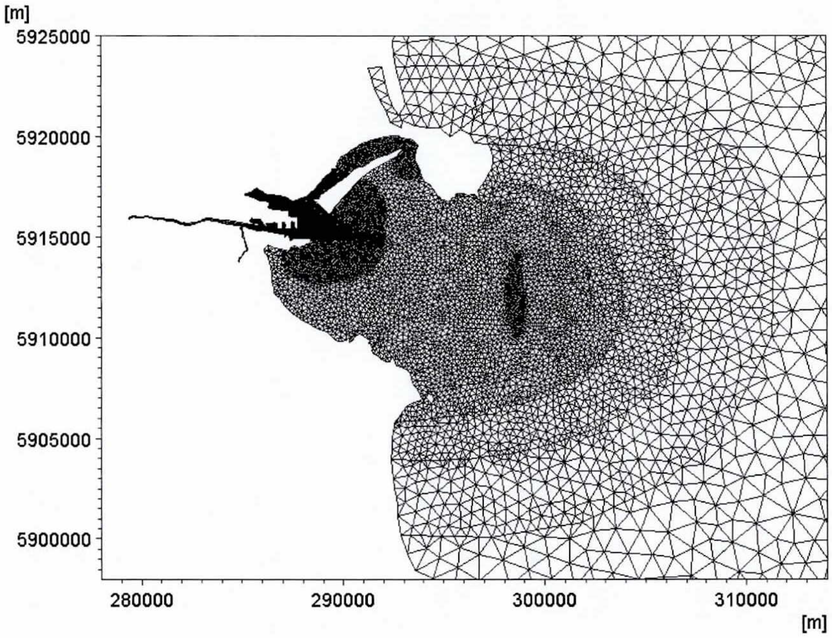


Figure 5.4 Details of the horizontal mesh for the Ringsend hydrodynamic model within Dublin Bay.

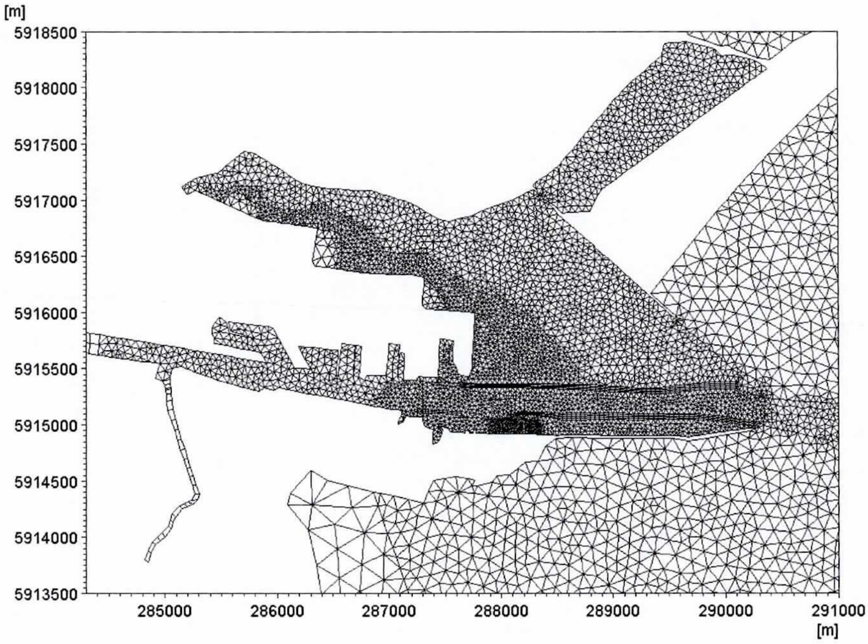


Figure 5.5 Details of the horizontal mesh for the Ringsend hydrodynamic model within Lower Liffey, Liffey Estuary, Tolka Estuary, and Bull Island.

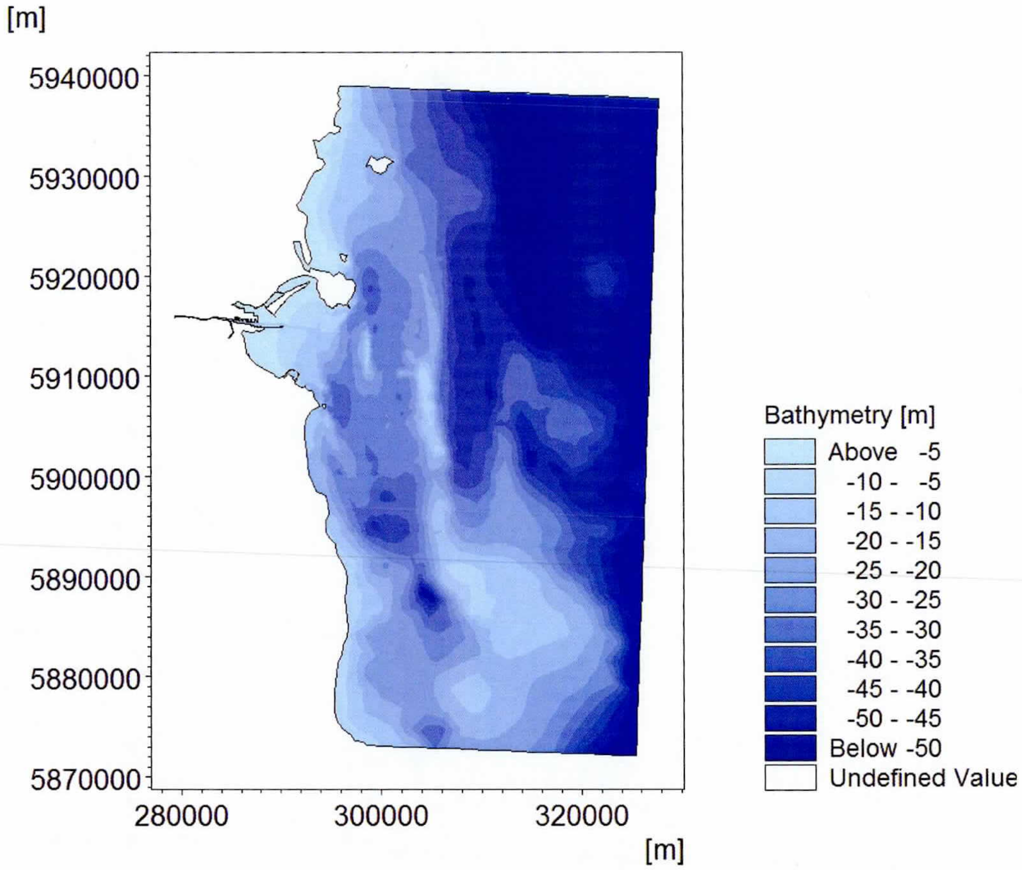


Figure 5.6 Model bathymetry interpolated onto computational mesh.

5.4 Model Setup

The hydrodynamic model was set-up to include flooding and drying of inter-tidal areas, tidal forcing along the open boundary towards the Irish Sea and freshwater river run-off from the Rivers Cammock, Liffey, Tolka, Santry and Dodder. Table 5.1 below summarises the model set-up that was used during the calibration runs. Information on the boundary conditions, river discharges and outfall specifications are detailed in the below.

Table 5.1 General settings for the hydrodynamic model calibration runs.

Periods	20th September 2015 – 24th October 2015 (Cal 1) 01st April 2010 – 12th May 2010 (Cal 2) 01st July 2009 – 11th July 2009 (Cal 3)
Overall Time step	300 seconds
Mesh, number of horizontal elements	11,474
Number of vertical layer	8
Horizontal turbulence	Smagorinsky formulation
Vertical turbulence	k-epsilon formulation
Bottom friction (bed roughness)	0.15m in the Estuary 0.05m in Dublin Bay and offshore
Horizontal diffusion factor	1
Vertical diffusion factor	0.1

5.4.1 Boundary Conditions

Tidal forcing was applied along the offshore open boundaries of the hydrodynamic model. The offshore boundary data were extracted from a regional model of the Irish Sea developed and maintained by DHI (Figure 5.7). The regional tidal model was in turn driven by surface elevations from a global tidal model.

The tidal data were specified as varying (spatially and temporally) along each of the open boundaries, thereby enabling the spatial variation in water surface elevation to be captured by the model.

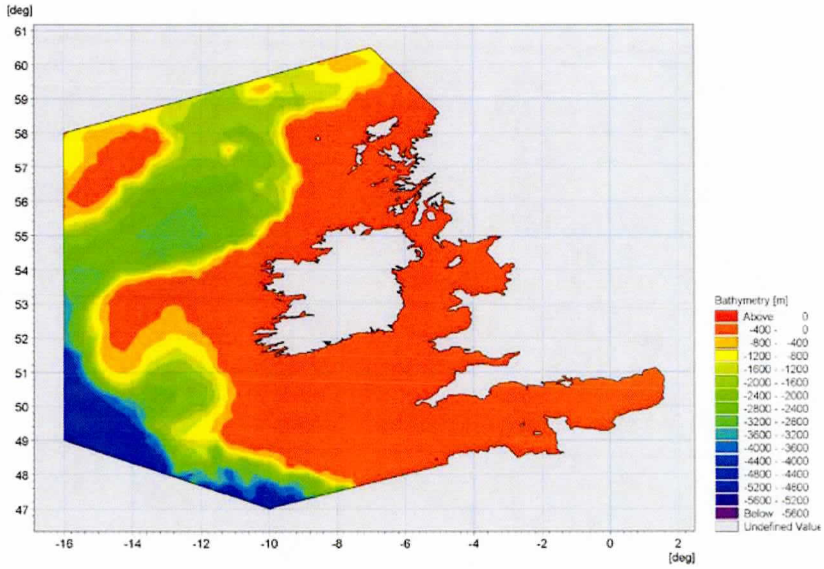


Figure 5.7 Domain of the regional model hydrodynamic model.

5.4.2 Meteorological Conditions

As the 3D model interacts with the atmosphere through heat exchange there was a requirement to include atmospheric temperature effects for the calibration period. The atmospheric conditions were determined using data from a 5-year meteorological model (2010 – 2015) for the periods of calibration as mentioned in Section 4.2 Other Data Sources.

Wind data was initially excluded from the calibration model as it had previously been considered insignificant in the overall calibration from previous studies in Dublin Estuary.

5.4.3 Vertical Mixing

The vertical mixing processes are affecting how fast freshwater runoff from the catchments and the discharge from the Ringsend WwTP are being mixed with and diluted into the saline water from Dublin Bay. In the model a vertical dispersion factor of 0.1 is applied for the mixing of salt. The value of 0.1 is, by experience and as stated in the MIKE Manual, a value that has been used with success for other estuary studies. However, the vertical mixing and exchange of non-saline and saline water can be weakened by applying a lower dispersion factor than 0.1. This factor is therefore just as important as the volume of non-saline water being discharged into the estuary.

5.4.4 Bottom Friction (Bed Roughness)

As noted in the initial model setup, bed roughness was varied from 0.05 m to 0.15 m in blocks around the domain following some initial sensitivity checks during the calibration process. Variation was undertaken to represent the relatively deep dredged channel compared to the shallow intertidal flats of the Tolka Estuary.

5.4.5 River Discharge

The locations of the river sources that were input into the hydrodynamic model are shown in Figure 5.8. The River Liffey and River Cammock discharges into the Upper Liffey Estuary. The River Dodder, Grand Canal, and Royal Canal all flow into the Lower Liffey Estuary. The flow from the River Tolka enters at the head of the Tolka Estuary. Finally, the River Santry enters the model domain behind Bull Island.

Gauged river daily flow rates during the calibration period were available from the EPA Hydronet data-portal for the River Cammock, Dodder, Slang and Tolka (Figure 5.9 to Figure 5.11). Note that as River Slang is a tributary of the River Dodder, both of which are gauged upstream of their confluence. The flow rate in Figure 5.11 is therefore calculated as the sum of the flow in those two rivers.

For the River Liffey, mean daily flow rates were provided for the ESB at Leixlip. It was noted that during the calibration period the data for the Liffey were sparse and may contain missing or constrained data. The River Liffey is a major river and the gauge at Leixlip was located some distance upstream of the location at which it entered the hydrodynamic model domain (at Islandbridge Weir). It was therefore decided to scale the gauged flow rate for the Liffey to account for additional run-off into the river between the gauging station and the receiving water.

$$Q_{Liffey,Scaled} = Q_{Liffey,Leixlip} \times \frac{A_{Liffey,Islandbridge}}{A_{Liffey,Leixlip}}$$

$A_{Liffey,Islandbridge}$ and $A_{Liffey,Leixlip}$ were the catchment area of the Liffey at Islandbridge Weir and Leixlip Power Station, respectively. These values were taken from the Eastern CFRAM Study Hydrology Report (Ref. /10/) which gave a scale factor of 1.132. Finally, the contributions from two tributaries, the River Rye (as gauged at Leixlip) and the River Grifeen (as gauged at Lucan) were also included.

Figure 5.12 shows the final time-series for the daily mean River Liffey flow rate during the model calibration period. The flow rates in the Liffey were, on average, larger than other rivers in the model. However, the coarser temporal resolution means that short-duration events (such as the high flow that occurred 5th of October 2015) are not fully captured.

A comparison was therefore made with other studies in the area (e.g. Ref. /7/) which suggested values as shown in Table 5.2. It is considered that river inflow (both in terms of quantity and time variability) to the model domain remains an area where consensus between studies has not been reached.

The final selection of river discharge rates for the calibration period are shown in Table 5.3. Note that the rates for the Royal Canal and Grand Canal are estimated values as no detail on the operation of the gates was available to this project.

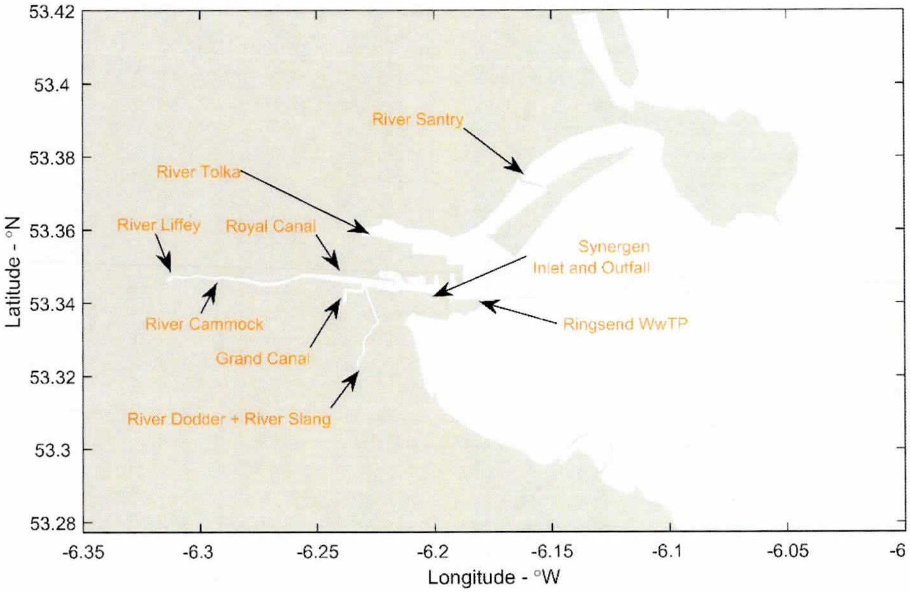


Figure 5.8 Location of rivers and outfalls specified for hydrodynamic model at calibration stage.

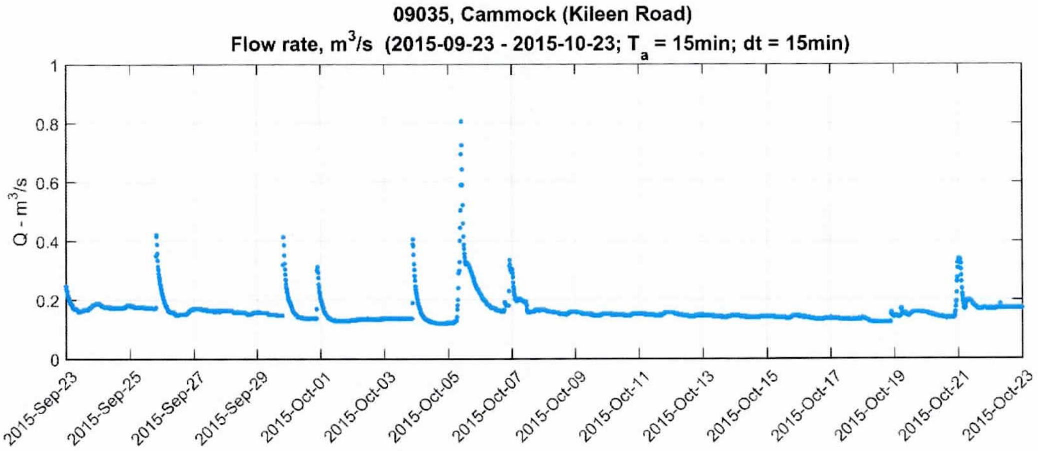


Figure 5.9 Time-series of gauged flow rate for the River Cammock during the hydrodynamic model calibration period (23 Sept. – 23 Oct. 2015).

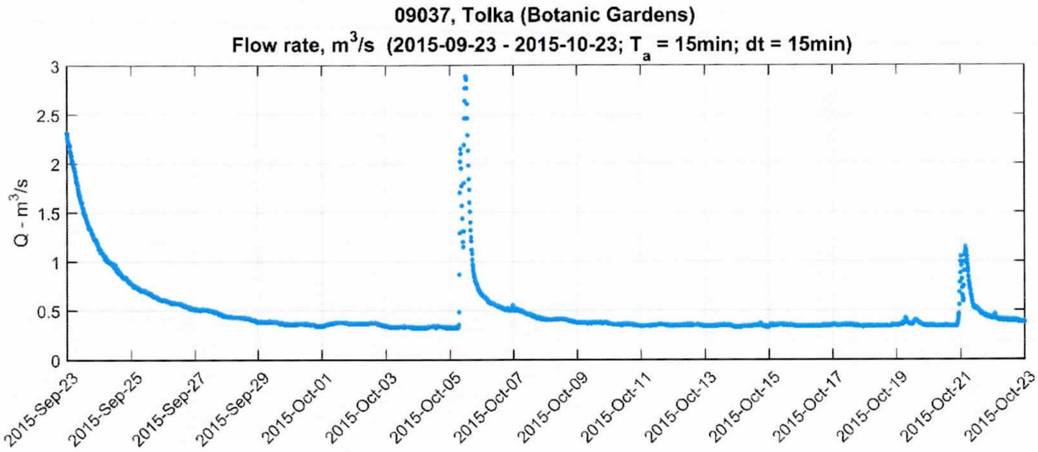


Figure 5.10 Time-series of gauged flow rate for the River Tolka during the hydrodynamic model calibration period (23 Sept. – 23 Oct. 2015).

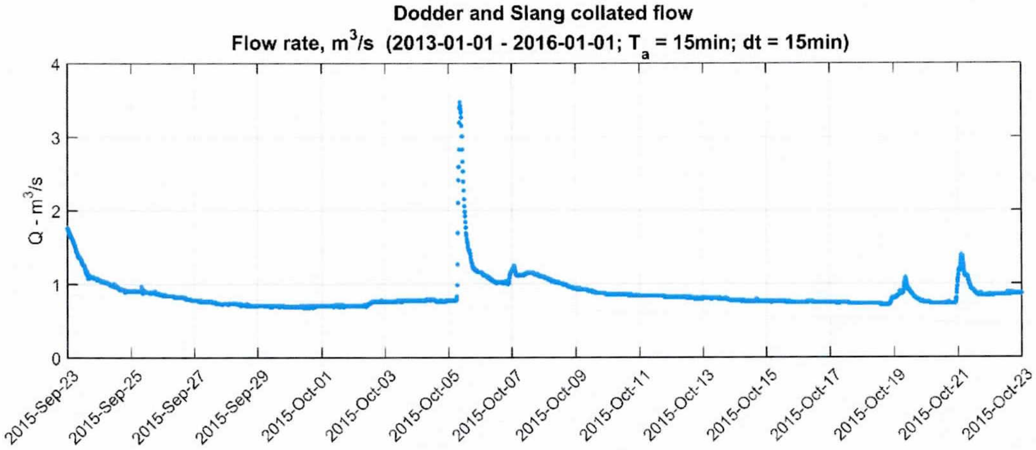


Figure 5.11 Time-series of gauged flow rate for the combined River Dodder and Slang during the hydrodynamic model calibration period (23 Sept. – 23 Oct. 2015).

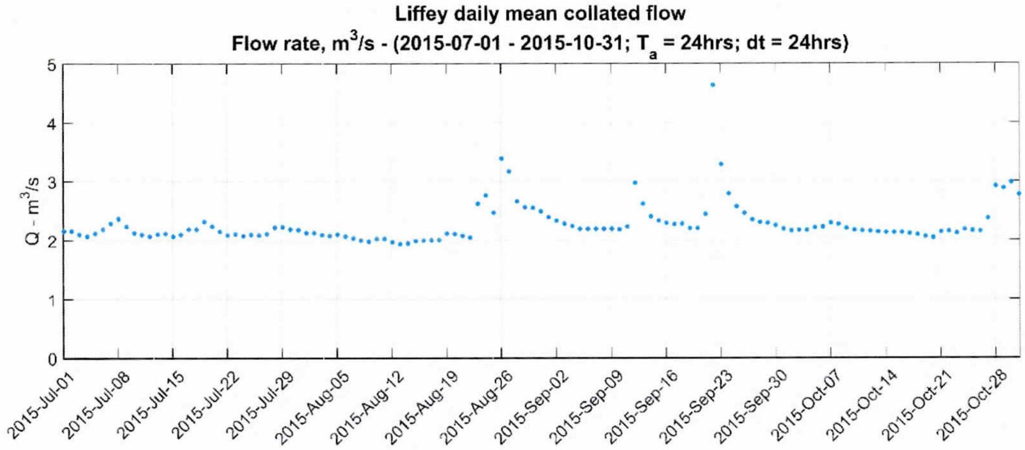


Figure 5.12 Time-series of gauged flow rate for the River Liffey during the hydrodynamic model calibration period (23 Sept. – 23 Oct. 2015).

Table 5.2 Discharge rates for main rivers from other studies (Ref. /7/).

River	Mean annual flow rate Q_{av} [m3/s]	Mean winter flow rate $Q_{av,winter}$ [m3/s]
Liffey	15.6	25.0
Cammock	2.3	2.6
Dodder	1.4	1.6

Table 5.3 Discharge rates specified for main rivers in calibration model setup.

River	Flow rate, Q [m3/s]	Temperature [°C]	Salinity [PSU]
Liffey	Figure 5.12	15	0
Cammock	Figure 5.9	15	0
Dodder + Slang	Figure 5.11	15	0
Tolka	Figure 5.10	15	0
Santry	0.2	15	0
Royal Canal	0.1	15	0
Grand Canal	0.1	15	0

5.4.6 Outfalls

The locations of the inlets and outfalls on the Lower Liffey Estuary that were specified in the hydrodynamic model are shown in Figure 5.8.

It should be noted that in the calibration runs, there was no allowance for freshwater input to the system from the city drainage.

Synergen Power Station

The Synergen Power Station is a combined cycle gas generating plant located on the south side of the River Liffey. The plant extracts cooling water from the Lower Liffey and discharges this water via a channel back into the estuary approximately 1 kilometre upstream of the Ringsend WwTP.

Figure 5.13 shows the measured hourly discharge and temperature of water from the Synergen outfall during the model calibration period. These data were specified in the 3-dimensional hydrodynamic and thermal model for the Synergen Outfall location. For maintenance of continuity, the discharge at the Synergen intake was set to the same volume flux, but with opposite sign (i.e. negative discharge).

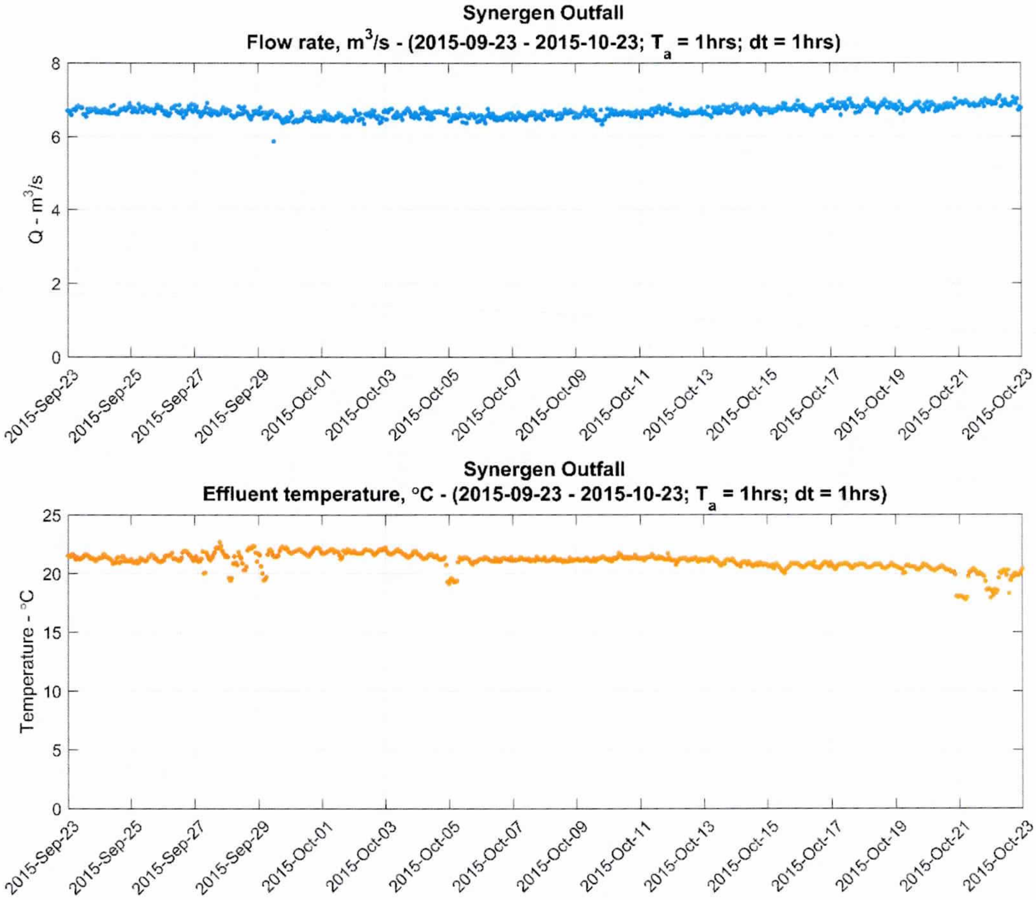


Figure 5.13 Time-series of discharge (upper panel) and temperature (lower panel) of Synergen Power Station outfall during model calibration period.

Ringsend WwTP

The Ringsend WwTP outfall is located on the south side of the River Liffey, adjacent to the South Bull Wall. There are two outfall locations for the Ringsend WwTP:

- SW1, Primary Wastewater Discharge on the Lower Liffey and within the ESB Poolbeg Cooling water Channel.
- SW2, Storm Water Overflow Discharge, located approximately 500m upstream of SW1 on the Lower Liffey Estuary.

It was assumed during the calibration period that only the primary wastewater discharge point (SW1) was active.

Figure 5.14 shows the measured daily mean discharge and temperature for the primary discharge point SW1 during the model calibration period. These data were specified in the 3-dimensional hydrodynamic and thermal model for the Ringsend WwTP outfall.

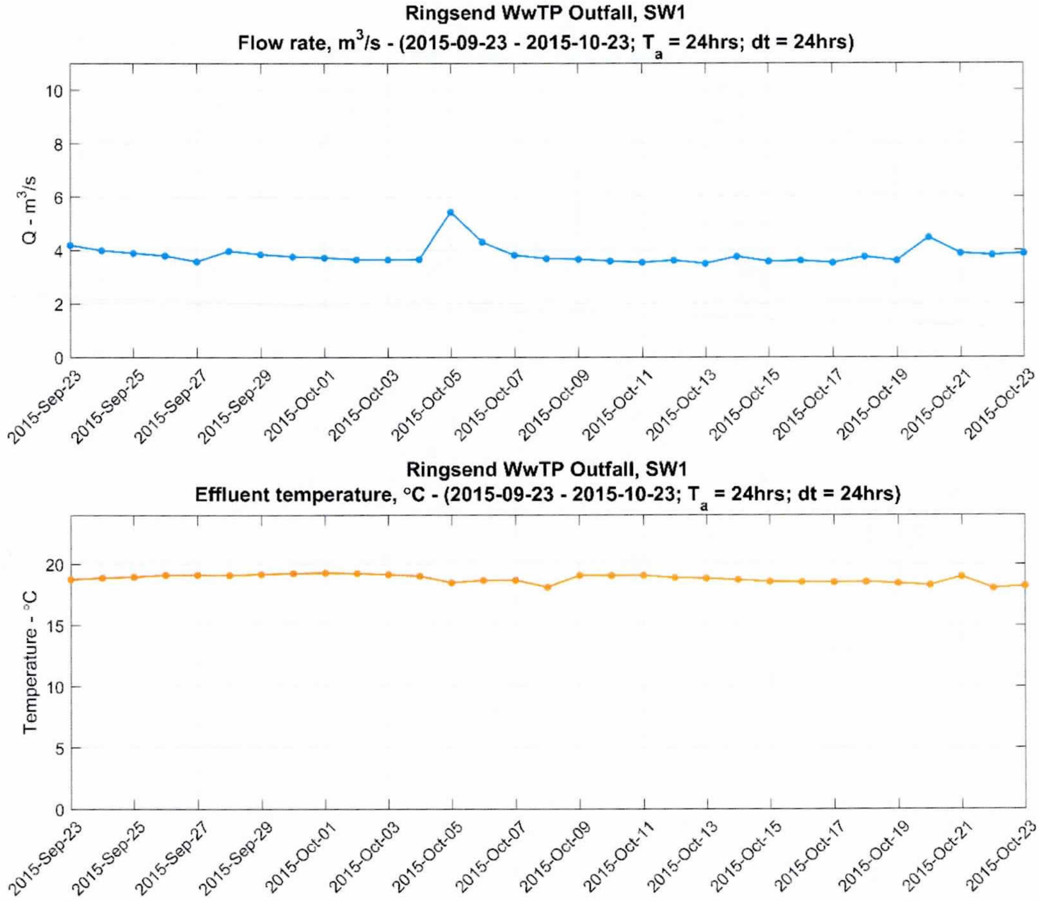


Figure 5.14 Time-series of discharge (upper panel) and temperature (lower panel) at primary Ringsend WwTP outfall during model calibration period.

5.4.7 Structures

North Bull Wall

The North Bull Wall is a 3-km long breakwater that separates the Tolka Estuary from Dublin Bay. From the evidence of satellite imagery and local knowledge, it is understood that the outer section of the North Bull Wall (approximately 1 km) is submerged during certain stages of the tide (Figure 5.15). Navigational charts indicate that wall is covered by 0.6 to 2.7m at high-water along its length.

To account for the fact that the outer part of the North Bull Wall is semi-submerged and consequently its influence on the circulation within the harbour is dependent on the stage of the tide, this structure was specified in the hydrodynamic model as a dike (see Figure 5.16). The dike acts as a physical barrier between Dublin Bay and the harbour when the water level is below the specified height of the dike. When water levels exceed the height of the dike, water discharges over the structure according to the pressure gradient (upstream to downstream water levels).

In the hydrodynamic model, the height of the dike representing the North Bull Wall varies linearly from 1m above mean-sea-level at the northern end to -1.1m below mean-sea-level at the southern end (Figure 5.15).

Figure 5.15 Aerial image of Dublin Harbour highlighting the semi-submerged section of the North Bull Wall (image Courtesy of Google Earth).

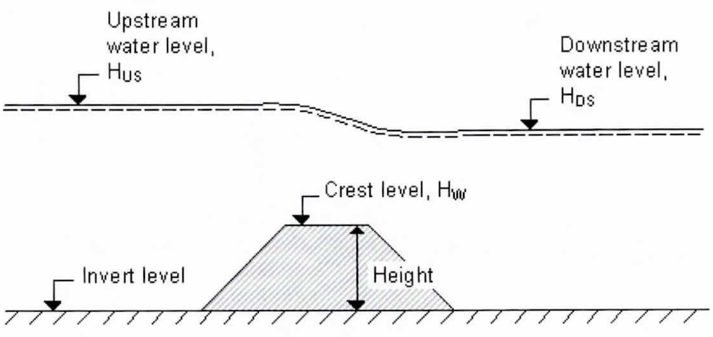


Figure 5.16 Schematisation of a dike structure as applied for the North Bull Wall in the hydrodynamic model.

ESB Cooling Water Channel

The effluent from the Ringsend WwTP primary discharge (SW1) flows into the outer part of the ESB Poolbeg Power Station cooling water channel and then into the Lower Liffey Estuary via a weir.

Figure 5.17 shows an annotated aerial satellite image of the channel and the weir. The treated effluent discharges into the channel at position **Point 1**. The weir is located at position **Point 2** and faces downstream of the WwTP and towards Dublin Bay. The water in the channel will flow over the weir when the water level in the Liffey Estuary is lower than the height of the weir. It was thus assumed that the weir was originally designed to discharge the treated effluent into the Liffey primarily during ebb tide (i.e. out-going tide).

However, it was evident from Figure 5.17 that water enters the Lower Liffey along the back section of the weir (at positions **Point 3** and **Point 4**). This was confirmed by visual inspection during a site visit during August 2016. At low water, it was observed that the sheet piles that form the outer walls of the channel between Point 3 and Point 4 were either heavily corroded or missing entirely (lower panel of Figure 5.18). As a result, water was discharging into the Lower Liffey primarily via these two flow routes during low tide. During high tide, the water level in the Lower Liffey is higher than both the weir and the level of the damaged sheet piles, allowing the discharged water to mix (Figure 5.19).

To take this into account, the weir was included in the hydrodynamic model as four sections as described in Figure 5.20 and Table 5.4. Along the existing weir (Section D), the crest level was set to 0 m relative to mean-sea-level, meaning that the water will flow out only when the tide falls below this level.

Sections C and B were set to be flowing out at most stages of the tide, with levels of -1mMSL, signifying the flow paths at locations Point 3 and Point 4. No specific elevation information was available for the remaining pile line at Section A so a nominal level of 1 mMSL was selected to represent the highly corroded nature of this line of wall.

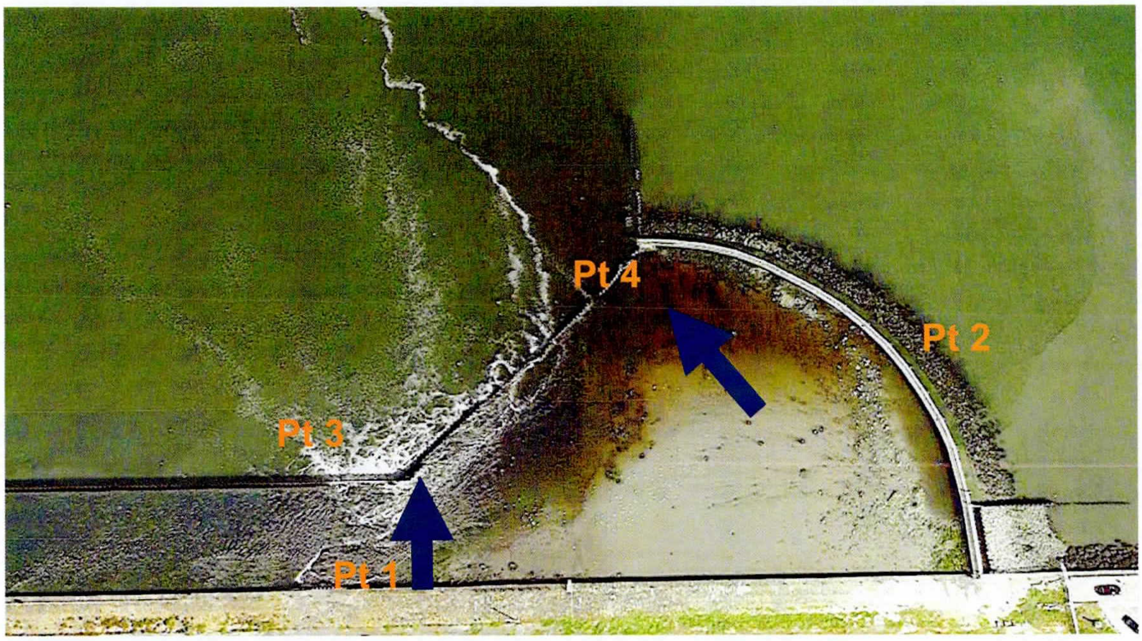


Figure 5.17 Aerial image of Ringsend WwTP outfall SW1 (Courtesy of Google Earth).

Figure 5.18 Photographs of the Ringsend WwTP outfall location during low tide on 2nd of August 2016. Top image shows the weir and walkway (section D of Figure 5.17). Lower image shows broken and damaged sheet piles along the back section of the existing weir (Section B and C of Figure 5.17).

Figure 5.19 Photographs of the Ringsend WwTP outfall location during high tide on 3rd of August 2016. Top image shows the weir and walkway (section D of Figure 5.17). Lower image shows water flowing into the Lower Liffey over the damaged sheet piles along the back section of the existing weir (Section B and C of Figure 5.17).

Table 5.4 Crest levels of Ringsend Weir sections specified in the hydrodynamic model.

Weir Section	Weir Crest Level [mMSL]
Section A	1
Section B	-1
Section C	-1
Section D	0

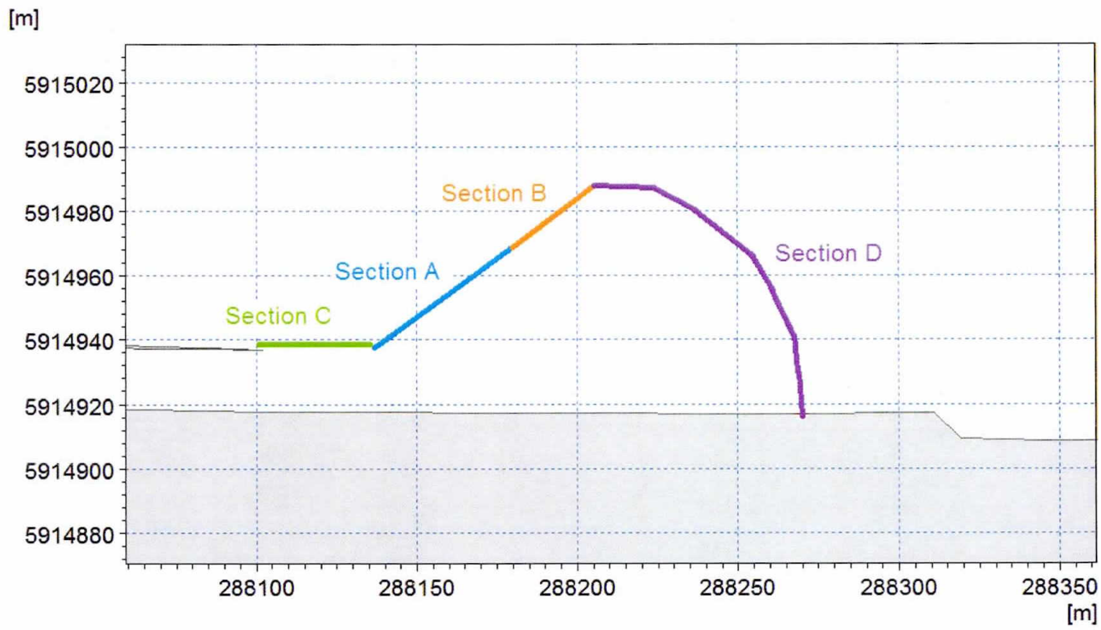


Figure 5.20 Weir sections as specified in the hydrodynamic model.

5.5 Model Results and Calibration

Calibration of the hydrodynamic model was performed based on time-series comparison between observed and modelled conditions. Further, a quantitative assessment of model performance was undertaken for specific parameters using the guidelines as specified in the UKFWR Framework for Marine and Estuarine Model Specification (Ref. /11/) combined with a more qualitative assessment of the results.

5.5.1 Waters Levels

Figure 5.21 shows a comparison of observed and modelled tidal surface elevations at the Dublin Port Tide Gauge and Ringsend Tide Gauge.

Figure 5.22 shows a comparison of observed and modelled surface elevation against the ADCP pressure sensor data (converted to water depth) from the surveys in the estuary and Dublin Bay during September and October 2015.

The model captured the timing and variation in observed water levels within the estuary over the spring-neap and semi-diurnal tidal cycle. However, it was notable that the model tidal range was lower than the observed tidal range.

For estuarine waters, the guidelines for water level validation as specified by the UKFWR (Ref. /11/) state that the following should be achieved during at least 90% of the period considered:

- Levels to within ±0.3m; and
- Timing of high water to within ±25 minutes.

Table 5.5 shows the validation statistics for water levels for the tide gauge locations at Dublin Port and Ringsend. The above criteria for timing of high water was found to be achieved for 4 of 5 locations. For water levels the UKFWR criteria was achieved for 2 of 5 locations.

Table 5.5 Model validation statistics for water levels.

Station	Mean absolute water level error [m]	Water levels ±0.3m [% of time]	Timing of high water ±25 minutes [% of time]
Dublin Port Tide Gauge	0.2	84	100
Ringsend Tide gauge	0.2	78	100
ADCP 1 – Liffey	0.2	92	100
ADCP 2 – Dublin Bay	0.1	94	100
ADCP 3 – Clontarf	0.2	79	86

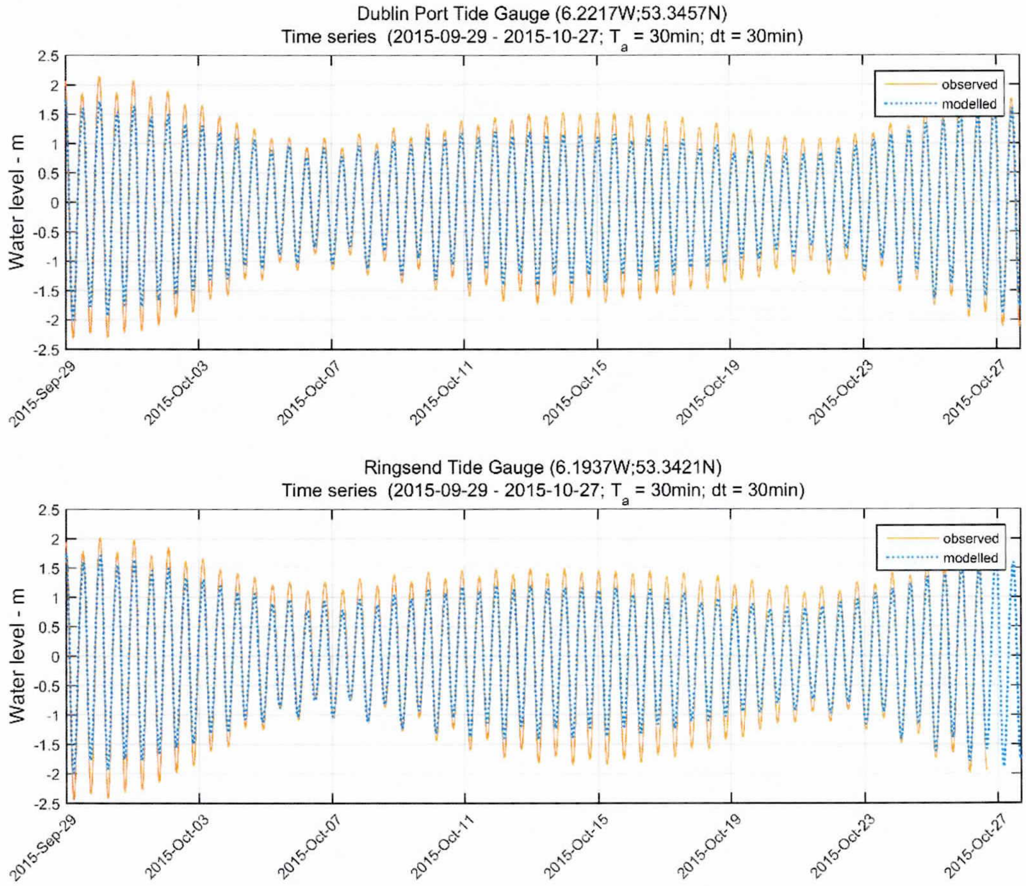
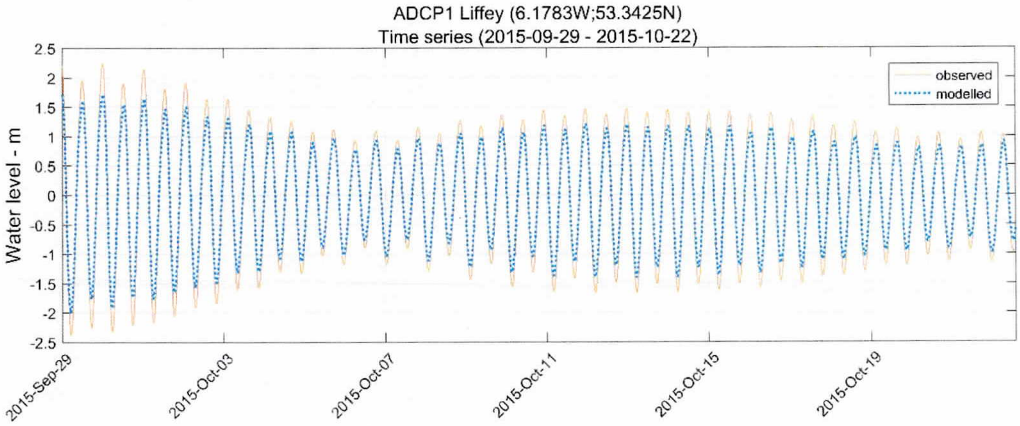
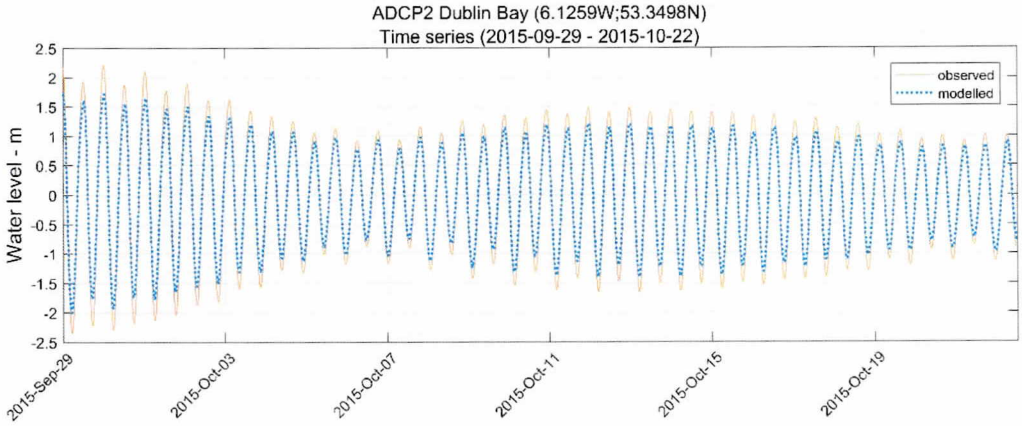


Figure 5.21 Comparison of observed (orange) and modelled (blue dashed line) tidal elevations at Dublin Port Tide Gauge (upper panel) and the Ringsend Tide Gauge (lower panel).





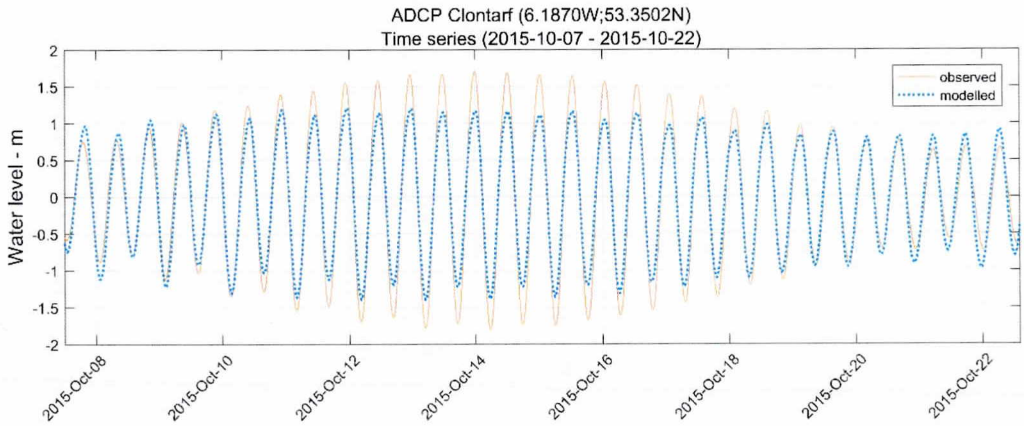


Figure 5.22 Comparison of observed (orange) and modelled (blue dashed line) tidal elevations at ADCP1 (upper panel), ADCP2 (central panel) and ADCP3 (lower panel) during model calibration period.

5.5.2 Currents

The calibration of currents in the model domain was performed using all available current speed observations from within Dublin Bay and its estuaries.

Dublin Bay

The general distribution of current speeds during peak flood tide (c. 3.5 hours before HW Dublin) and peak ebb tide (c. 4 hours after HW Dublin) within Dublin Bay are shown in Figure 5.23.

The flood tidal currents flow from south-to-north in Dublin Bay. The ebb tidal currents flow from north-to-south. The fastest currents speeds during both flood and ebb tide were found to occur around Howth Head and over the relatively shallow waters of Burford Bank at the eastern limit of Dublin Bay. Within the Bay itself, current speeds decrease from offshore-to-nearshore (i.e. as the water depth decrease). During both peak flood flow and peak ebb flow current speeds within the Bay were typically between 0.1 – 0.3 m/s.

Figure 5.24 and Figure 5.25 show a time-series comparison between observed and modelled current speed and current directions for two DHI ADCPs in Dublin Bay during 2010. In addition, further comparison with the 2015 data collection exercise in the bay has been provided in Figure 5.26 and Figure 5.27.

From these plots the hydrodynamic model provides an excellent description of the current speed and direction in Dublin Bay. In the inner 2015 survey, the data is excellent for speeds but has lower correlation for directions.

An additional validation of the hydrodynamic model within Dublin Bay was performed using data from a moving vessel ADCP survey performed by DHI in 2009. Comparison were performed by finding the model current speed in the cell and time-step matching the instantaneous observations from 7 vessel tracks (see Figure 4.16). All results are based on depth-averaged values.

Figure 5.28 and Figure 5.29 show resulting comparison between of observed and modelled current velocity vectors. The results show that the hydrodynamic model provides a very good replication of the spatial variability in current speed and direction throughout Dublin Bay; from the entrance to the harbour to beyond Howth Head.

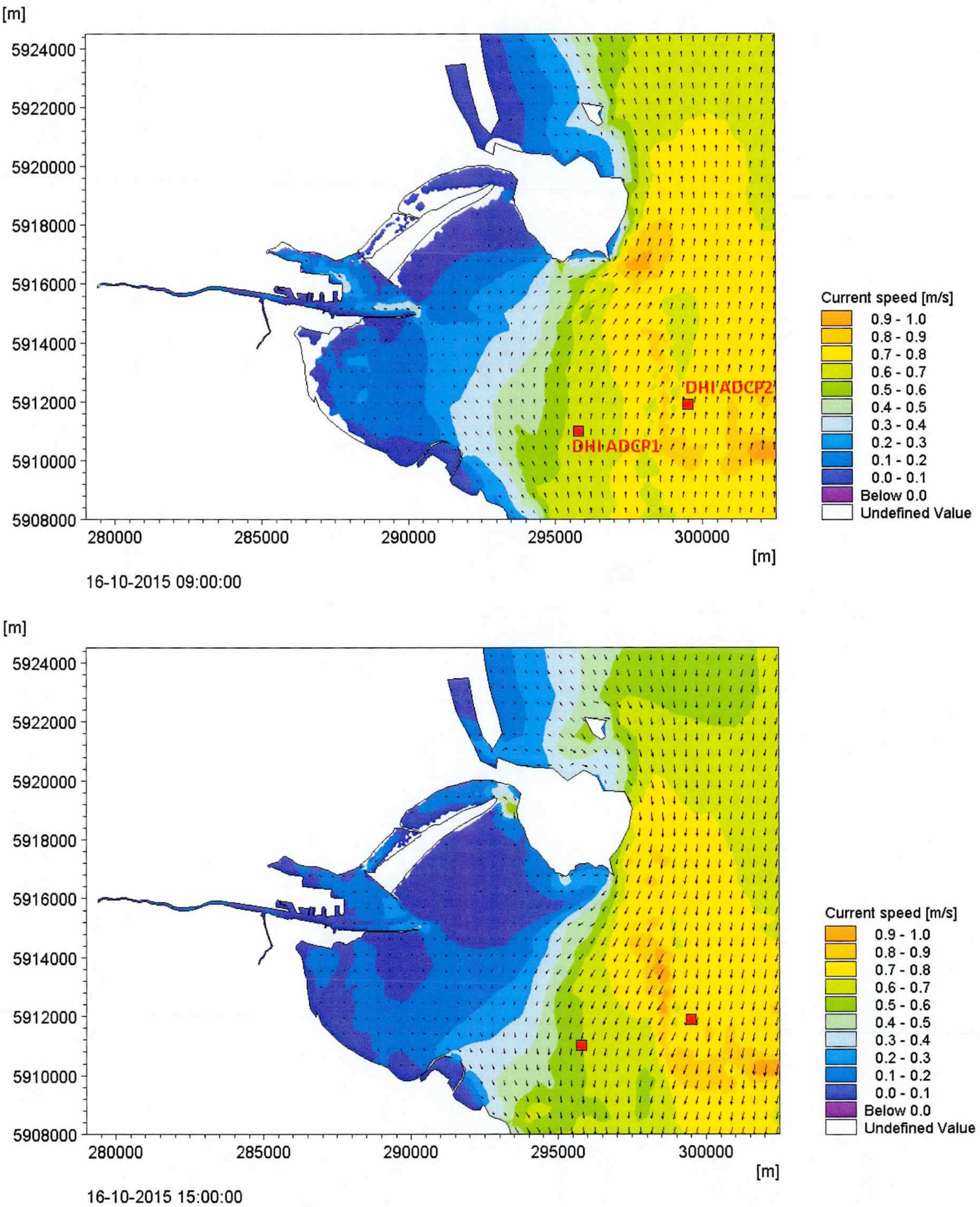


Figure 5.23 Depth-averaged current speeds in Dublin Bay for a near-spring flood tide (upper panel) and near-spring ebb tide (lower panel). Vectors show the direction that the current is flowing towards. Red markers show the location of two DHI ADCPs deployed in 2010.

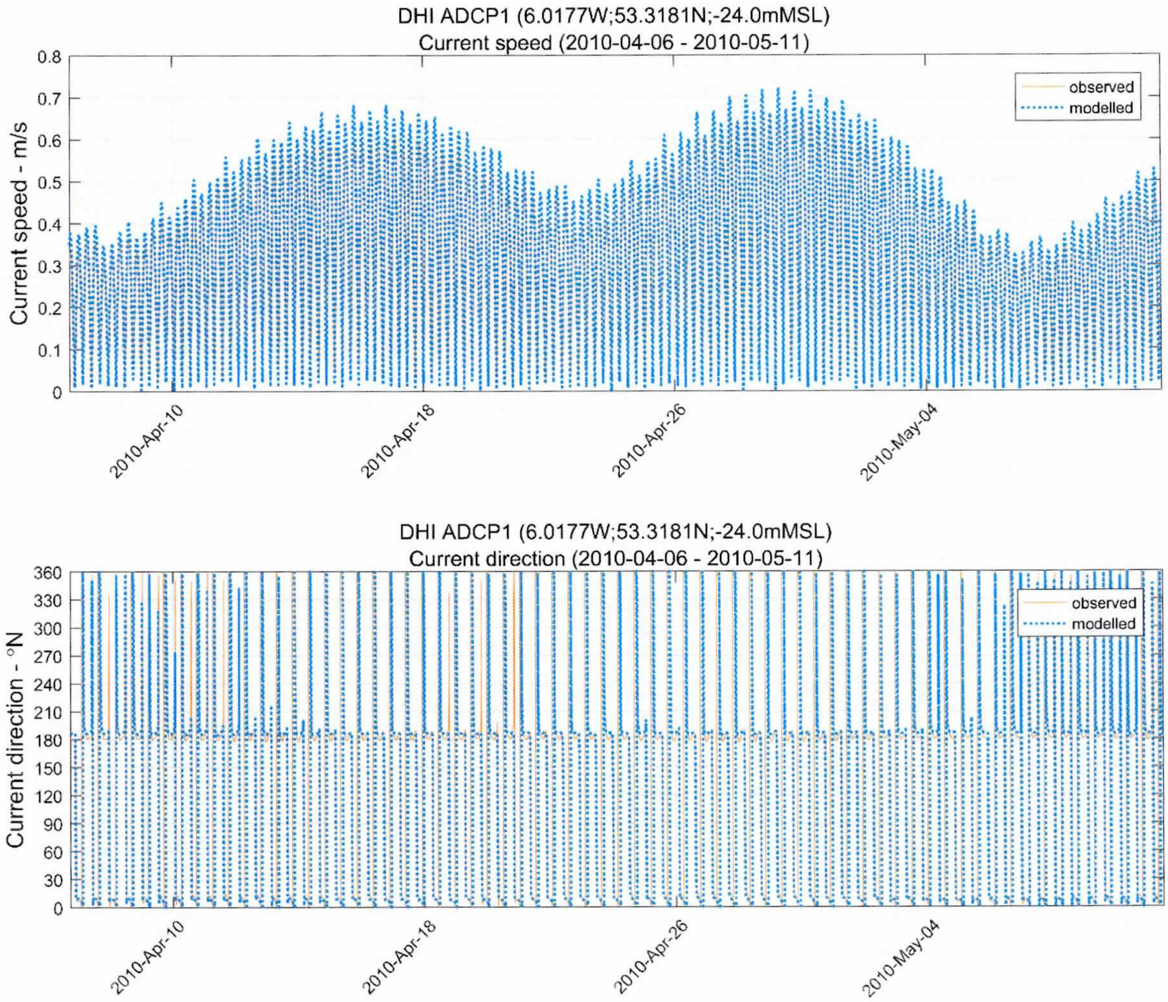


Figure 5.24 Time-series comparison of modelled and observed depth-averaged current speed (upper panel) and depth-averaged current direction (lower panel) for DHI ADCP1 in outer Dublin Bay.

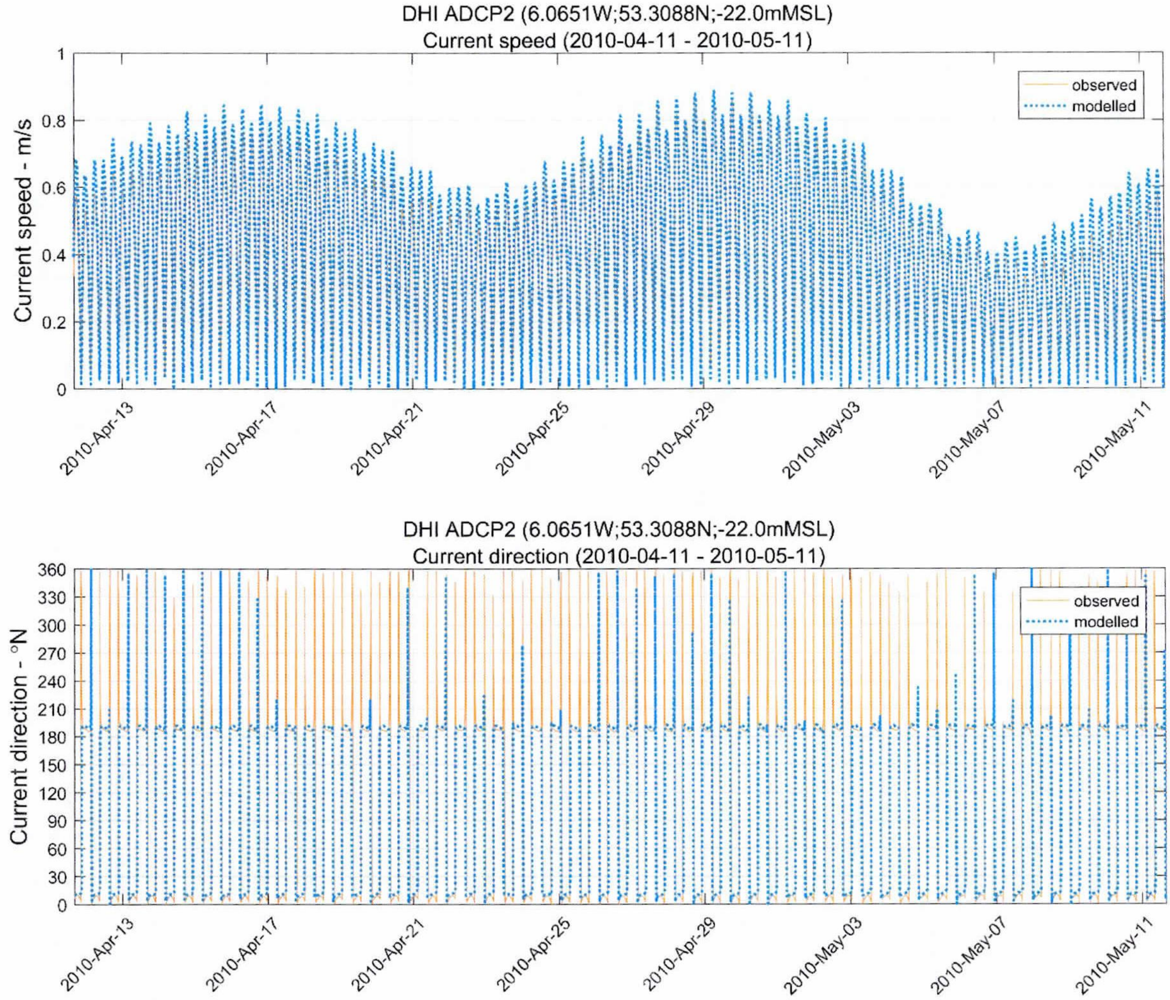


Figure 5.25 Time-series comparison of modelled and observed depth-averaged current speed (upper panel) and depth-averaged current direction (lower panel) for DHI ADCP2 at Burford Bank.

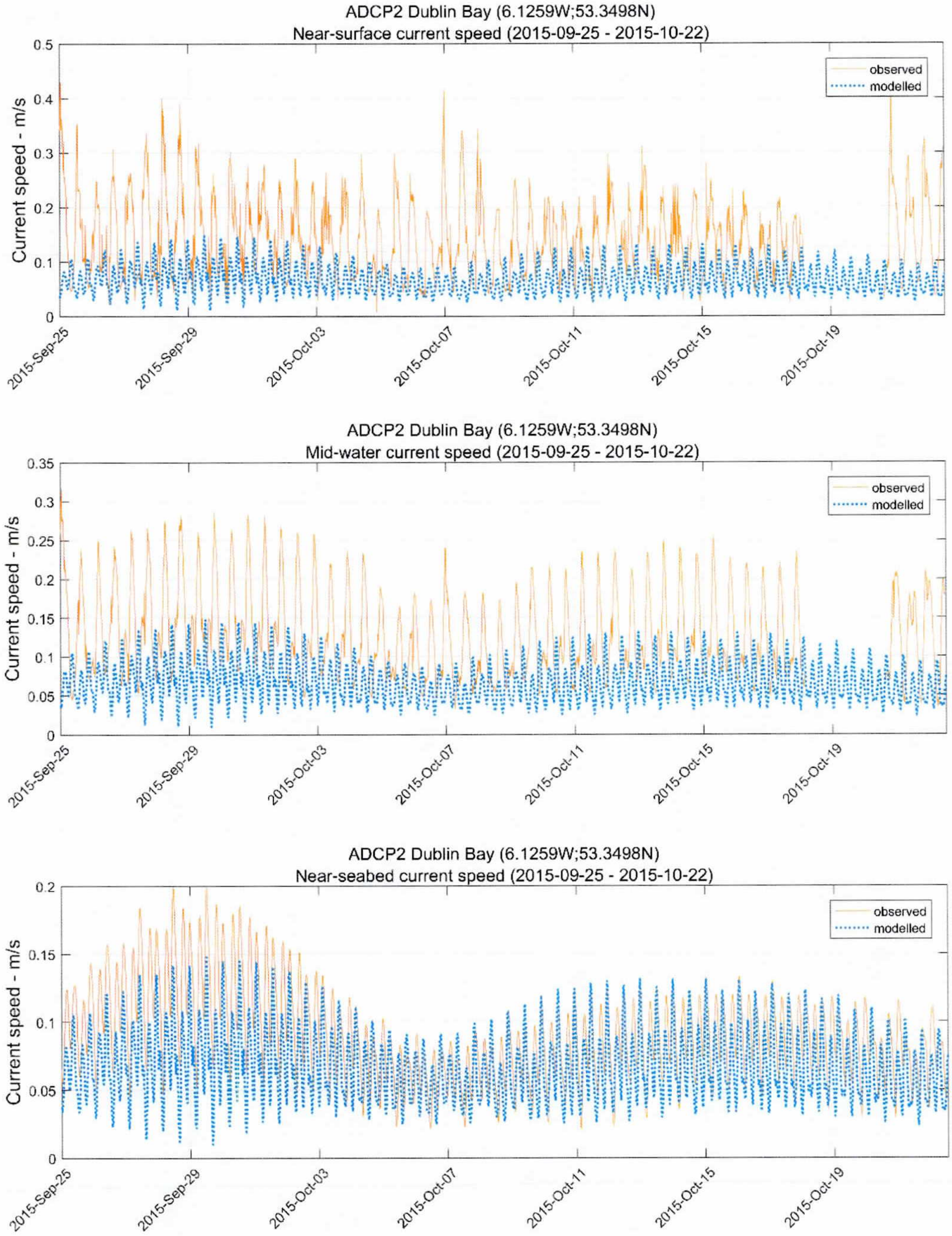


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Figure 5.26 Time-series comparison of observed (orange) and modelled (blue) current speed for ADCP2 – Dublin Bay at near-surface (upper panel), mid-water (central panel) and near-seabed (lower panel).

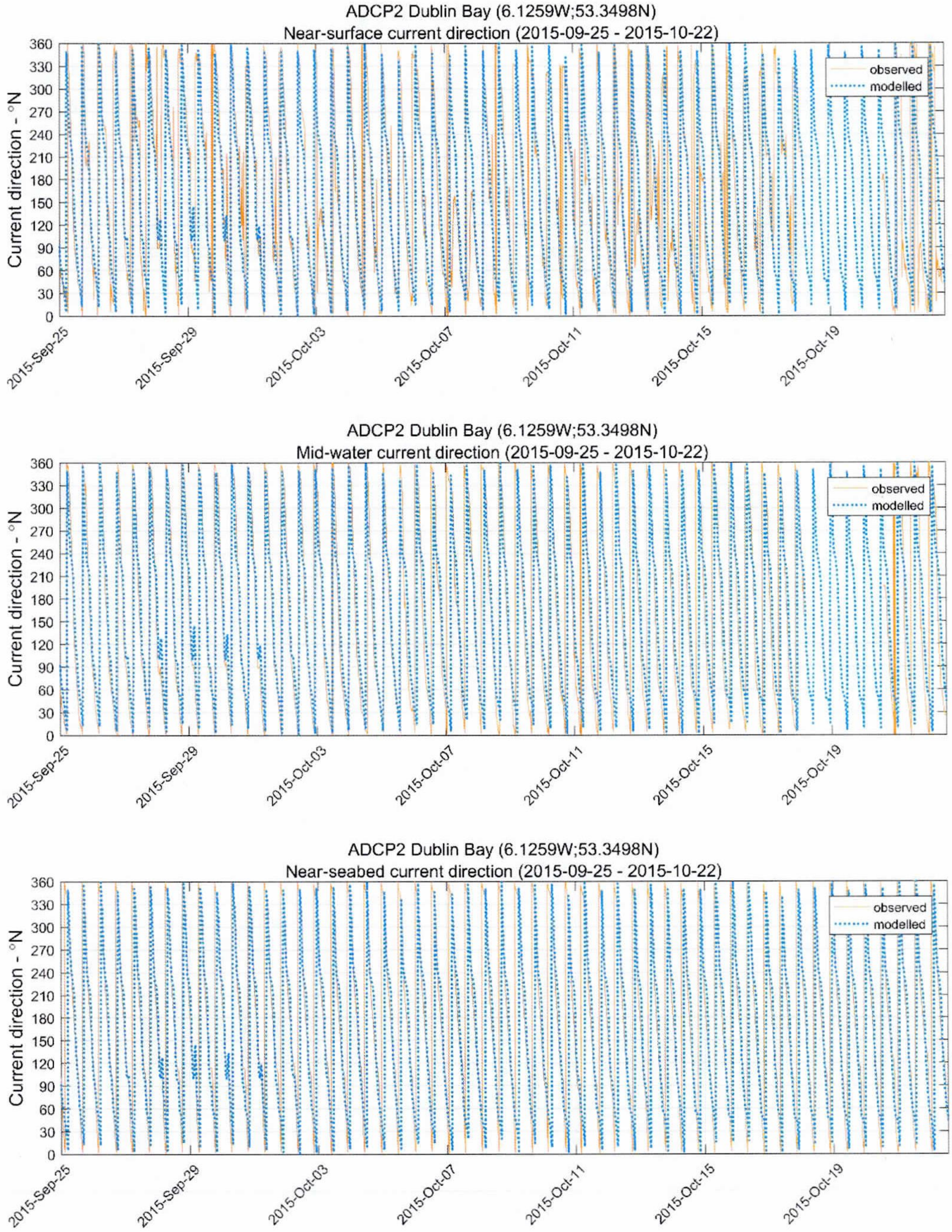

Figure 5.27 Time-series comparison of observed (orange) and modelled (blue) current direction for ADCP2 – Dublin Bay at near-surface (upper panel), mid-water (central panel) and near-seabed (lower panel).

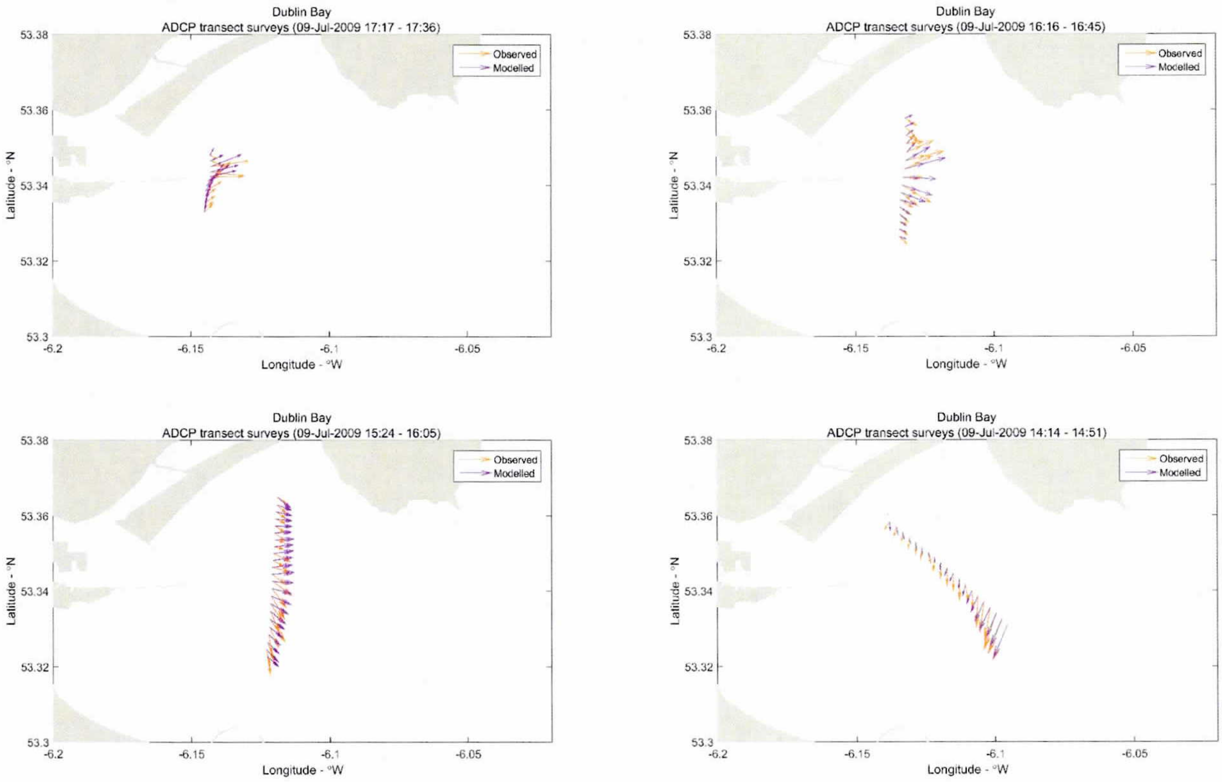

Figure 5.28 Comparison of observed (orange) and modelled (purple) current velocity vectors within Dublin Bay for (clockwise from top-left) Track 1, Track 2, Track 3 and Track 4.

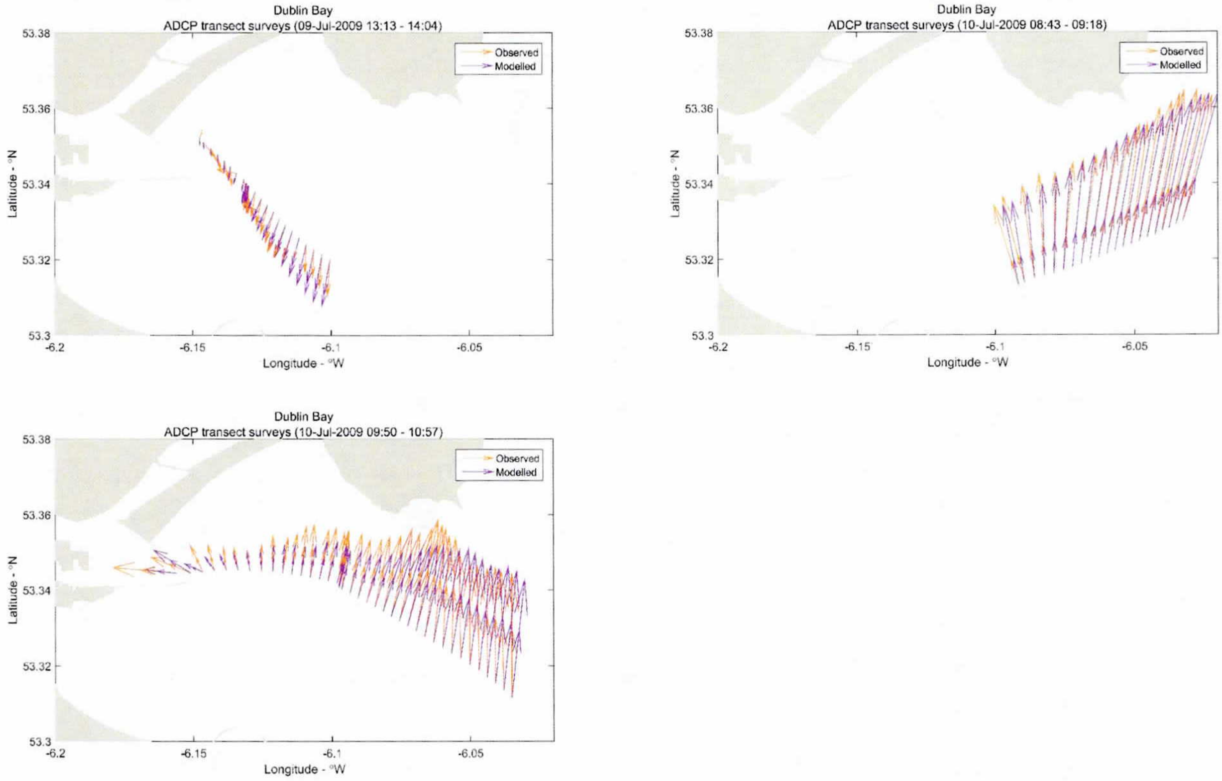

Figure 5.29 Comparison of observed (orange) and modelled (purple) current velocity vectors within Dublin Bay for (clockwise from top-left) Track 5, Track 6, and Track 7.

Dublin Port and Estuaries

The general distribution of current speeds during peak flood tide (c. 3.5 hours before HW Dublin) and peak ebb tide (c. 4 hours after HW Dublin) within the Outer Liffey and Tolka estuaries are shown in Figure 5.30.

Within the estuary, the fastest current speeds during peak flood flow were located through the harbour entrance and within the harbour approach channel. Localised areas of high current speeds are also identified in the Tolka Estuary around Dublin Port. During peak ebb flow, current speeds exceeded 0.5 m/s over a large section of the Tolka Estuary and Lower Liffey Estuary including the area adjacent to the South Bull Wall. Constrained by the outer harbour walls, this forms a 'jet' of water that discharges into Dublin Bay.

Figure 5.31 to Figure 5.36 show time-series comparisons of observed and modelled near-surface current speed and current direction at ADCP locations within the estuary (see section 4.1.3). The hydrodynamic model successfully captured the variations in current speed over the spring-neap and semi-diurnal tidal cycles. The model provided an excellent replication of the observed current speeds at each of the ADCP locations.

Current directions when compared to the measured data were seen to be less consistent. The directional measurement data for ADCP1 (Liffey) were seen to be "noisy" with a rapid temporal variation likely to be caused by its location on the edge of the deep channel in a location prone to eddies. Similarly, at other sites the directions are less well represented.

Further investigation of this discrepancy highlighted that the tidal component was well represented, however the residual (or non-tidal) signals were less well predicted. It was noted that this was particularly true for the ADCP 1 (Liffey) and ADCP 3 (Clontarf Basin).

Further investigation of the measured data has noted a discrepancy in the current directions. For example, at ADCP 1, the current rose shown in Figure 4.6 illustrates current directions with a more NW-SE dominant axis (going towards). The CTD measurement studies (Ref. /6/) noted that near this location the currents should be aligned more with the east-to-west direction than shown in the ADCP results. In addition, it would be expected that even with the influence of the Tolka the currents at ADCP 1 should be more aligned with the predominant axis of the approach channel.

Figure 5.35 shows a comparison of the measured (ADCP and CTD) and model results for the location around ADCP 1 (Liffey). The model shows a very thin surface layer, with significant differences in current direction from near surface (layer 7, red line in Figure 5.35) and surface (layer 8, pink line in Figure 5.35). Similar discrepancies are seen between the CTD measurements (considered to be representative of the surface) and the ADCP (considered to be representative of near surface due to the side-lobe interference).

For ADCP 3, in the Tolka, there appears to be a consistent 45-degree bias in the directions compared to the model. These are likely due to the rapid spatial variability of directions in this very shallow location. Consequently, it is considered that directions from the measured 2015 ADCP's should be treated with caution when comparing to the model.

Based on this uncertainty, additional data from previous surveys in 2013 for the Alexandra Basin Redevelopment have also been incorporated. This is shown in Figure 5.36 and Figure 5.37, for current speed and direction respectively, and confirms the model validity within the estuary when compared to measurements.

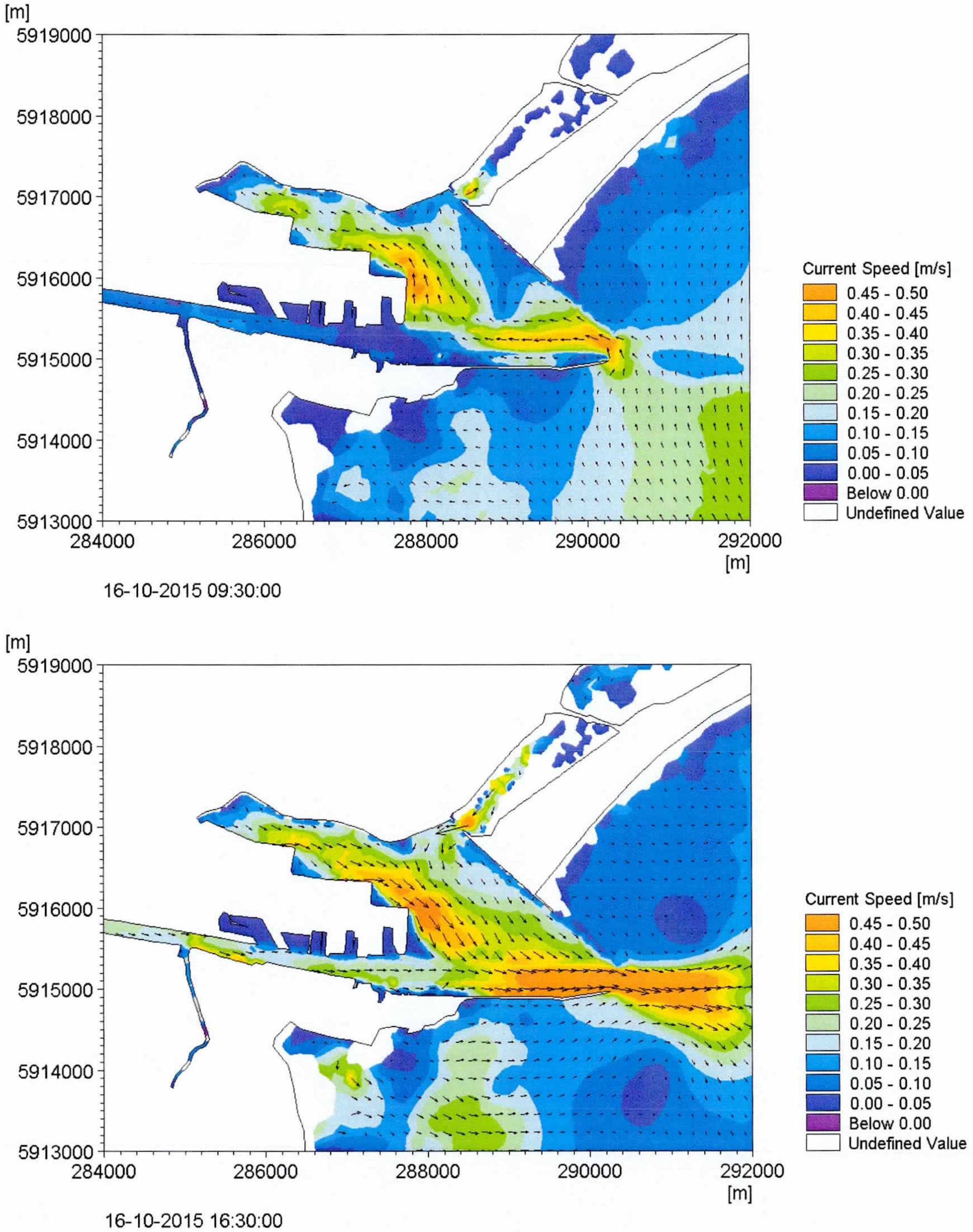

Figure 5.30 Depth-averaged current speeds in the Tolka Estuary and Dublin Port for a near-spring flood tide (upper panel) and near-spring ebb tide (lower panel). Vectors show the direction that the current is flowing towards.

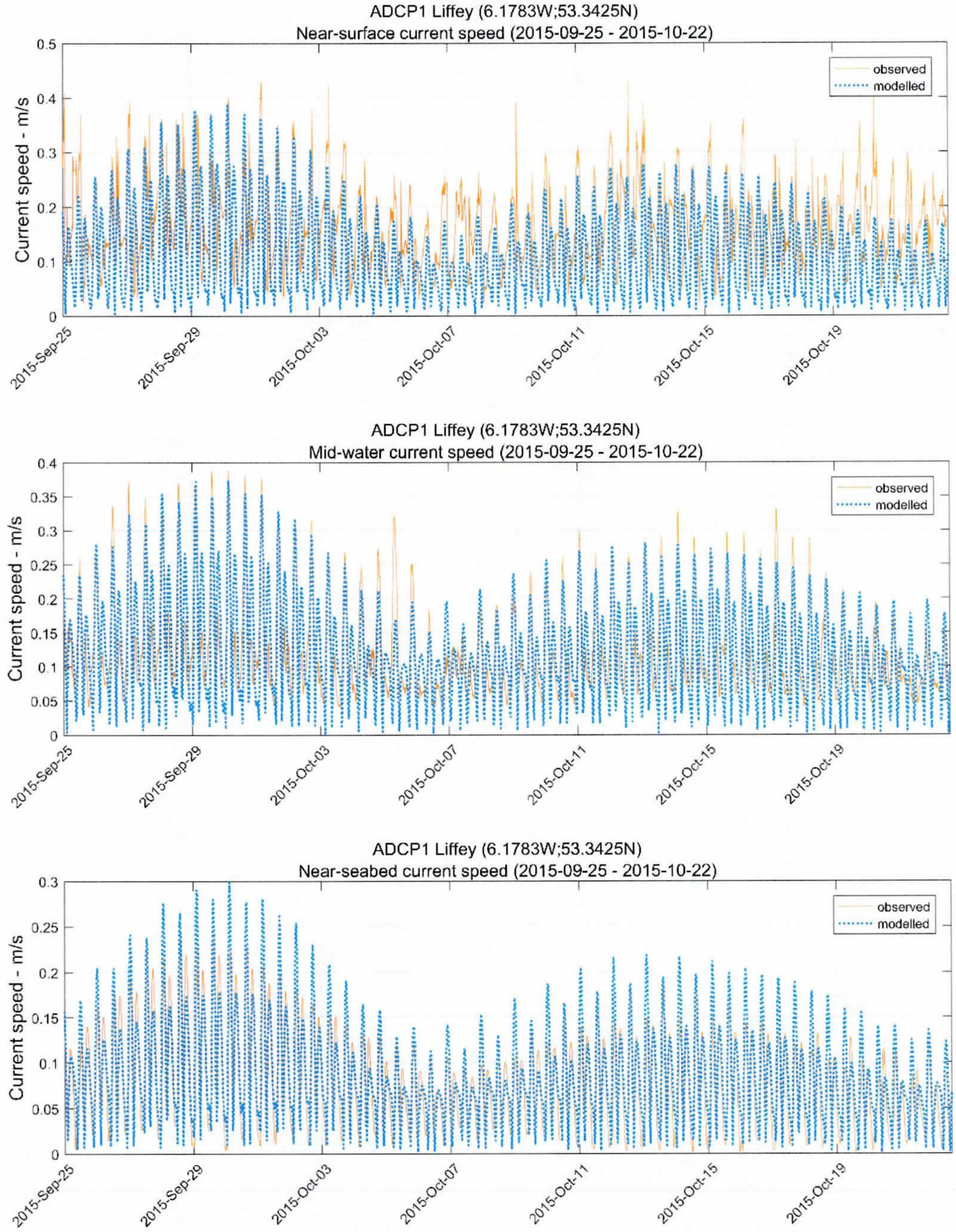

Figure 5.31 Time-series comparison of observed (orange) and modelled (blue) current speed for ADCP1 – Liffey at near-surface (upper panel), mid-water (central panel) and near-seabed (lower panel).

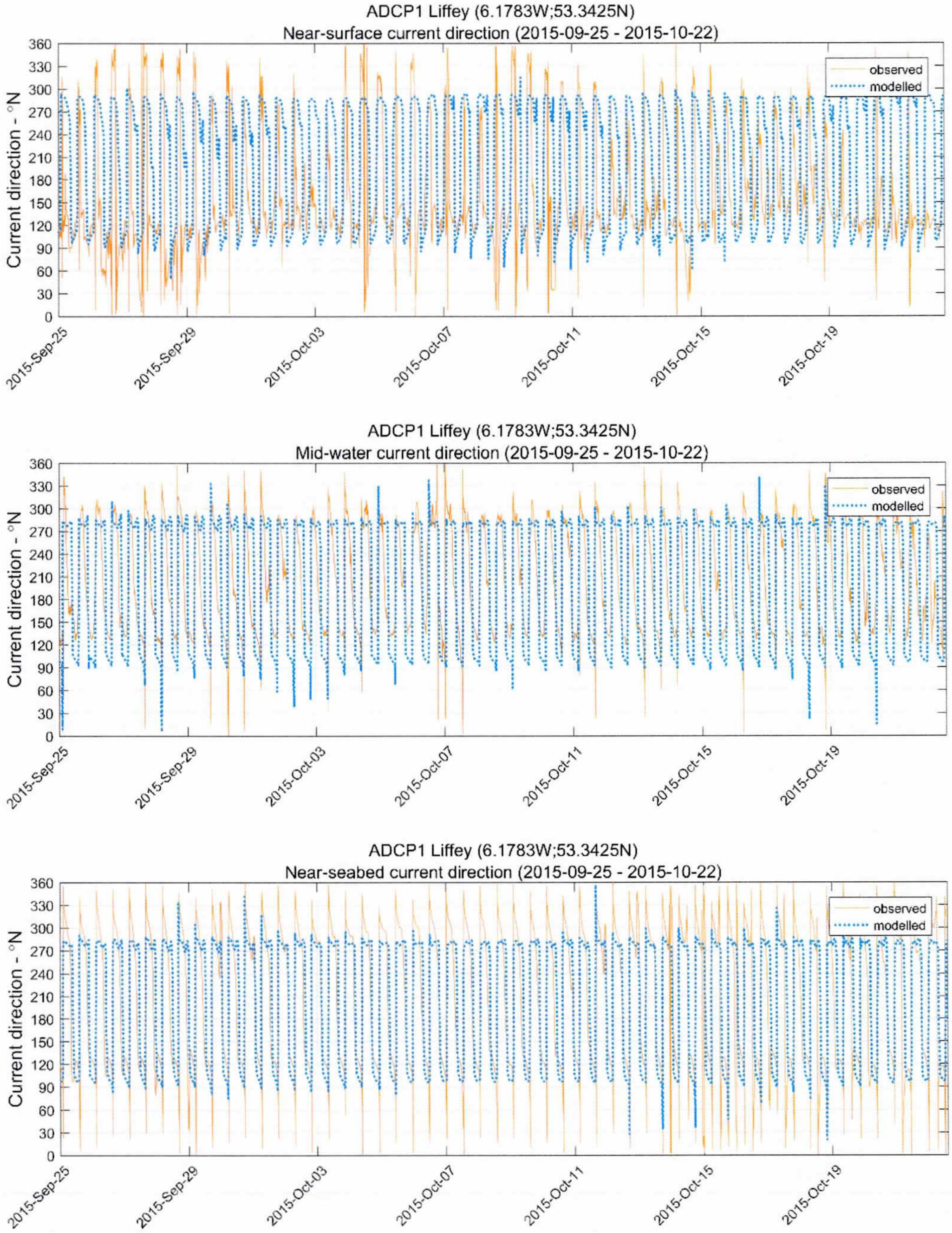

Figure 5.32 Time-series comparison of observed (orange) and modelled (blue) current direction for ADCP1 – Liffey at near-surface (upper panel), mid-water (central panel) and near-seabed (lower panel).

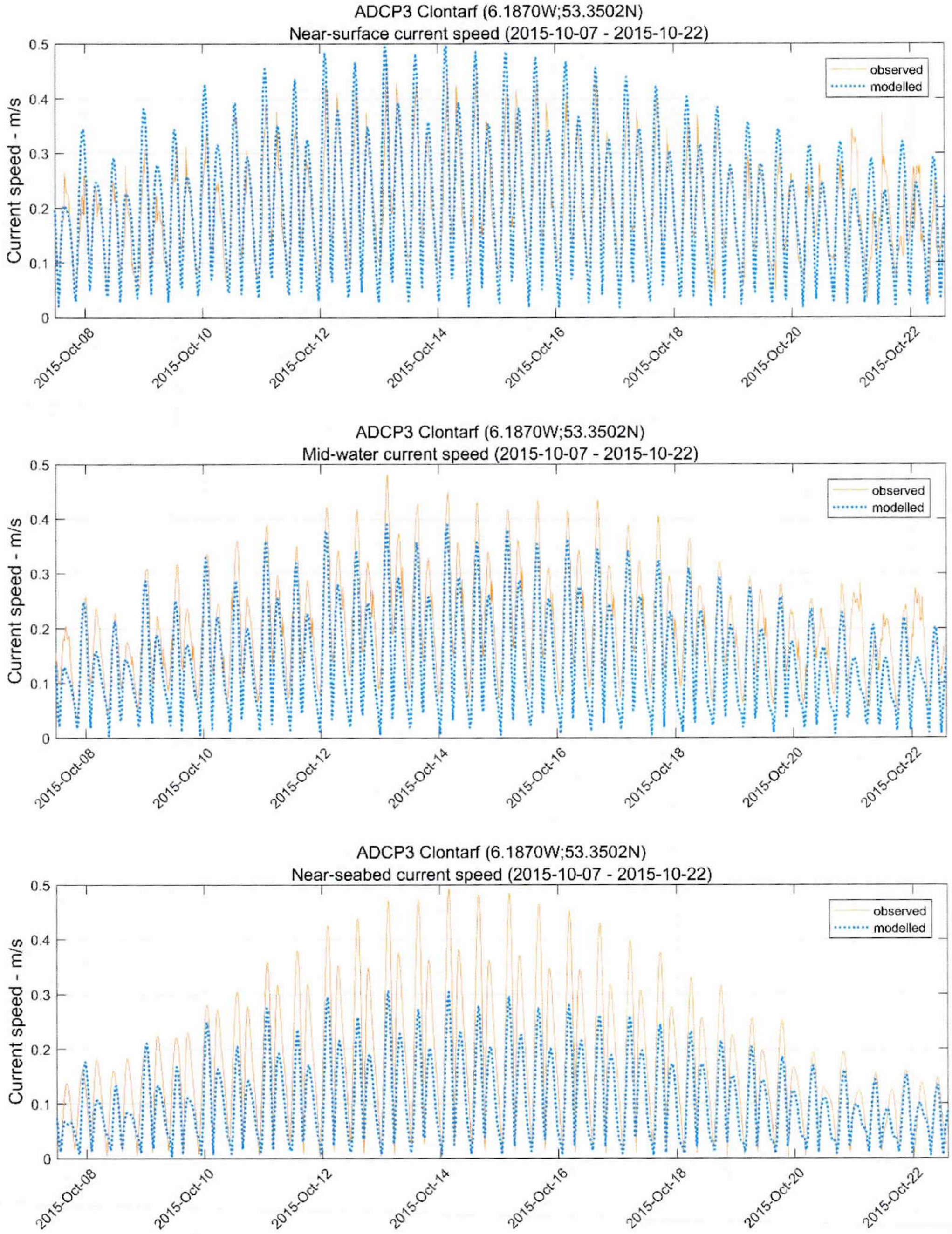

Figure 5.33 Time-series comparison of observed (orange) and modelled (blue) current speed for ADCP3 – Clontarf at near-surface (upper panel), mid-water (central panel) and near-seabed (lower panel).

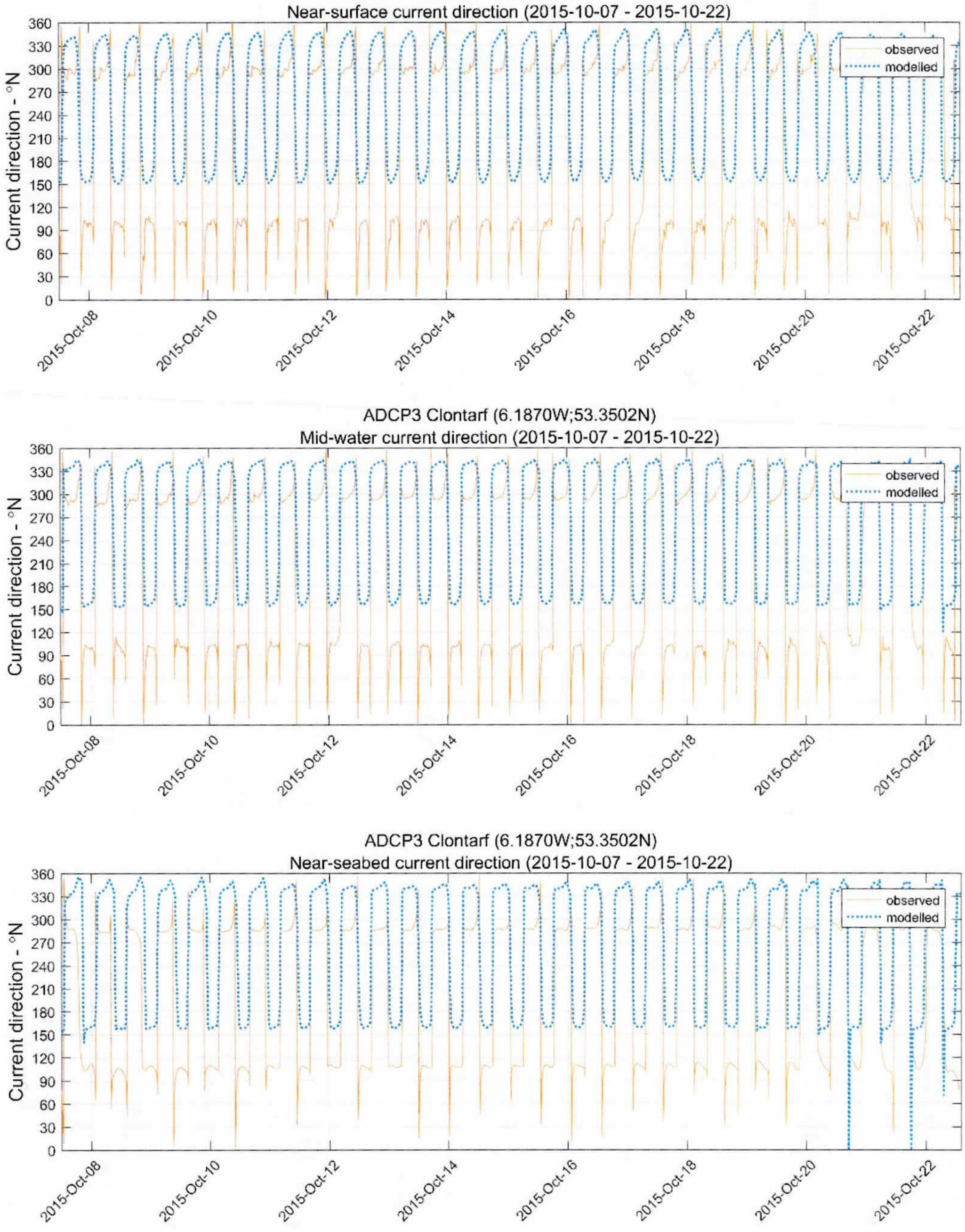

Figure 5.34 Time-series comparison of observed (orange) and modelled (blue) current direction for ADCP3 – Clontarf at near-surface (upper panel), mid-water (central panel) and near-seabed (lower panel).

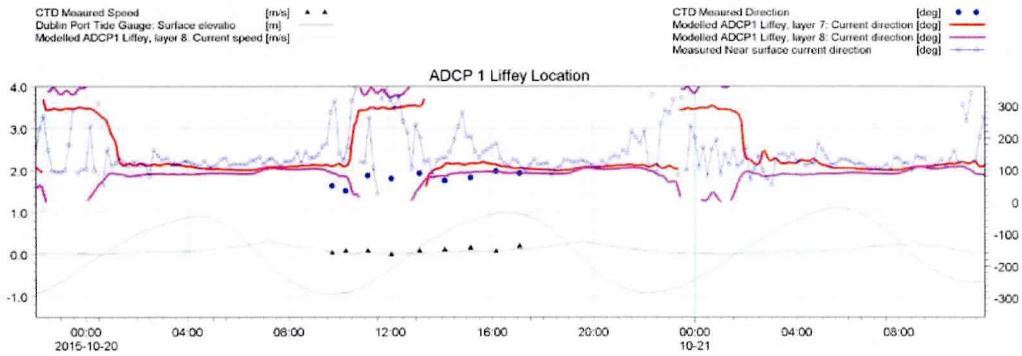

Figure 5.35 Comparison of directions from the measured ADCP (fine blue), measured CTD (blue dots) and the modelled surface (purple) and near surface (red). Also shown are current speed and tidal elevation.

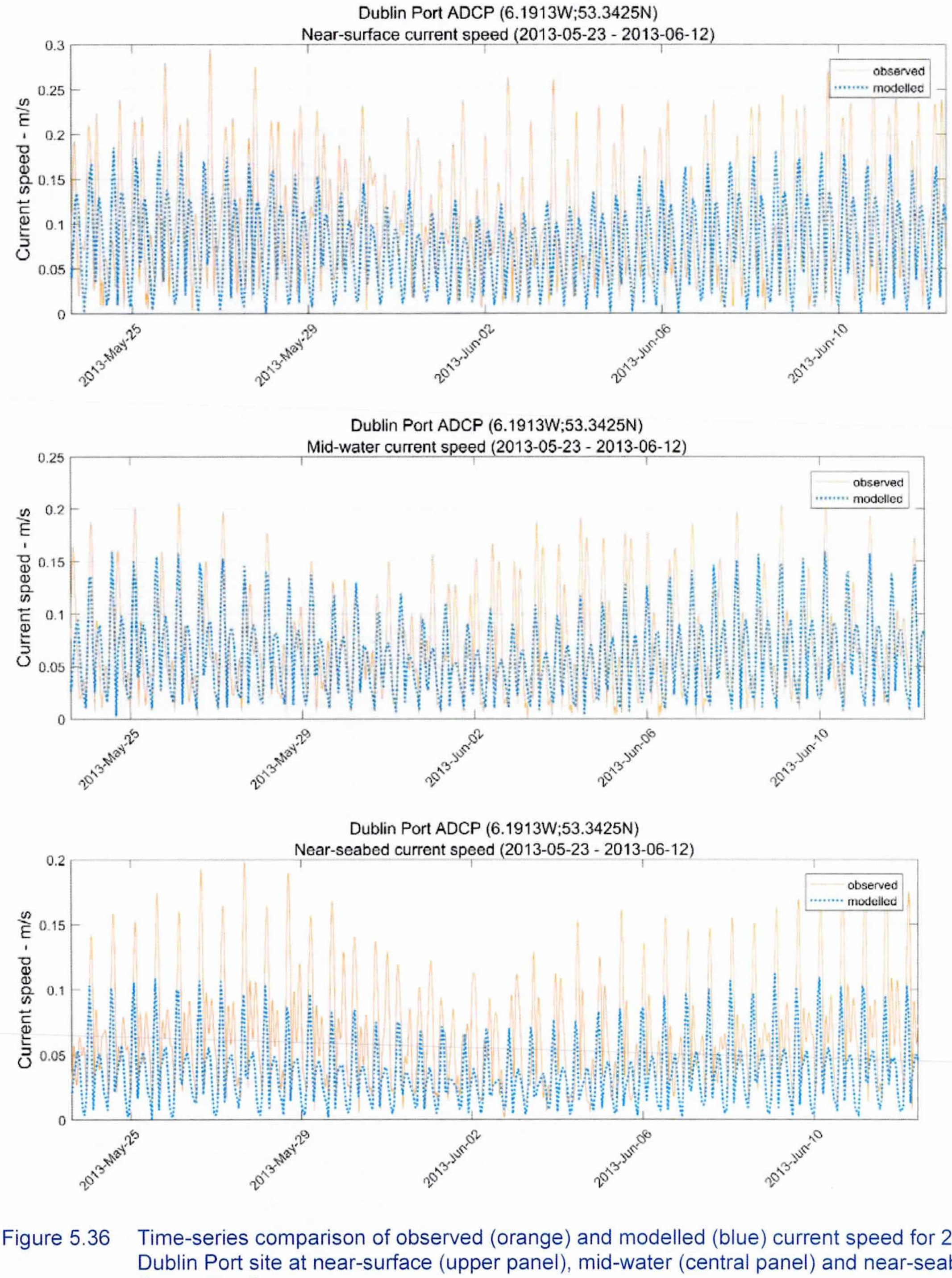

Figure 5.36 Time-series comparison of observed (orange) and modelled (blue) current speed for 2013 Dublin Port site at near-surface (upper panel), mid-water (central panel) and near-seabed (lower panel).

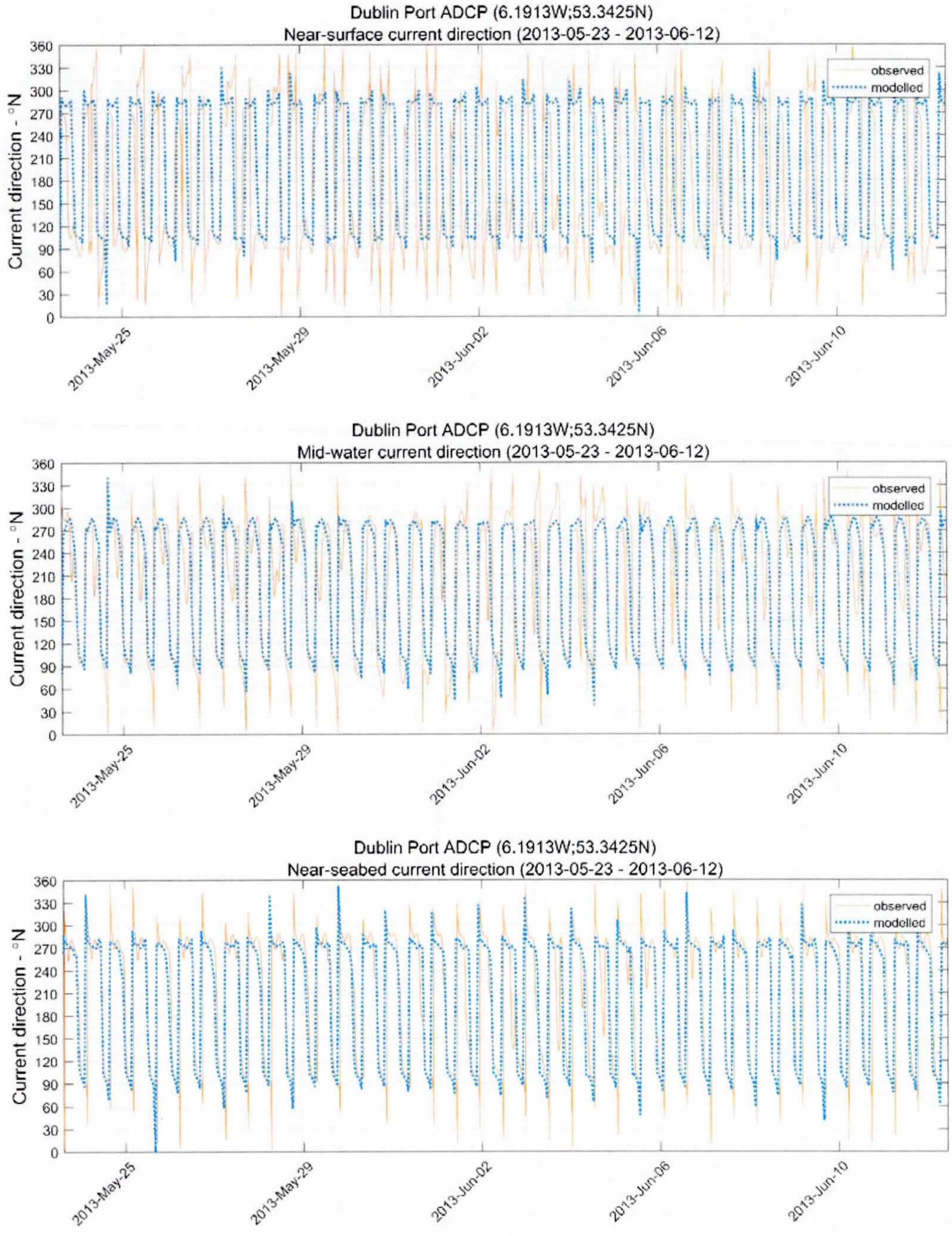

Figure 5.37 Time-series comparison of observed (orange) and modelled (blue) current direction for 2013 Dublin Port site at near-surface (upper panel), mid-water (central panel) and near-seabed (lower panel).

5.5.3 Temperature

As described in section 4.1.3, information on water temperature were available from sensors on the three ADCP's deployed within the Lower Liffey and Tolka estuaries during September and October 2015. The ADCP were bottom-mounted thus the available data represent the water temperature near the seabed.

A comparison of observed and modelled near-seabed water temperatures are shown in Figure 5.38. The hydrodynamic model provided an excellent replication of the observed temperatures and captures the variation in temperature that occur over the tidal cycle with very good accuracy.

There are no criteria for success in validation of water temperature within estuarine waters specified in the UKFWR guidelines (Ref. /11/). The guidelines do state, however that the following should be achieved within coastal waters:

- Temperature to within 0.5 °C.

Table 5.6 shows the validation statistics for near seabed water temperatures at the three ADCP locations within the estuary. The above criteria for water temperature validation was found to be achieved for over 90% of the available period within Dublin Bay. This result gives confidence in the hydrodynamic model's ability to replicate the variation in water temperature in the coastal areas.

Within the estuary the water temperature achieved the coastal criteria for 75% of the time at ADCP1 and 88% of the time at ADCP2. Considering the fact that validation criteria for coastal waters are typically stricter than for estuarine waters, this gives confidence in the model's representation of water temperature within the estuary.

Table 5.6 Model validation statistics for near seabed water temperature at three ADCP locations.

Station	Mean absolute error [°C]	Water temperature ±0.5°C [% of time]
ADCP1 – Liffey	0.3	75
ADCP 2 – Dublin Bay	0.3	90
ADCP 3 - Clontarf	0.3	88

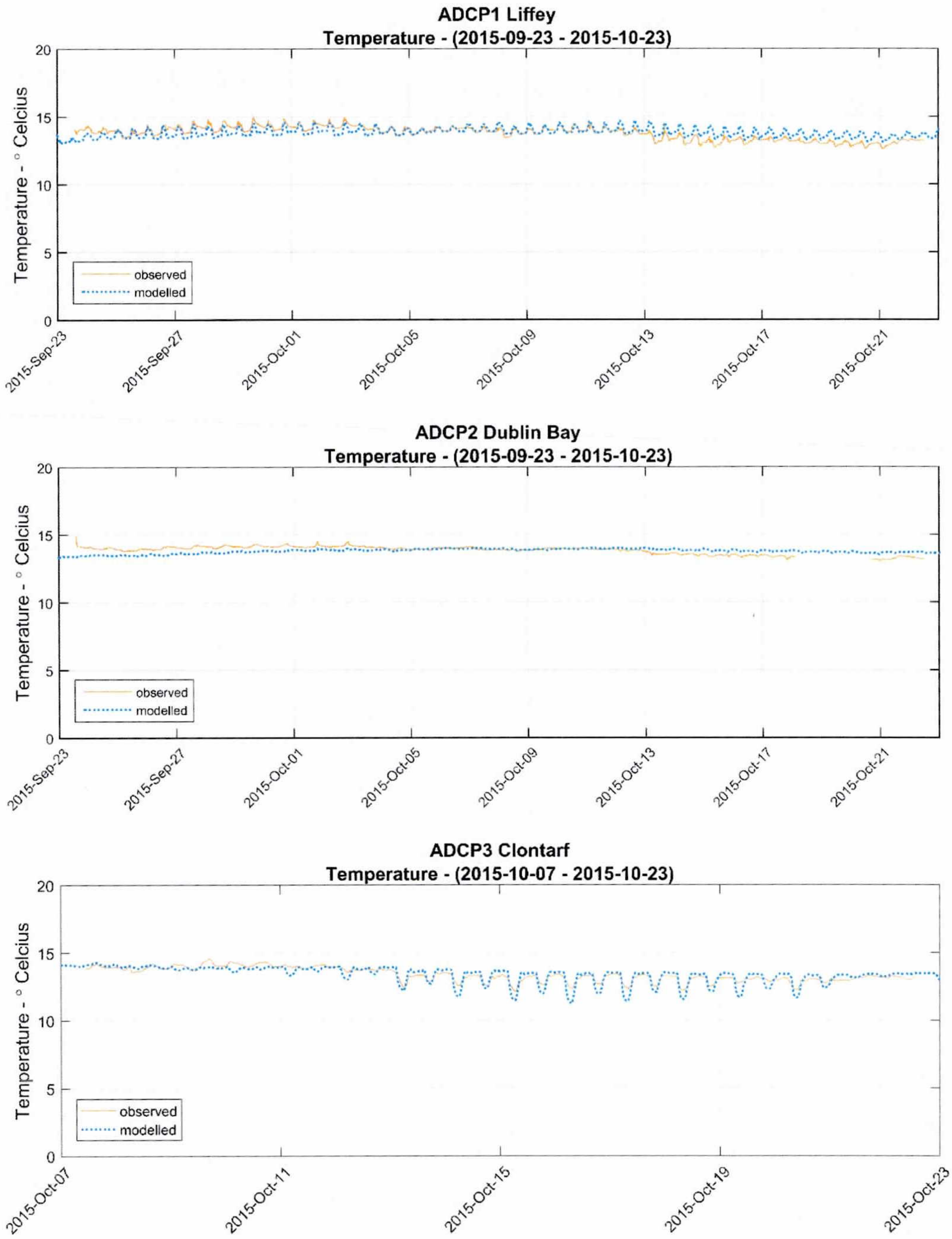

Figure 5.38 Comparison of observed (orange) and modelled (blue) near seabed temperature at location of three ADCP locations during September – October 2015.

5.5.4 Salinity

For estuarine waters, the guidelines for salinity validation by the UKFWR (Ref. /11/) state that the following criteria should be achieved:

- Salinity ±1 PSU at the mouth and head; or
- Salinity ±5 PSU or more in regions of rapid change.

For Dublin Bay South, Figure 5.39 shows that the hydrodynamic model captures the observed salinity profile with very good accuracy. At this location, there is an absence of clear vertical density stratification and salinities are constant. The model salinities at the surface, mid-layer and seabed are within 1 PSU of the observed values, thus satisfying the UKFWR criteria.

For the CTD locations within the Liffey Estuary and Tolka Estuary, the salinity can change rapidly due to freshwater input from the rivers and outfalls and the influence of the tide. Notwithstanding the outlying values for the Liffey Downstream (as previously discussed in section 4.1.4) the modelled salinities are typically within 4-5 PSU of the observed values. This was considered a good level of agreement within these complex estuarine waters. It was noted that the surface salinities show better agreement than the salinities near the seabed, for which the modelled values tended to be slightly overestimated.

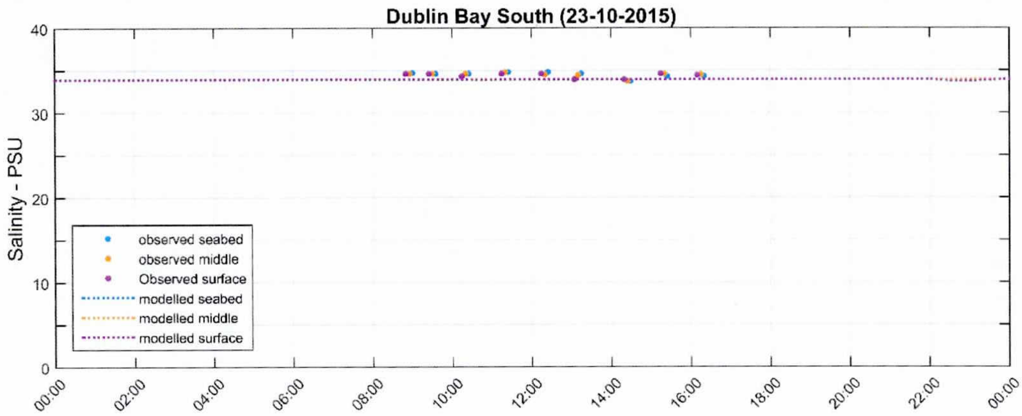

Figure 5.39 Comparison of observed (circles) and modelled (dashed lines) salinity for Dublin Bay South. Results are shown for three depths through the water column, surface waters (blue), middle (orange) and seabed (purple).

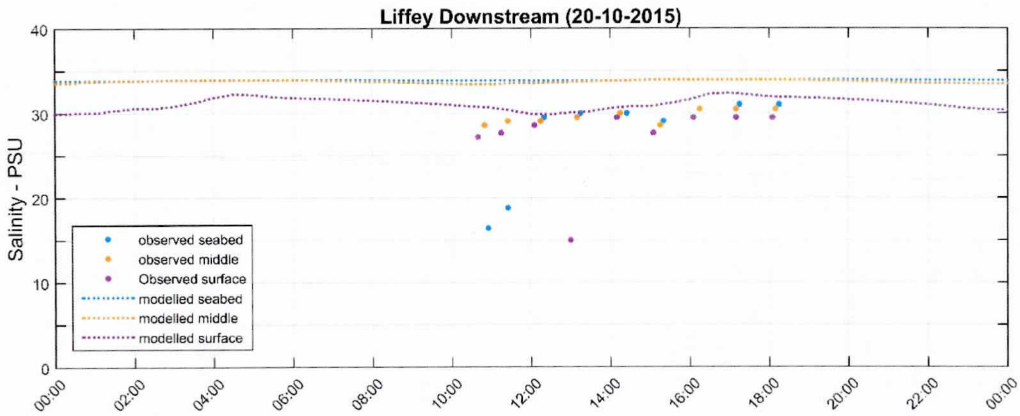

Figure 5.40 Comparison of observed (circles) and modelled (dashed lines) salinity for Liffey Downstream. Results are shown for three depths through the water column, surface waters (blue), middle (orange) and seabed (purple).

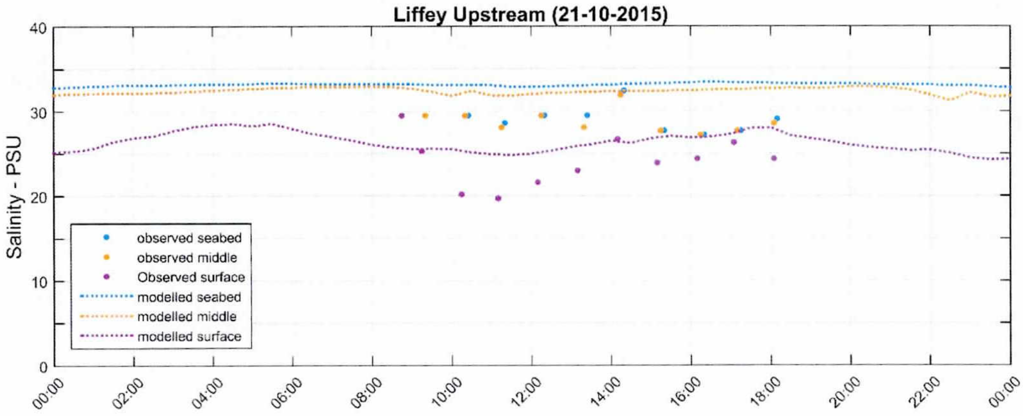

Figure 5.41 Comparison of observed (circles) and modelled (dashed lines) salinity for Liffey Upstream. Results are shown for three depths through the water column, surface waters (blue), middle (orange) and seabed (purple).

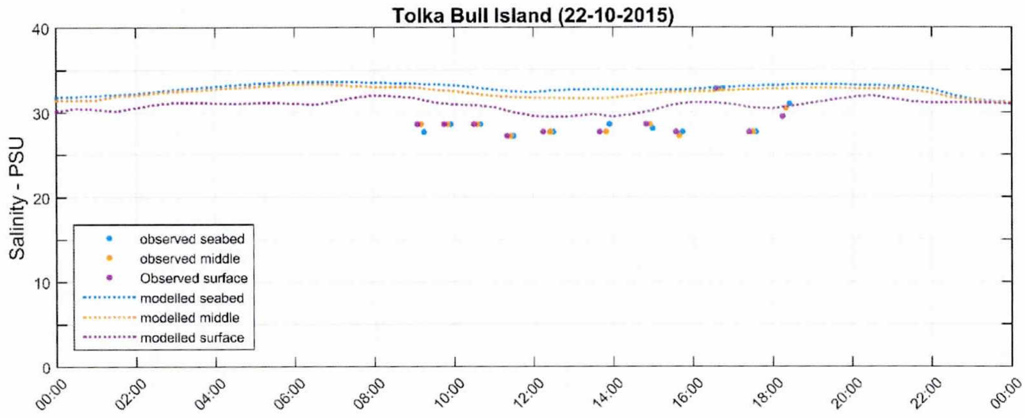

Figure 5.42 Comparison of observed (circles) and modelled (dashed lines) salinity for Tolka Bull Island. Results are shown for three depths through the water column, surface waters (blue), middle (orange) and seabed (purple).

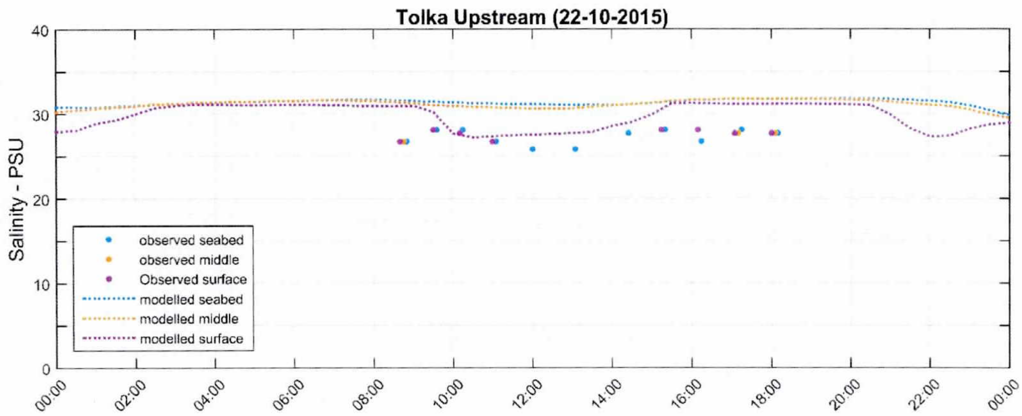

Figure 5.43 Comparison of observed (circles) and modelled (dashed lines) salinity for Tolka Upstream. Results are shown for three depths through the water column, surface waters (blue), middle (orange) and seabed (purple).

5.6 Discussion of Model Calibration

The purpose of the model calibration is to use observed data, that represent the hydrodynamic characteristics of the area being modelled, and to adjust the model parameters considered critical for capturing the physical processes of interest. It should be considered that numerical models are a parameterisation of the driving physical processes and, therefore, the principal concern is whether these parameters are suitably selected for the application.

Overall, the calibration discussed and achieved in Section 5.5 is acceptable for the purposes of comparing the proposed Ringsend WwTP with the existing situation. It was noted that given the complex estuarine processes and the fine balance in these processes, seen from both the collected data and the modelling, that further tasks were required to assess the overall sensitivity of the approach, particularly for direction within the estuary

This section describes and discusses the activities performed during the model sensitivity check and also summarises the dynamics seen in the Liffey and Tolka estuaries. Specifically, the following are addressed:

1. Assess the uncertainties in model inputs, parameters and data used.
2. Conduct a sensitivity assessment of current speeds to factors such as freshwater flow rate and wind.

As discussed in Section 5.5, calibration of the hydrodynamic model was performed based on time-series comparisons between observed and modelled conditions.

5.6.1 Consideration of Uncertainties in Input Parameters

5.6.1.1 Ringsend WwTP Outfall

During initial review of the model approach, it was noted that uniform temperature for the effluent discharged from the Ringsend WwTP was likely to be unrepresentative. Following provision of further information on the effluent water temperature discharged from the existing Ringsend WwTP (daily average temperature and flows rates for the existing Ringsend WwTP outfall for the period January 2014 – September 2016), the MIKE 3 hydrodynamic model was updated to include these observed flow rates and effluent temperature data. It should be noted that temperature variations have a smaller impact on fluid density than variations in salinity, i.e. that the temperature fluctuations will have a limited impact on the vertical mixing.

Figure 5.44 shows a time-series of daily mean discharge and effluent temperature for the full measurement period and Figure 5.45 shows the flow and temperature data used in the calibration period and described further in Section 5.4.6.

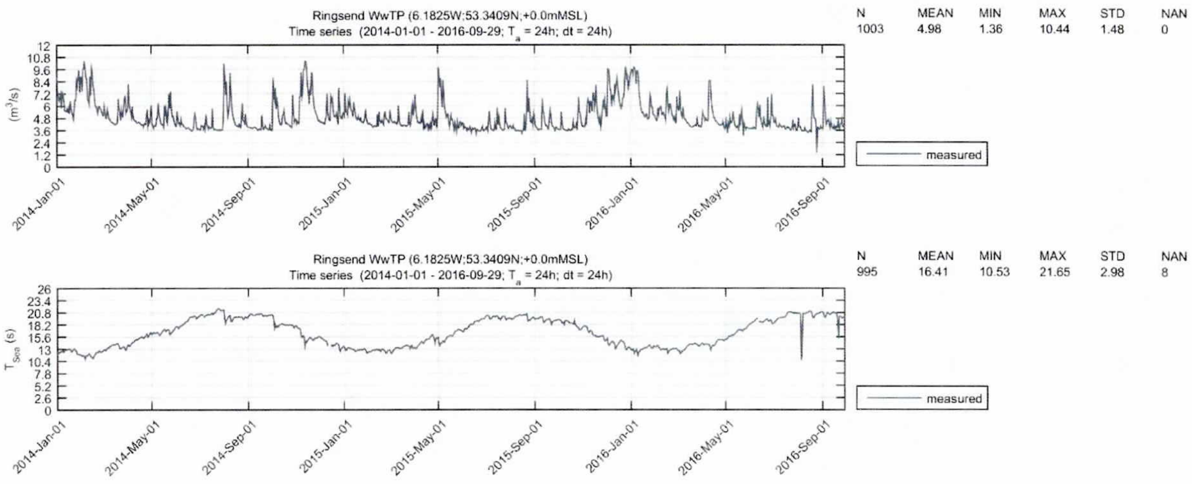

Figure 5.44 Ringsend WwTP daily mean flow rate (upper panel) and effluent temperature (lower panel) for January 2014 – September 2016.

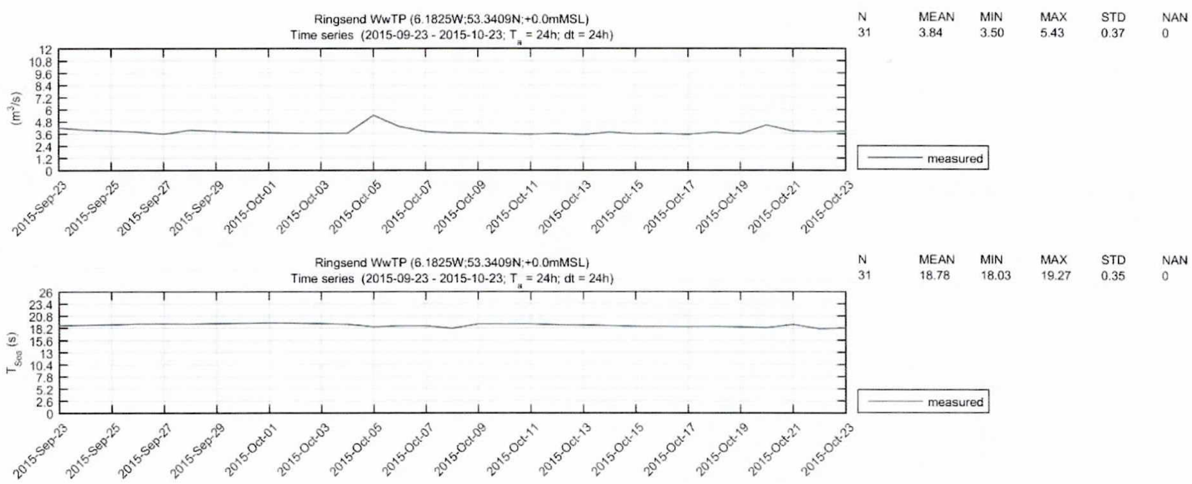

Figure 5.45 Ringsend WwTP daily mean flow rate (upper panel) and effluent temperature (lower panel) for model calibration period (September – October 2015).

5.6.1.2 River Inputs

Flow Rates

Further consideration of the fluvial inputs to the hydrodynamic model was undertaken to assess the relative importance. It was noted in review that a tributary of the River Slang (a tributary of the River Dodder) was not included and hence the Dodder appeared to have a low flow rate.

The reason for the low flow rate for the river Dodder event was that originally daily mean discharges were used as input to the hydrodynamic model. Where available, river flow rates with a higher temporal resolution of 15-minute flow rate data were obtained from EPA hydronet data portal (http://www.epa.ie/hydronet/#Flow). These were available for the following rivers:

- River Dodder at Waldrons Bridge;
- River Slang at Frankfort
- River Tolka at Botanic Gardens
- River Santry at Cadbury's
- River Cammock at Killeen Road

Figure 5.9 to Figure 5.12 shows the updated river flow rates used in the final model calibration period (September – October 2015). No data was available for the Santry during the calibration period so the long-term average flow rate of 0.2 m^3/s was used.

As the River Slang flows into the River Dodder upstream of the source of the River Dodder in the hydrodynamic model, the combined Dodder and Slang flow were used.

Although gauged flow data was available for the River Liffey during the calibration period (from the ESB plant at Leixlip) these data were regarded as insufficient for the study due to their distance from the estuary. Furthermore, no Liffey flow data were available for the summer storm period. Published information from other studies was used to quantify the input. In addition, the large urban area of Dublin discharges through Storm Water Overflows (SWO's) into the estuary. This information was not available from any quantifiable source for the calibration period.

As stated in Section 5.4.5, freshwater inputs (both in terms of quantity and time variability) remain an uncertainty in the hydrodynamic model. A sensitivity assessment was therefore performed to investigate the effects of varying the flow rate in the River Liffey (Section 5.6.2).

Temperatures

In the initial stages of the model calibration uniform water temperatures were applied for the rivers during the calibration period. Following this, further consideration was given to this assumption. It was noted however that no sufficiently detailed (in time and space) data was available during the model calibration period for all locations. Therefore, the fixed values were retained. This is considered suitable as it is unlikely that a small diurnal variation in river temperature will affect the overall density distribution in the entire Lower Liffey Estuary and particularly in the area around the Ringsend WwTP. The final figures used for temperature can be seen in Table 6.3

5.6.1.3 Wind Conditions

The hydrodynamic conditions, particularly near surface current speed and current direction are often strongly dependent on the local wind conditions (speed and direction). This was shown from the measurement data where total reversals in the current direction could be seen to occur during stronger winds. Typically, in areas where the tidal currents are small, the wind can be the dominant force for surface currents.

The model sensitivity to wind conditions was assessed by including wind forcing in the hydrodynamic model. The wind input was taken from the Dublin Bay Smart Buoy (see Figure 5.46 below for the variation during the period).

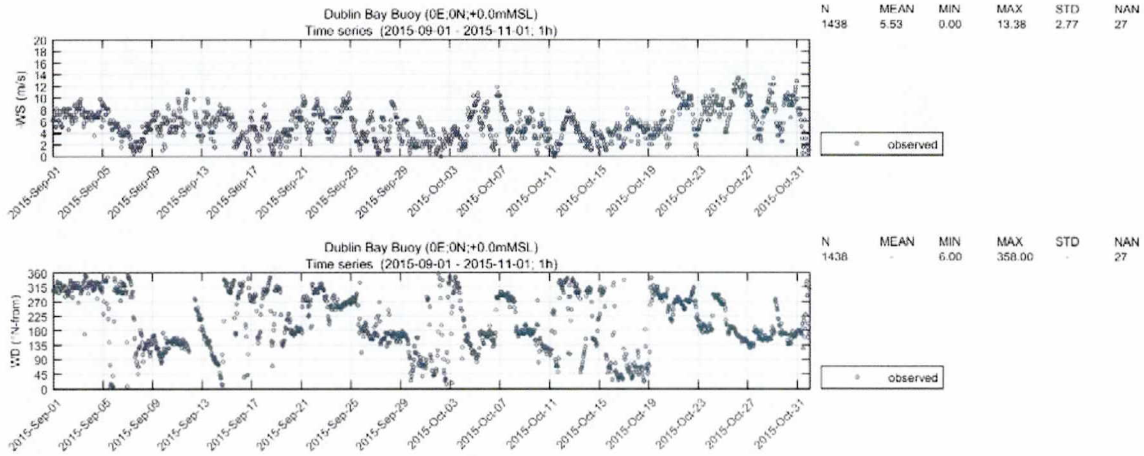

Figure 5.46 Dublin Bay Smart buoy wind conditions during model calibration period (September – October 2015).

5.6.2 Sensitivity Assessment

A model exercise was performed to test the sensitivity of the hydrodynamics (current speed and current direction) to:

- Varying freshwater flows in the river Liffey; and
- The effects of wind on surface flows

Flow Rate in the River Liffey

In the calibration model, the flow rate in the River Liffey was set at 15 m^3/s. This value was in line with previous studies in the area (e.g. the Alexandra Basin Redevelopment EIS). However, it has been acknowledged that the Liffey flow rate was an area of uncertainty in the hydrodynamic model.

The hydrodynamic model was therefore run for three different Liffey flow rates:

- Low flow: 7.5 m^3/s
- Medium flow: 15 m^3/s
- High flow: 30 m^3/s

Figure 5.47 compares the resulting current speed and current direction at the location of the ADCP1 (Liffey) at mid water column. The current speed and current directions showed sensitivity to the Liffey flow rate at this location, particularly with respect to current direction. This supports the assumption that knowing the input of freshwater into the system is critical to the final distribution of fresh and salt water at any given moment. Additionally, uncertainty over the outflow from the numerous drains and storm water systems within the city, which could provide additional freshwater input was considered unquantifiable in the calibration stage. The strength of the vertical mixing of freshwater with the saline water also has a significant impact on the current speed and directions and remains a calibration parameter.

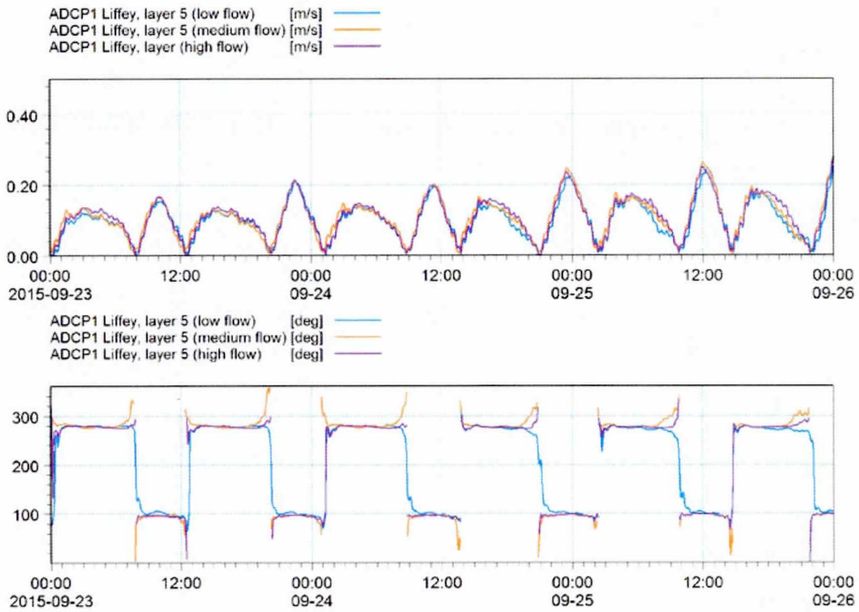

Figure 5.47 Sensitivity of ADCP 1 – Modelled Liffey current speed (upper panel) and current direction (lower panel) to the freshwater flow from the River Liffey.

Wind Speeds

A sensitivity test was performed to assess the influence of wind forcing on the hydrodynamics, particularly the current directions. The hydrodynamic model was re-run with wind conditions as measured at the Dublin Bay Smart Buoy (see Figure 5.46).

Figure 5.48 shows the current speed and direction at ADCP3 - Clontarf with and without the inclusion of wind forcing. It is shown that wind forcing did not have a significant effect (i.e. large reversals) on the current direction for this location, however there were some minor changes to speed and direction associated with this parameter.

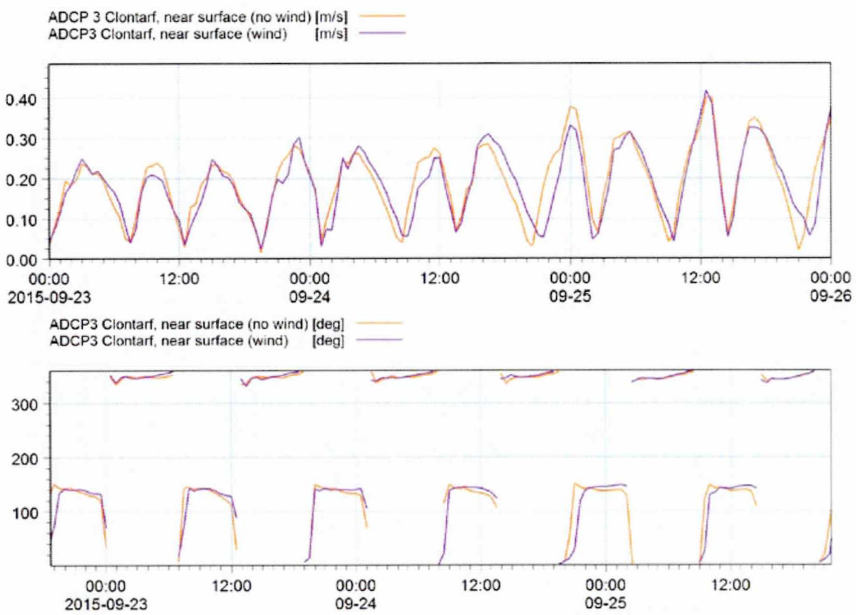

Figure 5.48 Sensitivity of ADCP 3 – Clontarf on near–surface current speed (upper panel) and near-surface current direction (lower panel) to wind forcing.

A more detailed plot is shown below for the effect of wind speeds at ADCP 1 (Liffey). This shows that the model does respond to the input of wind forcing. During the south-westerly winds the surface current direction is aligned with directions going to the east when compared to the model run without wind. With the change in wind direction on the afternoon of the 25th September the model results also show a change in the pattern of the surface flows. This figure also shows the large variability in the measured current direction near surface.

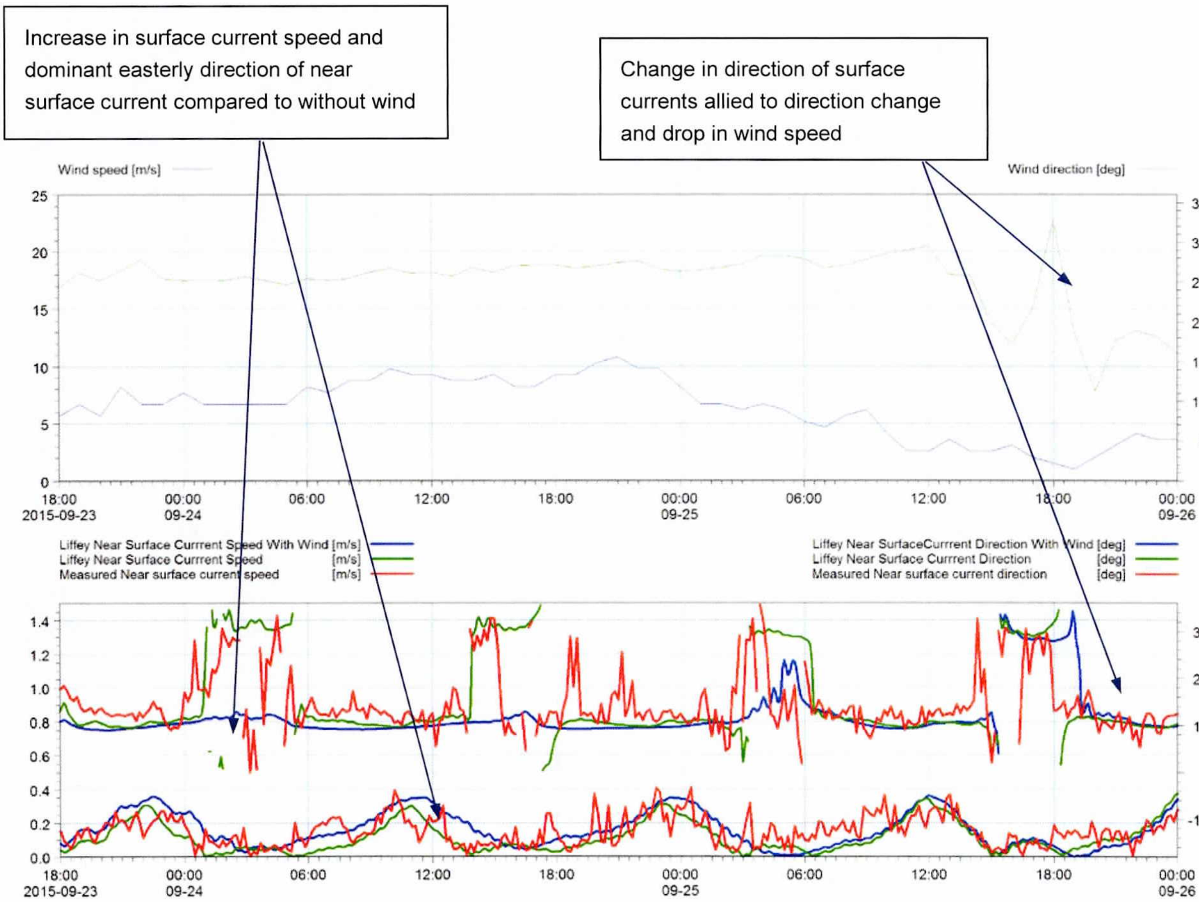

Figure 5.49 Detailed comparison at ADCP 1 of measured (red lines), model (green lines), model with wind (blue lines). The wind speed and directions are shown in the panel above.

Whilst this suggests that the model responds well to wind input, it also shows that to achieve closer parity with the measurements, a significantly more detailed wind measurement or wind model would be required.

5.6.3 Discussion of sensitivity assessment & additional information

From the results in Section 5.5 and a snapshot below, it has been shown that the model produces a distinct surface flow which has an extended period in the ebb direction. Otherwise the model shows a dominant flood flow at depth related to the density structure and the tide, with the ebb being for a relatively shorter period. The measurements show a large amount of variability, indicative of the relatively weak currents and the variable effects of stratification. Variability in direction of ~180 degrees is possible over timescales of 15 – 30 minutes, suggesting significant non-tidal factors.

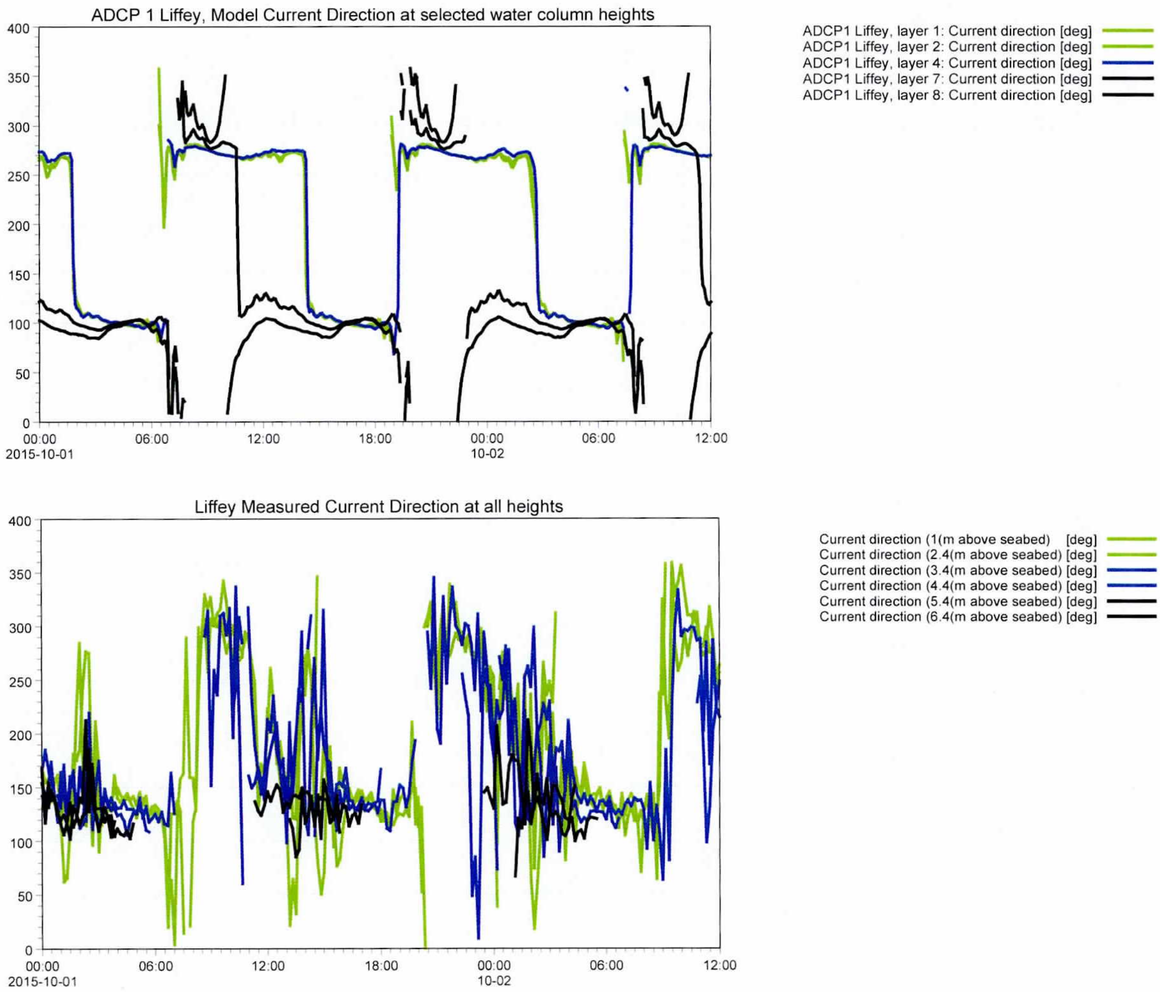

Figure 5.50 Detailed comparison at ADCP 1 of measured (bottom) and model (top) (green lines) through the water column for a selected period.

In addition, in this location the measurements show that at high water, the tide slowly turns from the flood dominant west/north-west direction to a south-east/east direction. The measured ebb flow is noted to be significantly more southward than the model, pushing flow against the Great South Wall.

Significantly, the measured CTD data (Figure 5.35) from further into the main channel and slightly upstream generally supports the model assessment of the dominant surface current direction at this location.

Further discussion of the model outputs in this location are shown in Figure 5.51. At the surface there is a noticeable density gradient extending into the entrance to the Tolka. In addition, there is the presence of an eddy immediately downstream. There is also a 180 degree separation in flow between the area immediately to the south of the ADCP location and the area to the north at the surface, which becomes less pronounced with depth.

The transects of current direction illustrates the very rapid variation, both with depth and with space, of the directions. IP2 on the transect is approximately in the position of the ADCP and small variations in the position of the ADCP can be seen to likely have a large effect on the directions. Additionally, the ADCP measurements showing a more pronounced south-east direction could be due to its location on the edge of one of the eddies, where the tide is deflected more southwards. These eddies are relatively persistent and are controlled by both the Ringsend outfall structure and the balance of freshwater flow, as well as the ebb and flood of the tide.

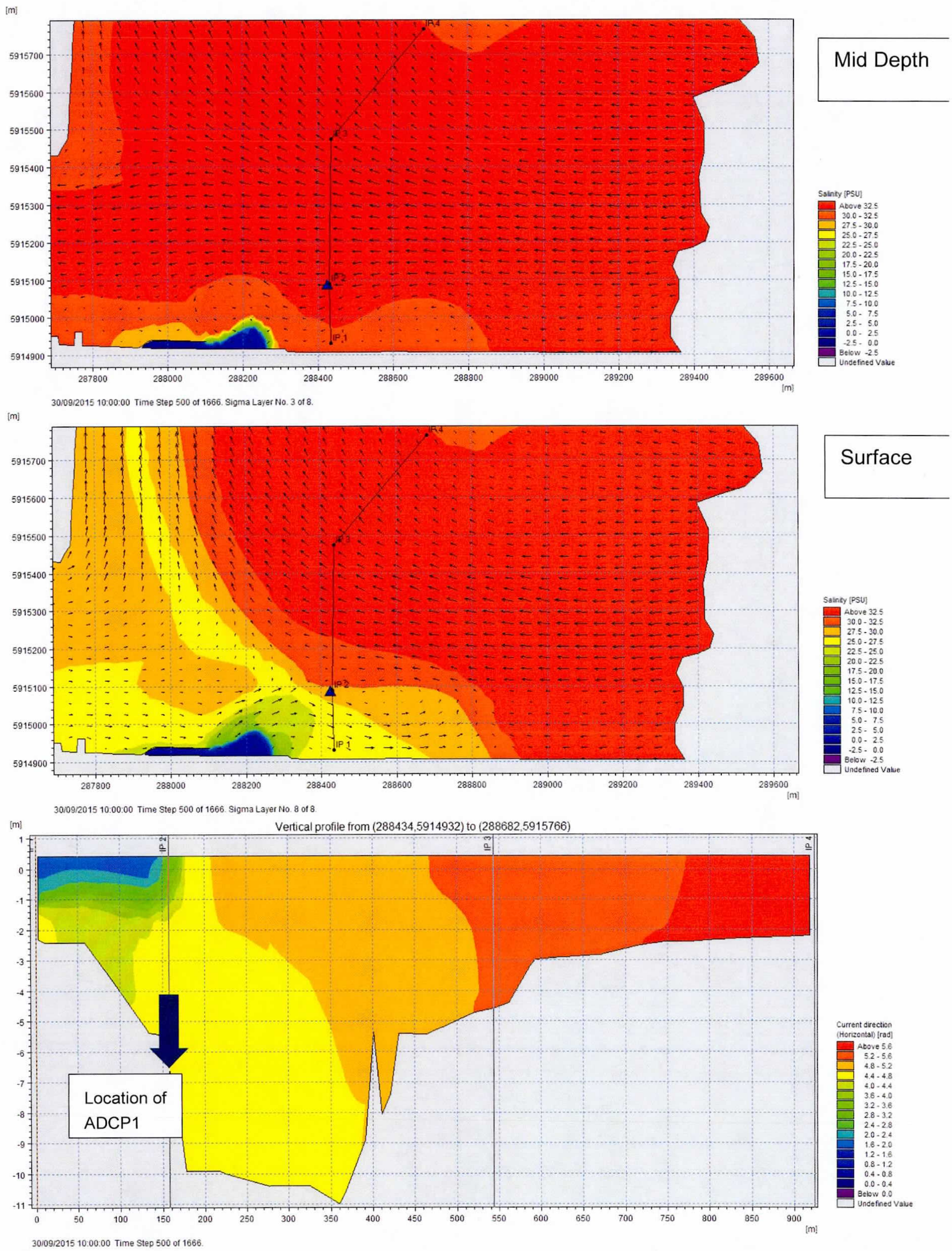

Figure 5.51 Transect through the Lower Liffey Estuary showing the structure of the water column both with respect to salinity and directionality. Top image shows the salinity (colours) and current speed and direction (vectors) at mid depth, the middle image shows the same at the surface and the lower image shows a cross section showing horizontal current direction in a vertical slice. Updated ADCP location is shown as a blue triangle.

Comparison with other data/studies in the area was requested and efforts were made to locate the 2013 ADCP data for the Alexandra Basin Redevelopment. This data has been included in the calibration section, with the results discussed in Section 5.5.2.

In addition, outputs from the Alexandra Basin model study for location S1 in close proximity to the 2015 Liffey output confirms the current model dominant axis being ~100 degrees for the ebb and ~275 for the flood, aligned with the main channel axis.

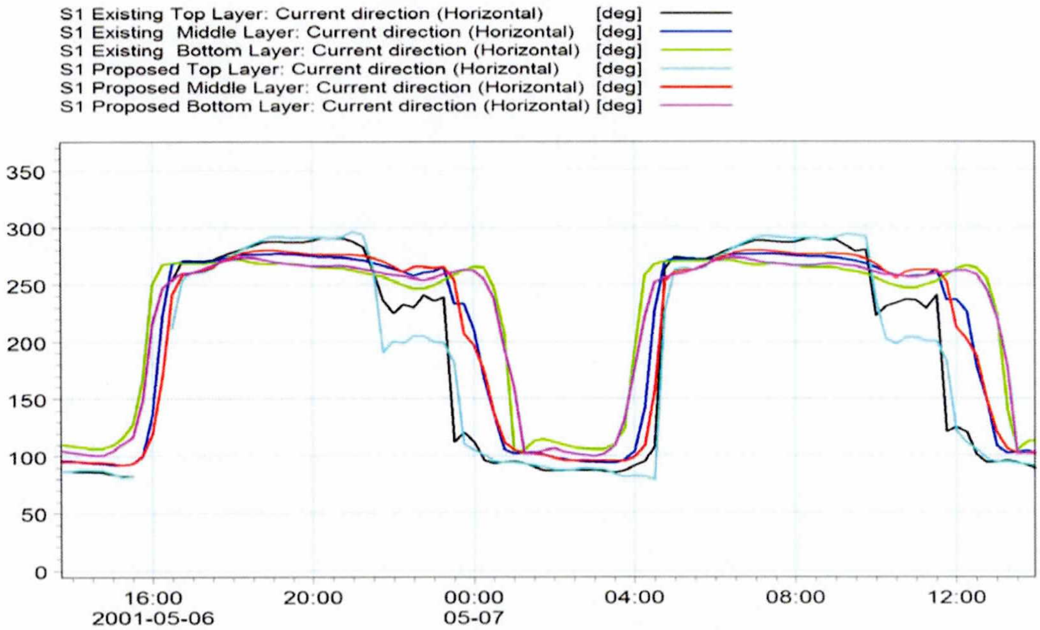

Figure 5.52 Directional outputs from RPS model as part of Alexandra Basin submission reponse to requests for further infomarion (Appendix F1 – Figure F2.1)

Finally, a visual comparison between aerial satellite photography and the surface salinity distribution from the hydrodynamic model show that the surface plume position is closely matched (Figure 5.53). This provides further confirmation of the overall suitability of the hydrodynamic model with respect to the dispersion of surface waters from the Ringsend WwTP outfall.

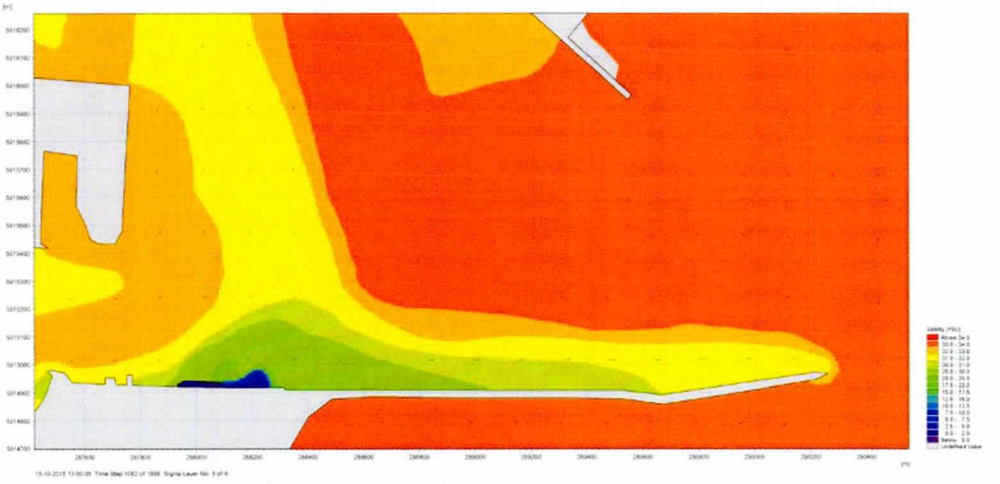

Figure 5.53 Comparison of satellite photo of the estuary and the modelled surface salinity distribution for similar states of the tide.

5.6.4 Conclusions of Model Calibration Discussion

The Lower Liffey Estuary comprises a deep dredged channel with freshwater inputs from the urban area of Dublin and the Rivers Liffey, Dodder, Slang and Cammock resulting in a complicated area as flows in the deep channel are significantly different in magnitude and direction to the upper reaches of the Liffey and on the margins. The Tolka Estuary comprises a broadly shallow area, which has extensive areas that dry out at low water. Dublin Bay is a crescent shaped bay, with gently changing characteristics as the tide circulates around it.

Within the Lower Liffey Estuary the dominant processes are the movement of the tide and the control of the position of the boundary between fresh and salt water. The position of this boundary, both spatially and vertically is very dynamic, varying with the tide, the wind and the freshwater discharges coming from upstream. This is important as this part of the Liffey is the immediate dilution and mixing zone for the discharge from the Ringsend WwTP. It should be noted however that this assessment is primarily focused on the position and movement of the surface plume, as the outfall from Ringsend is considerably fresher than the water it discharges into.

The tide propagating through the entrance to Dublin Harbour, is constrained in the deep channel and therefore flows are primarily bi-directional at depth here, with a net inward flow at depth. Closer to the surface, the control on the direction of flows is largely down to the balance of freshwater flow, prevalent wind conditions and the strength of the tide at that stage. It is to this end that the flow at the surface preferentially enters the Tolka estuary, as it presents a larger tidal prism volume than the deep narrow Liffey channel and port entrance. The tidal dilution is small in the Liffey compared to the Tolka estuary. The main reason for this is that the tidal volume is small compared to the total water volume in the Liffey, while for the Tolka the tidal volume is significant compared to the total water volume. As such the water transport in the Tolka Estuary is largely concerned with the mass transfer of flows in and out of the estuary, while the conditions in the Liffey are controlled by the upstream freshwater releases.

Figure 5.54 Perspective view West into the Lower Liffey Estuary, showing the deep dredged channel and the wide shallow expanse of the Tolka Estuary.

Water entering Dublin Bay is then dispersed by the dominant tidal flows around the bay over the course of several tidal cycles.

A sensitivity assessment was performed to investigate the influence of freshwater flow from the River Liffey and wind forcing.

The conclusions from the calibration and the sensitivity exercises were:

1. The model correctly represents the propagation of the tidal wave from the open boundary into Dublin Bay. The forcing along the model open boundary represent tidal amplitudes and phase consistently and agree between with observed conditions at Kish Bank. In addition, changes to the boundaries do not significantly alter the current directions at the measurement sites.
2. The hydrodynamic model provides an overall excellent representation of the dynamics of Dublin Bay. The distribution of modelled current speed and directions at Burford Bank show excellent correspondence with observed data. This is further supported by comparison between modelled and observed current directions from ADCP transect surveys within Dublin Bay (See section 5.5.2).
3. Additional data on current speeds and directions from upstream of the Ringsend plant in more Estuarine locations shows that the model is representative of the dynamics in this area.
4. The sensitivity of the model was tested for freshwater flows and wind conditions. Whilst neither test altered the comparison between the model result and the measured directions, it is still considered that the exact fluvial input prevailing at the time will influence the final balance of salinity. This in turn will influence the circulation patterns.
5. Wind speeds will likely influence the position of any surface plume. Again, this did not specifically corroborate differences in current direction seen above. However, the additional sensitivity tests undertaken here highlight that as part of the water quality assessment, the scenarios should consider a “representative” wind as a comparison. It is unlikely that real wind measurements for a single point could be utilised in the model as the spatial variability and the urban nature of the river catchment would likely be too complex.
6. As anticipated, no specific change in the model calibration was noted from the inclusion of different flow inputs or starting temperatures for the Ringsend outfall.

It should be noted that all other calibration parameters suggest that the models are representative. In addition, the additional data from periods prior to the 2015 survey data provides further validation of the overall suitability of the model in representing the complex system. Any remaining differences are explainable when considering the position and the complexity of the water column at the measurement sites.

Following this sensitivity assessment, it was suggested that for the scenario modelling, that the remaining uncertainties of wind and freshwater flow were considered in the with/without Ringsend WwTP Upgrade scenarios. As such it was proposed that scenarios 16 and 17 were included to assess the relative impact on the proposed scenarios modelled.

It is considered that this numerical model provides the most up to date and suitable tool for the assessment of the complex hydrodynamic conditions in the estuary and for assessing the fate of any future changes as a result of the changes to the Ringsend WwTP.

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6 Water Quality Modelling Scenarios

6.1 Methodology

The water quality in the Liffey Estuary, Tolka Estuary and Dublin Bay was modelled using the Transport Module within MIKE 3 FM.

The MIKE 3 FM **hydrodynamic module** is the basis for the transport module. The hydrodynamic model characterises the 3-dimensional flow in Dublin Bay and its estuaries due to the influence of tidal forcing and riverine inputs. The hydrodynamic model also simulates the effects of baroclinic flows setup by gradients in water temperature and density within the estuaries.

The **transport model** simulates the spreading and fate of dissolved or suspended substances in an aquatic environment under the influence of the fluid transport and associated dispersion processes. The substance modelled may be of any kind, conservative (inert, non-decaying) or non-conservative (active, decaying over time).

The setup of the water quality model for the Ringsend WwTP upgrade project is described in the following sections. An overview of the modelling scenarios performed is first described in section 6.2. This is followed by details of the setup of the hydrodynamic and transport models in section 6.3 and section 6.4, respectively. The results of a water quality model validation exercise are described in Section 6.5.

6.2 Overview of Modelling Scenarios

The definition of the water quality model runs for input to the Ringsend WwTP Upgrade scheme EIAR were agreed following discussion between JB Barry and Irish Water.

Each model run consisted of a hydrodynamic model scenario and a transport model scenario, which are broadly categorised as representing either:

- The **existing environment**: the present state of water quality environment in the estuaries and Dublin Bay. The period 2013 – 2015 was used as the reference for the baseline scenario to coincide with the most recent measurements in the area.
- The **future discharge environment**: the situation that would exist after the completion of the upgrade works at the Ringsend WwTP.

Hydrodynamic model scenarios

Seventeen (17) hydrodynamic modelling scenarios were performed as summarised in Table 6.1. The model runs were referenced by numbers (1, 2, 3, … 17). The settings of the scenarios were chosen to represent the existing environment or various permutations of the future hydrodynamic and environment. Combinations of the following inputs were simulated in the hydrodynamic model runs:

- Existing environment or future discharge from Ringsend WwTP;
- Normal/peak flow from the Ringsend WwTP;
- Discharge through the Ringsend WwTP storm overflow;
- Seasonal variations in flow rates and temperatures from outfalls, rivers, streams (annual average, summer, winter, or summer storm conditions);
- The operation/non-operation of industrial outfalls in and around Dublin Bay and the estuaries; and
- Infrastructure changes: Repair to the ESB cooling water channel and the Alexandra Basin Redevelopment Scheme.

More information on the specification of these settings is provided in section 6.3.

Water quality model scenarios

The approach for the water quality modelling was to assess the fate of key indicators using conservative and non-conservative tracers. Linear decay rates were applied to simulate the fate of the non-conservative tracers.

A total of ninety-four (94) water quality scenarios were simulated as summarised in Table 6.2. Each scenario was associated with one of the seventeen hydrodynamic model runs. The hydrodynamic model run was denoted by the integer part of the model run number, whereas the fractional part represents the water quality mode run. For example, model numbers 1.01 to 1.16 were based on hydrodynamic model scenario 1.00, and water quality model runs 2.01 to 2.04 were based on hydrodynamic model run 2.00.

As well as different hydrodynamic and thermal conditions (via the choice of hydrodynamic model), the water quality model scenarios involved varying the following model settings:

- Different chemical and biological components, including:
  - Faecal coliforms (Escherichia coli, *E. coli*);
  - DIN (dissolved inorganic nitrogen);
  - Ammonia (total and un-ionised);
  - MRP (Molybdate reactive phosphorus);
  - BOD (biochemical oxygen demand); and
  - Total suspended solids (TSS)
- Existing and future pollutant concentrations from the Ringsend outfall;
- Seasonal variations in pollutant concentrations from the Ringsend outfall
- Average or peak pollutant concentration from the Ringsend outfall;
- With/without background pollutant concentrations (from rivers, streams, canals, and other industrial outfalls)

More information on the specification of these settings is provided in section 6.4.

Table 6.1 Overview of hydrodynamic model scenarios.

Run No.	Description	Hydrodynamic Sources										External Factors	
		River, Streams and Canals	Ringsend Primary Discharge (SW1)	Ringsend Storm Overflow (SW2)	Dublin SWO	Poolbeg Power Station	Synergen Power Station	Covatna WtE Plant	GDD Outfall	Doldrum Bay Outfall	Shanganagh Outfall	ESB Cooling Water Channel	Alexandra Basin
1	Existing Environment – Average	✓	✓	×	×	×	✓	×	×	✓	✓	×	×
2	Existing Environment – Peak Flow	✓	✓	×	×	×	✓	×	×	✓	✓	×	×
3	Existing Environment - Winter	✓	✓	×	×	×	✓	×	×	✓	✓	×	×
4	Existing Environment – Summer	✓	✓	×	×	×	✓	×	×	✓	✓	×	×
5	Existing Environment – Storm Event	✓	✓	✓	✓	×	✓	×	×	✓	✓	×	×
6	Future Discharge – Average	✓	✓	×	×	×	✓	✓	✓	×	✓	×	×
7	Future Discharge – Peak flow	✓	✓	×	×	×	✓	✓	✓	×	✓	×	×
8	Future Discharge – Winter	✓	✓	×	×	×	✓	✓	✓	×	✓	×	×
9	Future Discharge – Summer	✓	✓	×	×	×	✓	✓	✓	×	✓	×	×
10	Future Discharge – Storm Event	✓	✓	✓	✓	×	✓	✓	✓	×	✓	×	×
11	Future Discharge – Average (Poolbeg Power Station On)	✓	✓	×	×	×	✓	✓	✓	×	✓	×	×
12	Future Discharge – Winter (Poolbeg Power Station On)	✓	✓	×	×	✓	✓	✓	✓	×	✓	×	×
13	Future Discharge – Summer (Poolbeg Power Station On)	✓	✓	×	×	✓	✓	✓	✓	×	✓	×	×
14	Future Discharge – Average (ESB Channel Repaired)	✓	✓	×	×	×	✓	✓	✓	×	✓	✓	×
15	Future Discharge – Average (Alexandra Basin Redevelopment)	✓	✓	×	×	×	✓	✓	✓	×	✓	×	✓
16	Future Discharge – Average (Wind Sensitivity)	✓	✓	×	×	×	✓	✓	✓	×	✓	✓	×
17	Future Discharge – Average (Average Flow Sensitivity)	✓	✓	×	×	×	✓	✓	✓	×	✓	×	✓

Table 6.2 Overview of water quality models.

Run No.	Description	Hydrodynamic Model	Ringsend Effluent Concentration, SW1	Ringsend Effluent Concentration, SW2
1.01	BOD – Average	1: Existing Environment – Average	20.6 mg/l	N/A
1.02	BOD – Peak		58. mg/l	N/A
1.03	Suspended Solids – Average		38.2 mg/l	N/A
1.04	Suspended Solids – Peak		129.1 mg/l	N/A
1.05	Not Used		N/A	N/A
1.06	Ammonia		10.3 mg N /l	N/A
1.07	Dissolved Inorganic Nitrogen		14. mg N /l	N/A
1.08	Molybdate Reactive Phosphate		2.49 mg P /l	N/A
1.09	BOD – Average (no background pollutants)		20.6 mg/l	N/A
1.10	BOD – Peak (no background pollutants)		58. mg/l	N/A
1.11	Suspended Solids – Average (no background pollutants)		38.2 mg/l	N/A
1.12	Suspended Solids – Peak (no background pollutants)		129.1 mg/l	N/A
1.13	Not Used		N/A	N/A
1.14	Ammonia (no background pollutants)		10.3 mg N /l	N/A
1.15	Dissolved Inorganic Nitrogen (no background pollutants)		14. mg N /l	N/A
1.16	Molybdate Reactive Phosphate (no background pollutants)		2.49 mg P /l	N/A
2.01	BOD – Peak	2: Existing Environment – Peak Flow	35.5 mg/l	N/A
2.02	Suspended Solids – Peak		79. mg/l	N/A
2.03	BOD – Peak (no background pollutants)		35.5 mg/l	N/A
2.04	Suspended Solids – Peak (no background pollutants)		79. mg/l	N/A
3.01	Dissolved Inorganic Nitrogen	3: Existing Environment – Winter	16.3 mg N /l	N/A
3.02	Molybdate Reactive Phosphate		1.97 mg P /l	N/A
3.03	Dissolved Inorganic Nitrogen (no background pollutants)		16.3 mg N /l	N/A
3.04	Molybdate Reactive Phosphate (no background pollutants)		1.97 mg P /l	N/A
3.05	E. coli		3.00E+5/100ml	N/A
3.06	E. coli (no background pollutants)		3.00E+5/100ml	N/A

Table 6.2 Overview of water quality models.

Run No.	Description	Hydrodynamic Model	Ringsend Effluent Concentration, SW1	Ringsend Effluent Concentration, SW2
4.01	Dissolved Inorganic Nitrogen	4: Existing Environment – Summer	9.8 mg N /l	N/A
4.02	Molybdate Reactive Phosphate		3.12 mg P /l	N/A
4.03	Dissolved Inorganic Nitrogen (no background pollutants)		9.8 mg N /l	N/A
4.04	Molybdate Reactive Phosphate (no background pollutants)		3.12 mg P /l	N/A
4.05	E. coli		1.00E+5/100ml	N/A
4.06	E. coli (no background pollutants)		1.00E+5/100ml	N/A
5.01	E. coli	5: Existing Environment – Storm Event	Time-varying	Time-varying
5.02	E. coli (no background pollutants)		Time-varying	Time-varying
6.01	BOD – Average	6: Future Discharge – Average	12. mg/l	N/A
6.02	BOD – Peak		25. mg/l	N/A
6.03	Suspended Solids – Average		17.5 mg/l	N/A
6.04	Suspended Solids – Peak		35. mg/l	N/A
6.05	Not Used		N/A	N/A
6.06	Ammonia		1. mg N /l	N/A
6.07	Dissolved Inorganic Nitrogen		8. mg N /l	N/A
6.08	Molybdate Reactive Phosphate		0.7 mg P /l	N/A
6.09	Conservative Tracer		N/A	N/A
6.10	BOD – Average (no background pollutants)		12. mg/l	N/A
6.11	BOD – Peak (no background pollutants)		25. mg/l	N/A
6.12	Suspended Solids – Average (no background pollutants)		17.5 mg/l	N/A
6.13	Suspended Solids – Peak (no background pollutants)		35. mg/l	N/A
6.14	Not Used		N/A	N/A
6.15	Ammonia (no background pollutants)		1. mg N /l	N/A
6.16	Dissolved Inorganic Nitrogen (no background pollutants)		8. mg N /l	N/A
6.17	Molybdate Reactive Phosphate (no background pollutants)		0.7 mg P /l	N/A
6.18	BOD – 3 Day Untreated Discharge		240. mg/l	N/A

Table 6.2 Overview of water quality models.

Run No.	Description	Hydrodynamic Model	Ringsend Effluent Concentration, SW1	Ringsend Effluent Concentration, SW2
7.01	BOD – Peak	7: Future Discharge – Peak Flow	21.7 mg/l	N/A
7.02	Suspended Solids – Peak	7: Future Discharge – Peak Flow	21.9 mg/l	N/A
7.03	Conservative Tracer		N/A	N/A
7.04	BOD – Peak (no background pollutants)		21.7 mg/l	N/A
7.05	Suspended Solids – Peak (no background pollutants)		9.7 mg N /l	N/A
8.01	Dissolved Inorganic Nitrogen	8: Future Discharge – Winter	0.7 mg P /l	N/A
8.02	Molybdate Reactive Phosphate		9.7 mg N /l	N/A
8.03	Dissolved Inorganic Nitrogen (no background pollutants)		0.7 mg P /l	N/A
8.04	Molybdate Reactive Phosphate (no background pollutants)		3.00E+5/100ml	N/A
8.05	E. coli		3.00E+5/100ml	N/A
8.06	E. coli (no background pollutants)		9.7 mg N /l	N/A
9.01	Dissolved Inorganic Nitrogen	9: Future Discharge – Summer	6.3 mg N /l	N/A
9.02	Molybdate Reactive Phosphate		0.7 mg P /l	N/A
9.03	Dissolved Inorganic Nitrogen (no background pollutants)		6.3 mg N /l	N/A
9.04	Molybdate Reactive Phosphate (no background pollutants)		0.7 mg P /l	N/A
9.05	E. coli		1.00E+5/100ml	N/A
9.06	E. coli (no background pollutants)		1.00E+5/100ml	N/A
10.01	E. coli	10: Future Discharge – Storm Event	100,000/100ml	Time-varying
10.02	E. coli (no background pollutants)		100,000/100ml	Time-varying
11.01	BOD	11: Future Discharge – Average (Poolbeg Power Station On)	12. mg/l	N/A
11.02	Suspended Solids		17.5 mg/l	N/A
11.03	Not Used		N/A	N/A
11.04	Ammonia		1. mg N /l	N/A
11.05	Dissolved Inorganic Nitrogen		8. mg N /l	N/A
11.06	Molybdate Reactive Phosphate		0.7 mg P /l	N/A
11.07	Conservative Tracer		N/A	N/A

Table 6.2 Overview of water quality models.

Run No.	Description	Hydrodynamic Model	Ringsend Effluent Concentration, SW1	Ringsend Effluent Concentration, SW2
11.08	BOD (no background pollutants)		12. mg/l	N/A
11.09	Suspended Solids (no background pollutants)		17.5 mg/l	N/A
11.10	Not Used	11: Future Discharge – Average (Poolbeg Power Station On)	N/A	N/A
11.11	Ammonia (no background pollutants)		1. mg N /l	N/A
11.12	Dissolved Inorganic Nitrogen (no background pollutants)		8. mg N /l	N/A
11.13	Molybdate Reactive Phosphate (no background pollutants)		0.7 mg P /l	N/A
12.01	Dissolved Inorganic Nitrogen	12: Future Discharge – Winter (Poolbeg Power Station On)	9.7 mg N /l	N/A
12.02	Molybdate Reactive Phosphate		0.7 mg P /l	N/A
12.03	Dissolved Inorganic Nitrogen (no background pollutants)		9.7 mg N /l	N/A
12.04	Molybdate Reactive Phosphate (no background pollutants)		0.7 mg P /l	N/A
12.05	E. coli		3.00E+5/100ml	N/A
12.06	E. coli (no background pollutants)		3.00E+5/100ml	N/A
13.01	Dissolved Inorganic Nitrogen	13: Future Discharge – Summer (Poolbeg Power Station On)	6.3 mg N /l	N/A
13.02	Molybdate Reactive Phosphate		0.7 mg P /l	N/A
13.03	Dissolved Inorganic Nitrogen (no background pollutants)		6.3 mg N /l	N/A
13.04	Molybdate Reactive Phosphate (no background pollutants)		0.7 mg P /l	N/A
13.05	E. coli		1.00E+5/100ml	N/A
13.06	E. coli (no background pollutants)		1.00E+5/100ml	N/A
14.01	Conservative Tracer	14: Future Discharge – Average (ESB Channel Repaired)	N/A	N/A
15.01	Conservative Tracer	15: Future Discharge – Average (Alexandra Basin Redevelopment)	N/A	N/A
16.01	Conservative Tracer	16: Future Discharge – Average (Wind Sensitivity)	N/A	N/A

Table 6.2 Overview of water quality models.

Run No.	Description	Hydrodynamic Model	Ringsend Effluent Concentration, SW1	Ringsend Effluent Concentration, SW2
17.01	Conservative Tracer	17: Future Discharge – Average (Flow Sensitivity)	N/A	N/A

6.3 Hydrodynamic Model

The setup and calibration of a 3-dimensional hydrodynamic and thermal model of the project site was described in Section 5.

The calibration of the hydrodynamic model was based on a period during September and October 2015, during which observed data on the hydrodynamic and thermal characteristics were available.

For the water quality modelling, a set of seventeen (17) hydrodynamic model scenarios were investigated. These scenarios represented both the existing environment over the baseline period (2013 – 2015, inclusive) and various permutations of the future discharge environment. This required an update to the setup of the hydrodynamic model setup as previously described in section 5. The settings for the hydrodynamic model are described in the following sections.

6.3.1 Sources

Hydrodynamic point sources were specified in the model to capture the effects of flow, temperature and salinity. Point sources include rivers, streams, canals, inlets, wastewater and industrial outfalls in and around Dublin Bay.

Figure 6.1 shows the location of all point sources in the hydrodynamic model scenarios. Not all point sources were included in every scenario: Some (e.g. Rivers, streams and canals) were included in all scenarios, whereas others (e.g. the Ringsend storm water overflow and GDD outfall) were only included in the storm scenario. Table 6.1 summarises which sources were included in the each of the seventeen hydrodynamic model scenarios.

Each source was specified within the model by the following three parameters (either constant in time or time-varying):

- Flow rate – m^3/s;
- Temperature – °C (either absolute or relative to ambient temperature);
- Salinity – PSU (either absolute or relative to ambient temperature); and
- Vertical position in the water column.

All point sources were set to discharge to the surface waters (i.e. upper-most layer) of the hydrodynamic model.

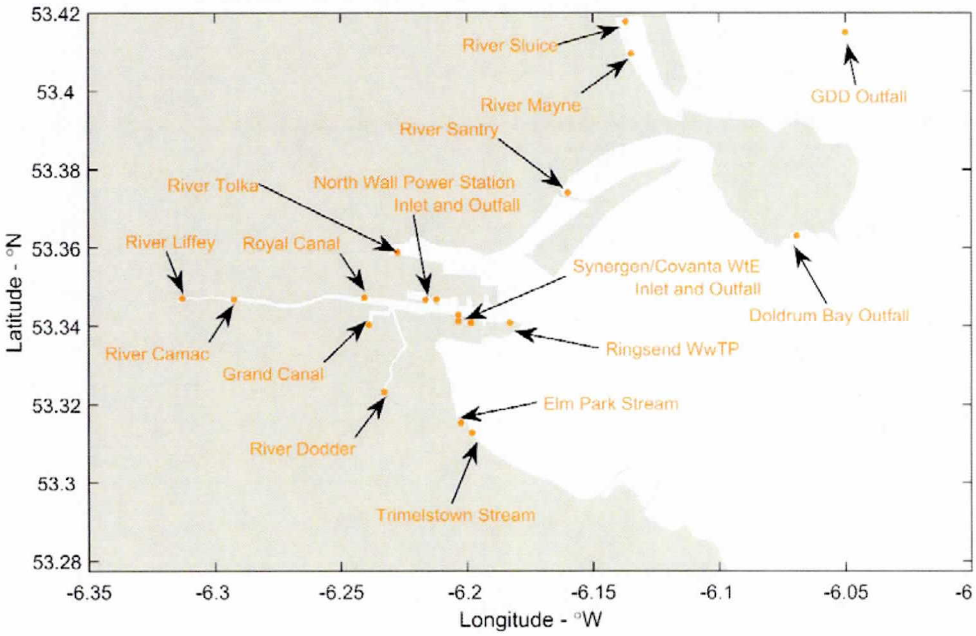


Figure 6.1 Map showing point sources specified in the hydrodynamic model.

6.3.1.1 Rivers, Streams, and Canals

There are eleven (11) freshwater sources in the hydrodynamic model.

- River Liffey;
- River Dodder and River Slang (combined);
- River Tolka;
- River Camac;
- River Santry;
- Royal Canal;
- Grand Canal;
- River Mayne;
- River Sluice;
- Elm Park Stream; and
- Trimelston Stream.

Average conditions

River flow rates were determined from a statistical analysis of gauged values (see section 4.1.1). To provide realistic estimates of discharges, the statistical analysis was based on a hydrometric record of up to 20 years. The analysis was performed for annual, summer (June – August), and winter (December – January). These returned values were used to represent the typical river flow rates during the model reference period 2013 – 2015.

For the River Liffey, flow data were available for the year 2015 only at the Leixlip Power Station. This gauge is owned and operated by ESB and is not a standard water level recording station as operated by the EPA or OPW. The River Liffey is a major river and the gauge at Leixlip was located some distance upstream of the location at which it entered the hydrodynamic model domain (at Islandbridge Weir). It was therefore decided to scale the gauged flow rate for the Liffey to account for additional run-off into the river between the gauging station and the receiving water.

$$Q_{Liffey,Scaled} = Q_{Liffey,Leixlip} \times \frac{A_{Liffey,Islandbridge}}{A_{Liffey,Leixlip}}$$

$A_{Liffey,Islandbridge}$ and $A_{Liffey,Leixlip}$ were the catchment area of the Liffey at Islandbridge Weir and Leixlip Power Station, respectively. These values were taken from the Eastern CFRAM Study Hydrology Report (Ref. /10/) which gave a scale factor of 1.132.

Figure 6.2 to Figure 6.8 show time series and statistics of the flow in the principal rivers during the period 2013 to 2015. The statistics are shown for annual, summer and winter conditions. As the rivers in the catchment were small and mean flow rates are often strongly influenced by episodic high flow events, it was considered that the median flow rate provided the best representation of the general conditions.

Table 6.3 gives the values for the typical annual, summer, and winter conditions that were set in the water quality modelling scenarios.

Note that, the Rye Water and River Grifeen are tributaries of the River Liffey that join between Leixlip Power Station and the start of the Upper Liffey Estuary at Islandbridge Weir (Figure 4.1). The specified flow rate for the Liffey in Table 6.3 represents the combined flow from these three water courses.

Other tributaries, such as the River Camac and Dodder, enter the Lower Liffey Estuary within the model domain. The value for the River Dodder also includes contributions from the River Slang. The Royal Canal and the Grand Canal also flow into the Lower Liffey. Note that the values for the canals have been estimated, as no gauged flow rates were available.

The Mayne River and Sluice River both discharge into the Baldoyle Estuary, north-east of Dublin City. These rivers were included due to their close proximity to the GDD outfall and were specified in both the baseline and future scenarios. Flow rates for the River Sluice and River Mayne were provided by the GDD team (Ref. /12/).

The Elm Park Stream and Trimleston Stream are minor urban watercourses in South Dublin. Though the streams are not large they receive urban runoff due to a surface water drainage. Both discharge into the south of Dublin Bay near designated bathing water beaches. The flow rates for these two streams are estimated values.

River temperatures were set according to median observed values from EPA monitoring sites during annual, summer, and winter conditions at the following locations (see Figure 4.22):

- DB010 – **Liffey** City, Heuston Station upstream of Cammock outfall;
- DB120 – **Dodder/Grand Canal** Basin; and
- DB310 – **Tolka** downstream of Annesley Bridge.

In all cases, the salinity of the river waters was set to 0 PSU (i.e. fresh water).

Storm conditions

For the summer storm scenario, river flow rates were based on the 15-minute gauged values during the event where available (Figure 6.9 to Figure 6.13).

For the Liffey (at Leixlip Power Station) and the Rye Water (at Leixlip) no observed data were available during the chosen storm event which occurred on the 2nd and 3rd of August 2014. Instead, the flow rate for the River Liffey was approximated by scaling the River Cammock using a flow by area method.

$$Q_{Liffey,Storm} = Q_{Cammock,storm} \times \frac{A_{Liffey,Islandbridge}}{A_{Cammock}}$$

$A_{Liffey,Islandbridge}$ and $A_{Cammock}$ were the catchment area of the Liffey up to Islandbridge Weir and the River Cammock, respectively. These values were taken from the Eastern CFRAM Study Hydrology Report (Ref. /10/) which gave a scale factor of approximately 14.

For the River Santry, the daily mean flow rates were used, as the 15-minute discharge values contained significant amounts of missing data.

Note that the values for the canals have been estimated, as no gauged flow rates were available.

Flow rates for the minor streams and canals were approximated and the River Sluice and River Mayne were set according to the values provided by the GDD team Ref. /12/).

Table 6.3 Flow rate, temperature, and salinity for all rivers in the water quality model for annual, summer, winter and storm conditions.

River	Median flow rate [m^3/s]				Temperature [°C]				Salinity [PSU]			
	Annual	Summer	Winter	Storm	Annual	Summer	Winter	Storm	Annual	Summer	Winter	Storm
Liffey	6.1	2.2	27.4	Figure 6.9	10.5	15	6	15	0	0	0	0
Dodder + Slang	1.5	0.9	2.6	Figure 6.11	10.5	14.5	7	14.5	0	0	0	0
Tolka	1.1	0.5	2.2	Figure 6.12	11	15	7.5	15	0	0	0	0
Camac	0.4	0.3	0.6	Figure 6.10	10.5	14.5	7	14.5	0	0	0	0
Santry	0.1	0.1	0.2	Figure 6.13	11	15	7.5	15	0	0	0	0
Royal Canal	0.1	0.1	0.1	0.1	10.5	14.5	7	14.5	0	0	0	0
Grand Canal	0.1	0.1	0.1	0.1	10.5	14.5	7	14.5	0	0	0	0
Mayne	0.2	0.2	0.2	0.2	11	15	7.5	15	0	0	0	0
Sluice	0.4	0.4	0.4	0.4	11	15	7.5	15	0	0	0	0
Elm Park Stream	0.05	0.05	0.05	0.5	10.5	14.5	7.5	14.5	0	0	0	0
Trimleston Stream	0.05	0.05	0.05	0.5	10.5	14.5	7.5	14.5	0	0	0	0

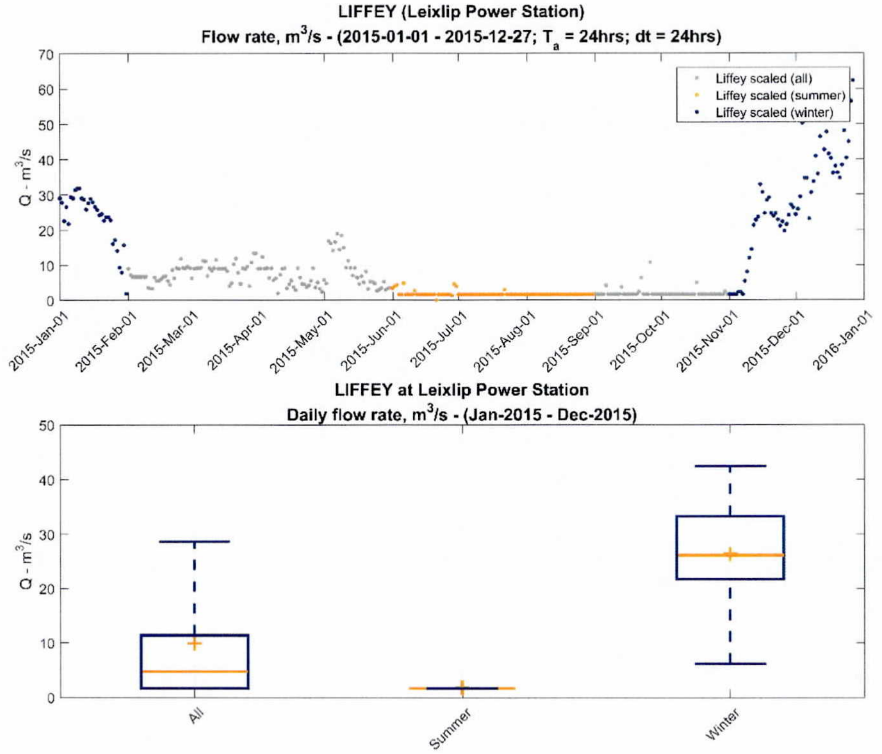


Figure 6.2 Flow rate in the River Liffey at Leixlip Power Station from 2013 – 2015. Time series of flow rate showing summer and winter periods (upper panel). Box plots showing the annual, summer and winter mean flow rates (orange cross), median flow rates (orange horizontal line), 25-75% quantile (blue box) and 10-90% quantile (whiskers).

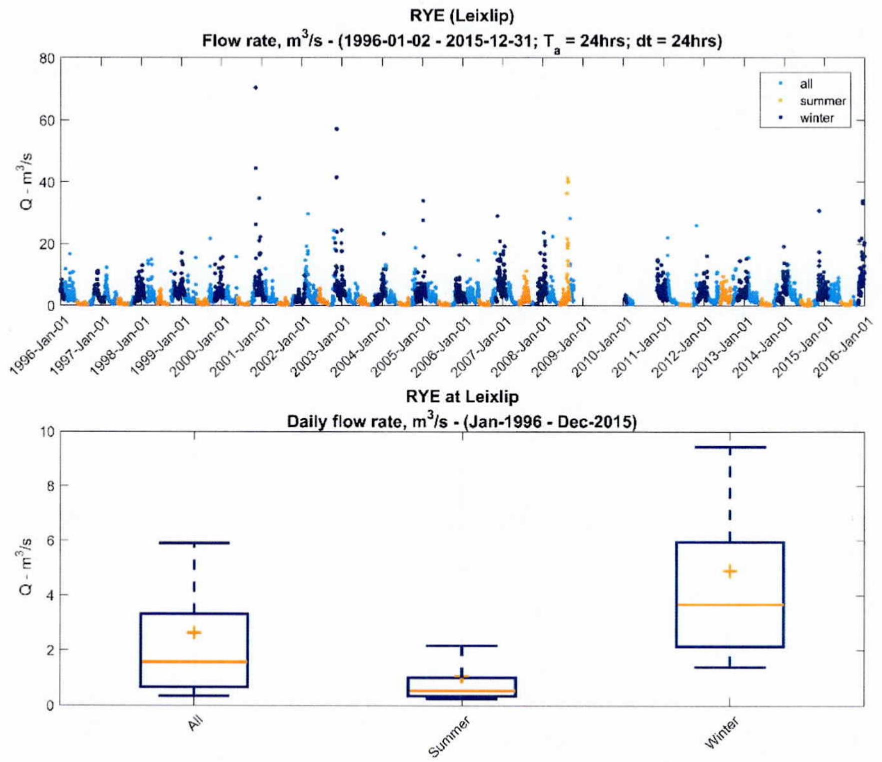


Figure 6.3 Flow rate in the River Rye at Leixlip from 2013 – 2015. Time series of flow rate showing summer and winter periods (upper panel). Box plots showing the annual, summer and winter mean flow rates (orange cross), median flow rates (orange horizontal line), 25-75% quantile (blue box) and 10-90% quantile (whiskers).

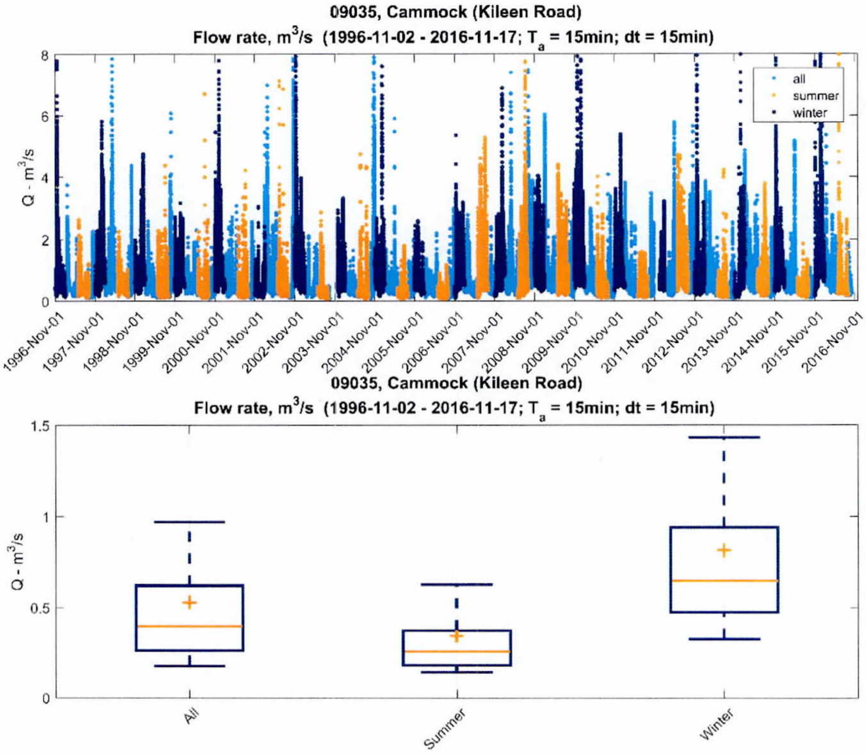


Figure 6.4 Flow rate in the River Camac at Killeen Road from 1996 – 2016. Time series of flow rate showing summer and winter periods (upper panel). Box plots showing the annual, summer and winter mean flow rates (orange cross), median flow rates (orange horizontal line), 25-75% quantile (blue box) and 10-90% quantile (whiskers).

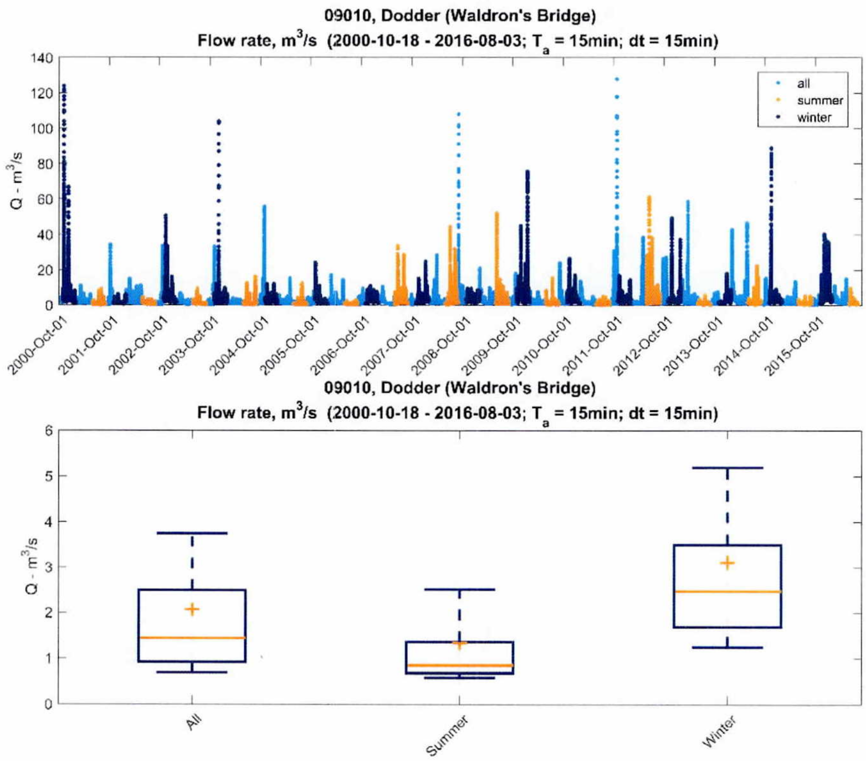


Figure 6.5 Flow rate in the River Dodder at Waldron's Bridge from 2000 – 2016. Time series of flow rate showing summer and winter periods (upper panel). Box plots showing the annual, summer and winter mean flow rates (orange cross), median flow rates (orange horizontal line), 25-75% quantile (blue box) and 10-90% quantile (whiskers).

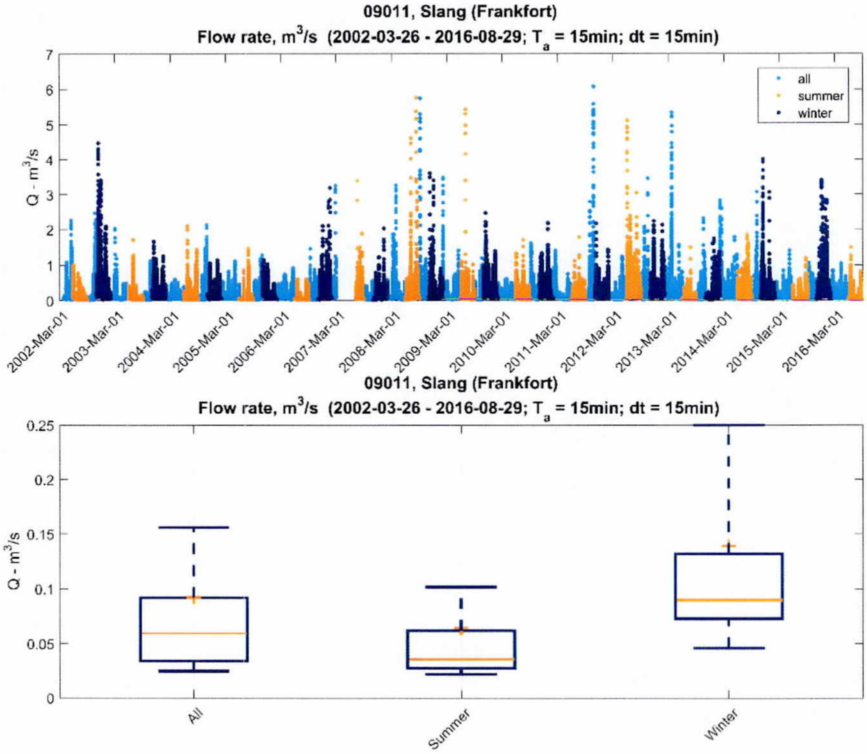


Figure 6.6 Flow rate in the River Slang at Frankfort from 2002 – 2016. Time series of flow rate showing summer and winter periods (upper panel). Box plots showing the annual, summer and winter mean flow rates (orange cross), median flow rates (orange horizontal line), 25-75% quantile (blue box) and 10-90% quantile (whiskers).

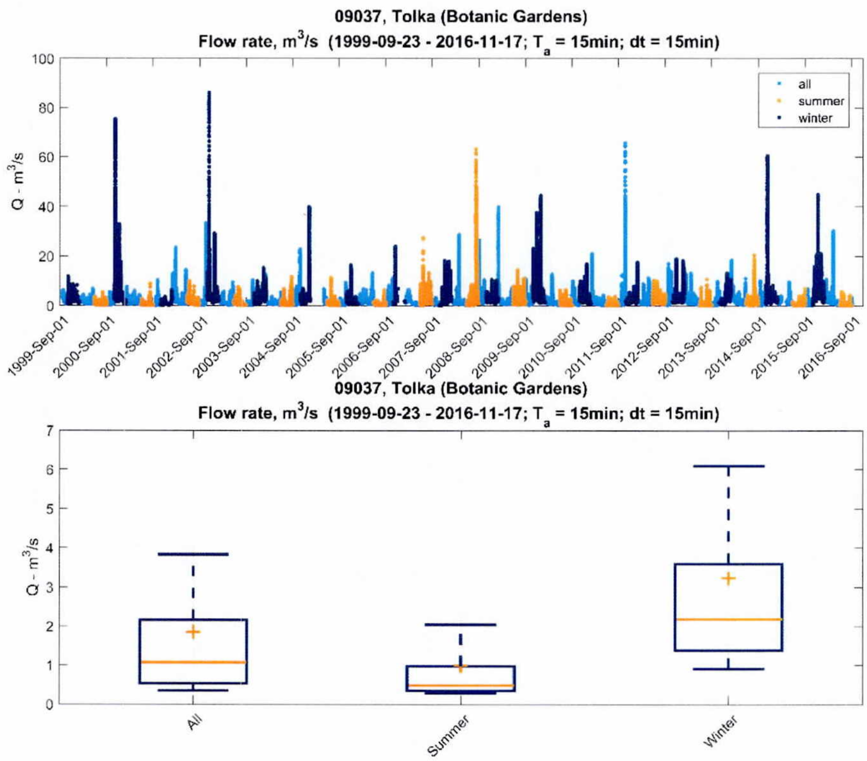


Figure 6.7 Flow rate in the River Tolka at Botanic Gardens from 1999 – 2015. Time series of flow rate showing summer and winter periods (upper panel). Box plots showing the annual, summer and winter mean flow rates (orange cross), median flow rates (orange horizontal line), 25-75% quantile (blue box) and 10-90% quantile (whiskers).

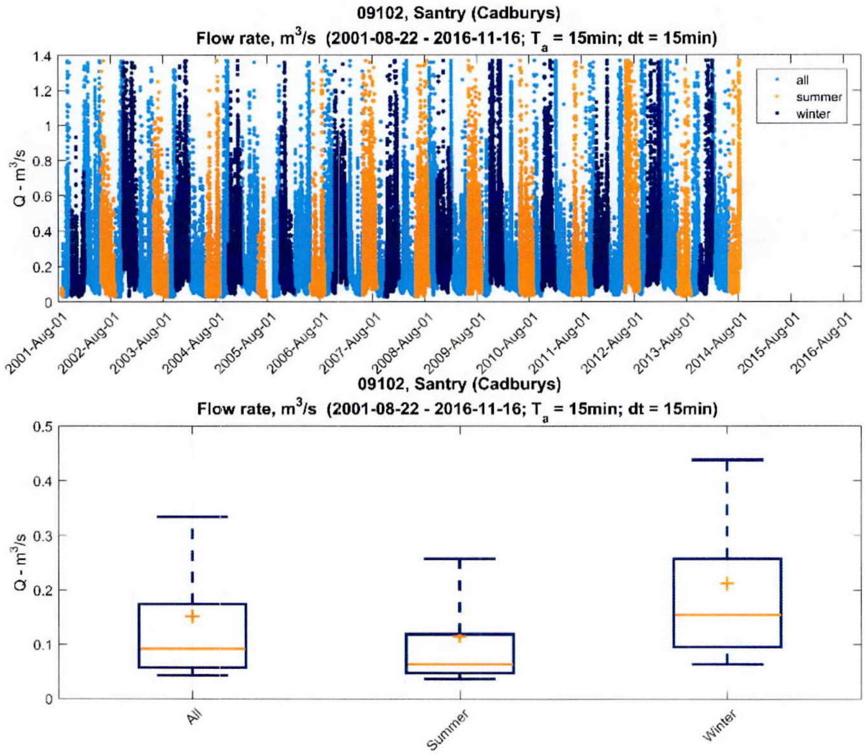


Figure 6.8 Flow rate in the Santry at Cadburys from 2001 – 2015. Time series of flow rate showing summer and winter periods (upper panel). Box plots showing the annual, summer and winter mean flow rates (orange cross), median flow rates (orange horizontal line), 25-75% quantile (blue box) and 10-90% quantile (whiskers).

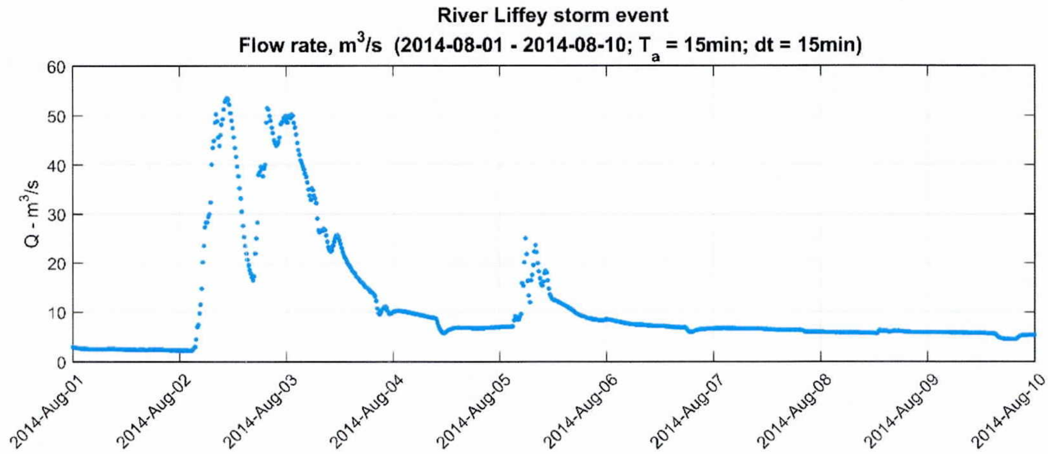


Figure 6.9 River Liffey flow rate before, during, and after storm scenario (2nd-3rd August 2014).

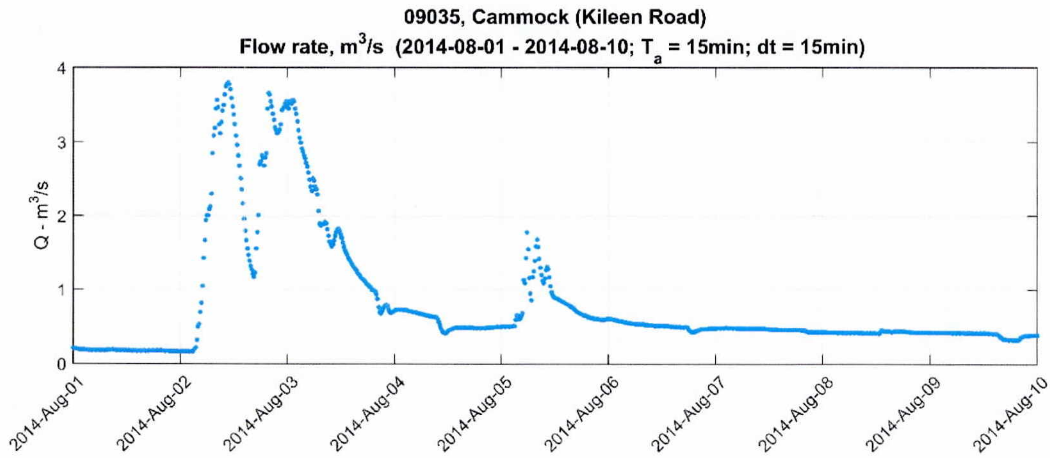


Figure 6.10 River Camac flow rate before, during, and after storm scenario (2nd-3rd August 2014).

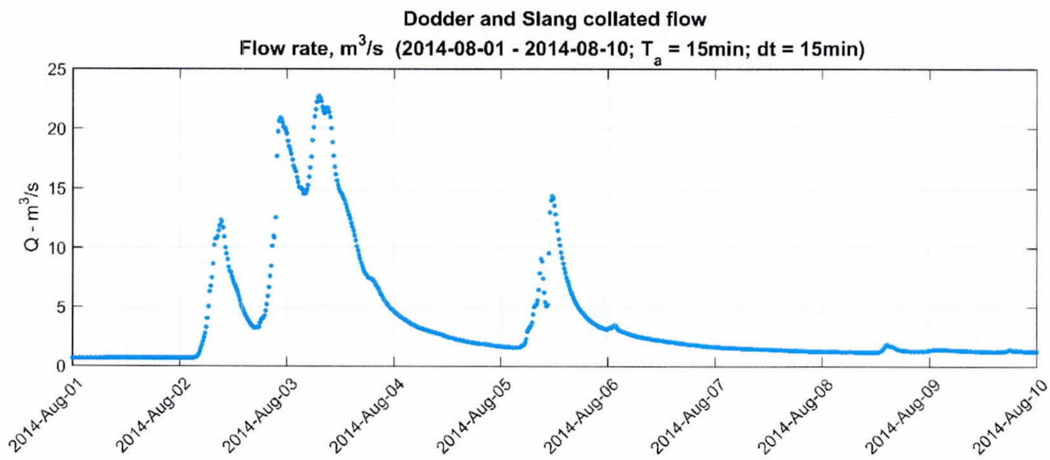


Figure 6.11 Combined River Dodder and River Slang flow rate before, during, and after storm scenario (2nd-3rd August 2014).

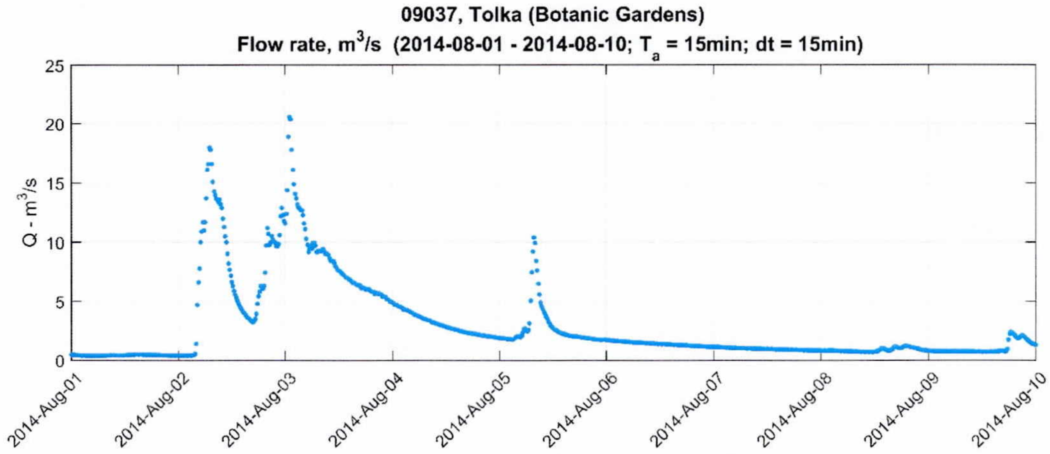


Figure 6.12 River Tolka flow rate before, during, and after storm scenario (2nd-3rd August 2014).

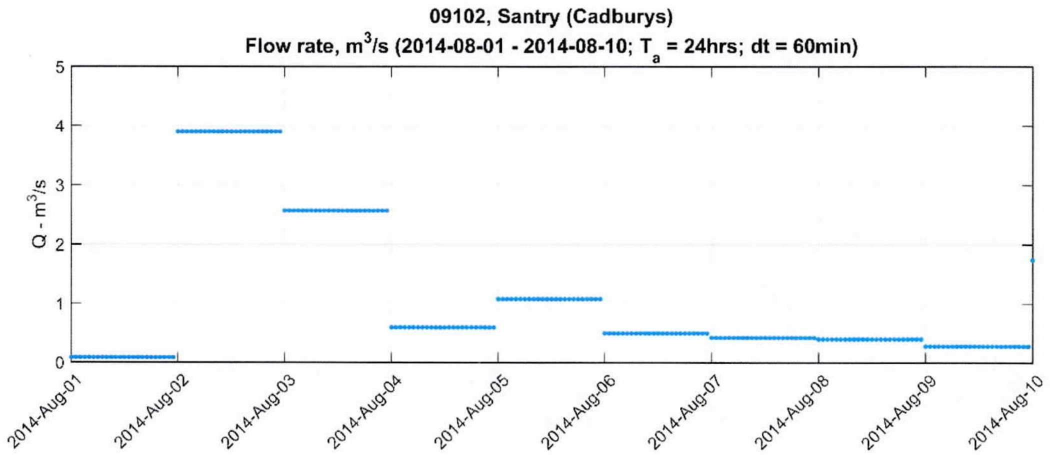


Figure 6.13 River Santry flow rate before, during, and after storm scenario (2nd-3rd August 2014).

6.3.1.2 Ringsend WwTP Discharge

There are two-point sources for the Ringsend WwTP:

- SW1, Primary Wastewater Discharge on the Lower Liffey and within the ESB Poolbeg Cooling Water Channel.
- SW2, Storm Water Overflow Discharge, located approximately 500m upstream of SW1 on the Lower Liffey Estuary.

Table 6.4 gives the flow rates for the both Ringsend SW1 and SW2 for the existing environment (hydrodynamic model scenarios 1 – 5) and the future discharge environment (hydrodynamic model scenarios 6 – 17).

The outfall at SW2 is only active when the WwTP storage tank capacity is exceeded. Figure 6.14 shows the measured effluent discharge rate at SW1 and SW2 during the period around the summer storm of the (2^{nd} – 3^{rd} August 2014).

Figure 6.15 shows the predicted effluent discharge rate at SW1 and SW2 for the future scenario. Once more, the outfall at SW2 is only active when the WwTP storage tank capacity is exceeded. However, as the volume of water discharged from the primary outfall at SW1 will increase in the future scenario, the total volume of water discharged at SW2 during the storm is lower than the future scenario.

Figure 6.16 shows a time-series of observed effluent temperature at SW1 during the period around the storm event of 2 – 3 August 2014. These data were used to describe the temperature at both SW1 (primary wastewater discharge) and SW2 (storm water overflow) during the event and for both the existing and future discharge scenarios.

Table 6.4 Flow rates at Ringsend WwTP outfalls SW1 and SW2 for baseline and future water quality model scenarios (Ref. /13/).

River	Median flow rate [m^3/s]					Temperature [°C]					Salinity [PSU]				
	Annual	Peak	Summer	Winter	Storm	Annual	Peak	Summer	Winter	Storm	Annual	Peak	Summer	Winter	Storm
Ringsend SW1 (existing environment)	4.91	8.04	4.28	5.76	Figure 6.14	16.2	16.2	19.8	13.6	Figure 6.16	0	0	0	0	0
Ringsend SW2 (existing environment)	0	0	0	0		0	0	0	0		0	0	0	0	0
Ringsend SW1 (future discharge)	6.95	11.1	6.05	8.15	Figure 6.15	16.2	16.2	19.8	13.6		0	0	0	0	0
Ringsend SW2 (future discharge)	0	0	0	0		0	0	0	0		0	0	0	0	0

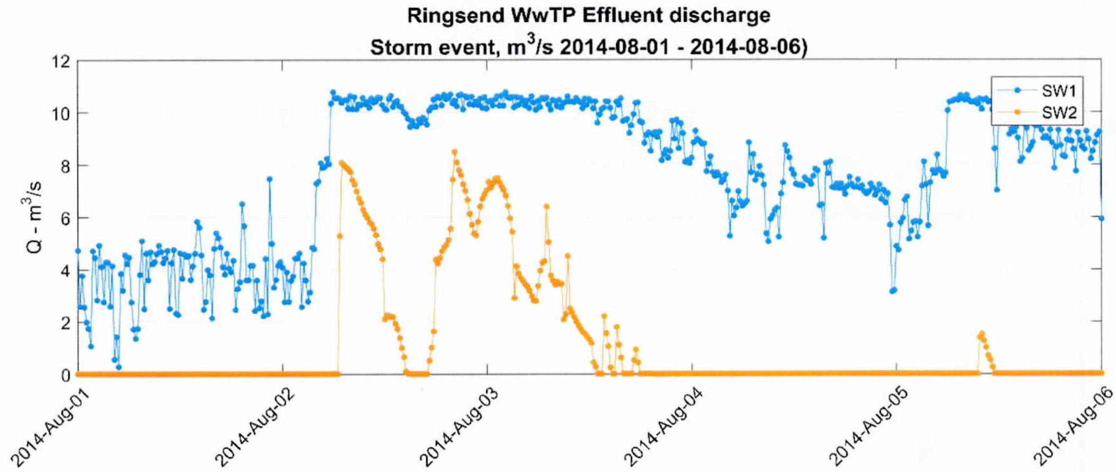


Figure 6.14 Flow rate at Ringsend WwTP outfalls SW1 (blue) and SW2 (orange) before, during, and after the summer storm scenario (2-3 August 2014) for baseline scenario.

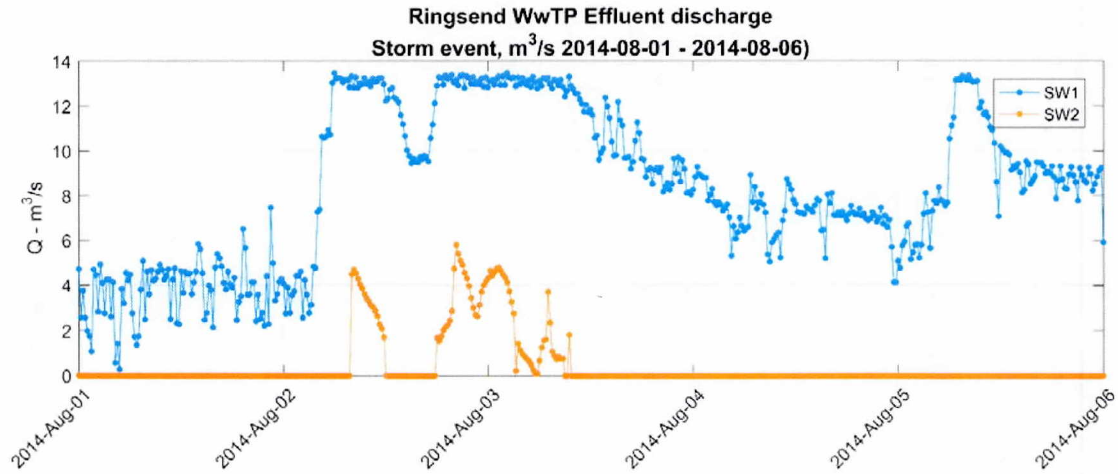


Figure 6.15 Flow rate at Ringsend WwTP outfalls SW1 (blue) and SW2 (orange) before, during and after the summer storm scenario (2-3 August 2014) for future scenario.

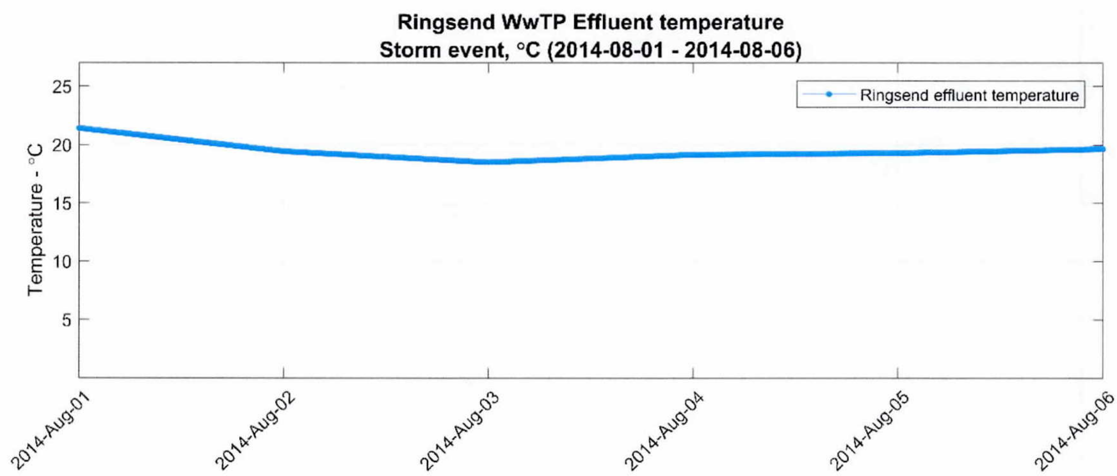


Figure 6.16 Temperature of Ringsend effluent during summer storm scenario (2-3 August 2014) for both existing and future discharge scenario at SW1 (primary discharge) and SW2 (storm water overflow).

6.3.1.3 Dublin Storm Water Overflows

During heavy rainfall events the flows may exceed the sewage treatment plant capacity. In this event, relief structures allow the combined Storm Water Overflow (SWO) to be discharged directly into the Lower Liffey Estuary. This scenario was included in the summer storm scenario at the request of Irish Water.

During the summer storm conditions (hydrodynamic model scenario 5 and 10), the contribution from Dublin SWO's were specified in the hydrodynamic model. The loads were provided by Irish Water for three (3) locations representing the Liffey North Bank, Liffey South Bank, and the River Dodder (Figure 6.17). These loads were calculated by Irish Water running the City Centre & Rathmines/Pembroke combined network model in InfoWorks CS for the August storm event and collating spill volumes from all SWOs discharging to the Liffey Estuary and River Dodder. Spill Volumes for the Dodder were applied as a point discharge at the model boundary while the loads for the River Liffey North Bank and River Liffey South Bank were equally split between 4 outfall locations (Figure 6.18). This approach was agreed with Irish Water on the basis that these locations are reflective of development within the catchment.

The SWO loads were specified as surface point sources with zero excess temperature and a specified salinity of 0 PSU.

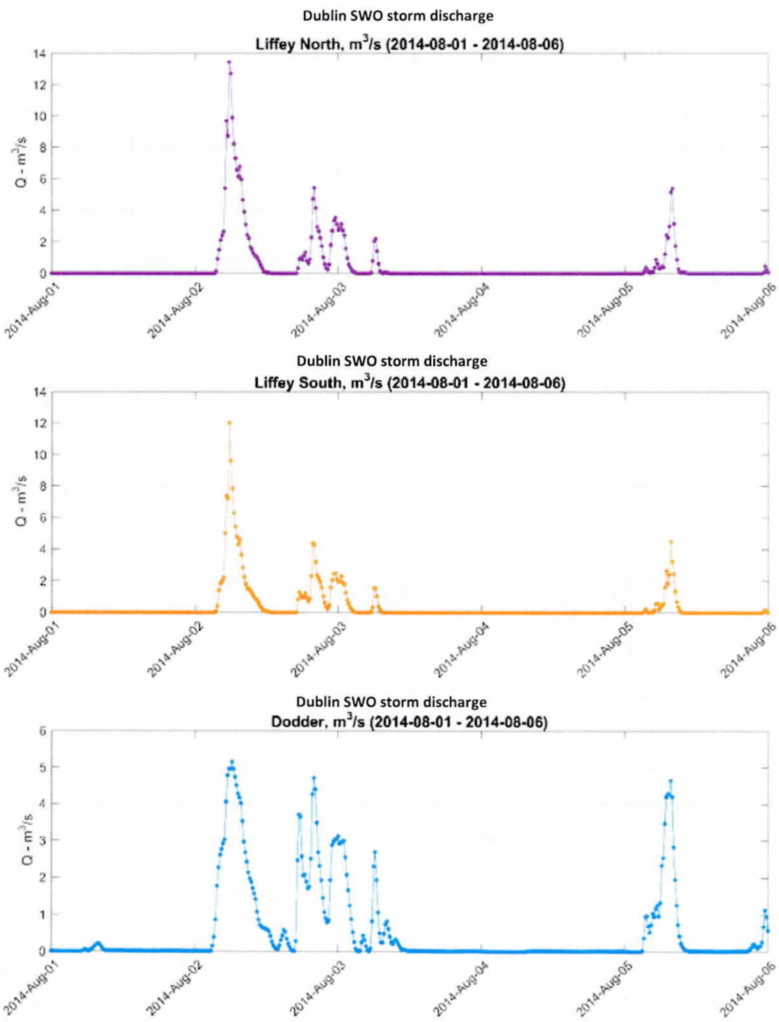


Figure 6.17 Dublin Storm Water Overflow (SWO) before, during, and after the summer storm scenario (2^{nd} – 3^{rd} August 2014).

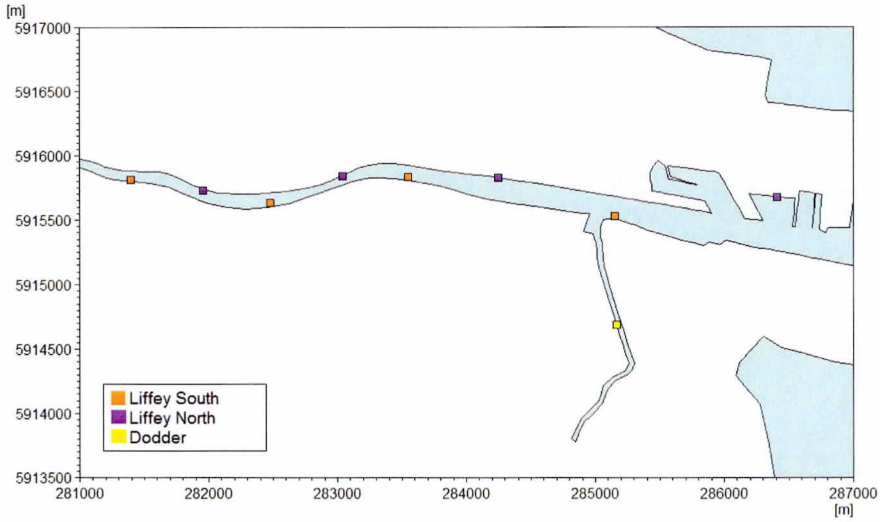


Figure 6.18 Location of the SWO's for the summer storm scenarios.

6.3.1.4 Other Wastewater and Industrial Outfalls

There are a number of additional sewage and industrial outfalls in and around the Lower Liffey Estuary and the Greater Dublin coastal area. The status of these outfalls was classed as being either:

- Operational in the existing environment scenario only;
- Operational in the future discharge environment scenario only;
- Operational in both the existing environment and the future discharge environment; and
- Intermittently operational in the future discharge scenario.

The values specified for the outfall discharges in the hydrodynamic model are shown in Table 6.5. The operation/non-operation of these outfalls and their locations can be identified from Table 6.1 and Figure 6.1, respectively. A summary of each of these outfalls is provided below.

Doldrum Bay Outfall

Doldrum Bay is a beach on the south side of Howth Head in the north of Dublin Bay. A raw sewage outfall is known to discharge into the bay at this location. It is understood that the untreated effluent is of domestic origin from approximately 40 homes. The discharge at Doldrum Bay was estimated based on the assumption of a wastewater personal load of 0.2 m^3/day (Ref. /14/) and a population of approximately 120.

The Doldrum Bay outfall was operational in the existing environment scenario only, as it was assumed that the raw sewage discharge will be removed in the near future.

Poolbeg Power Station

Poolbeg Generation Station is a power station located on the Poolbeg Peninsula at Ringsend, on the south bank of the Lower Liffey Estuary. There have been a number of power stations on the site since the early twentieth century. The modern-day plant consists of 480 MW combined-cycle gas turbine (CCGT) operated by the Electricity Supply Board of Ireland (ESB).

The cooling water discharge from the plant enters the Lower Liffey Estuary via a channel and weir. This is the same structure as used by the Ringsend WwTP outfall (see section 5.4.7).

It is understood Poolbeg Plant is currently reserved as back-up and only fired during peak system demand or unusual load demands (e.g. due to non-availability of other electrify generation sources). As such, the Poolbeg Power Station outfall was classed as being intermittently operational in the future discharge scenarios. The flow rate and temperature of the cooling water discharge were provided by JB Barry.

Synergen Power Station

The Synergen Power Station is a combined cycle gas generating plant located on the south side of the River Liffey. The plant extracts cooling water from the Lower Liffey and discharges this water via a channel back into the estuary approximately 1 kilometre upstream of the Ringsend WwTP.

The Synergen Power Station was included in both the existing environment and future discharge scenarios.

Data on the emissions to water were obtained from the annual environmental reports for the years 2013 – 2015 (Ref. /15/). During this period the average flow rate was approximately 6.1 m^3/s and the average rise in temperature of the cooling water above ambient water temperature was 6.5 °C. For the maintenance of continuity, there was a withdrawal of the same volume of ambient water from the Lower Liffey at the Synergen intake.

Covanta Waste-to-Energy Plant

The Dublin Waste-to-Energy (WtE) Project will see the construction and operation of a thermal treatment plant for the incineration of municipal waste. The plant will extract cooling water from the Lower Liffey and discharge this water via a channel back into the estuary approximately 1 kilometre upstream of the Ringsend WwTP. The plant was operational in future discharge scenarios only.

The discharge rate of cooling water from the Covanta WtE Plant was specified as 3.9 m^3/s, with an increase in ambient water temperature of 9.0 °C (Ref. /2/). For the maintenance of continuity, there was also a withdrawal of the same volume of ambient water from the Lower Liffey at the Covanta WtE Plant intake.

Greater Dublin Drainage (GDD) outfall

The Greater Dublin Drainage Project (GDD) involves the development of a new regional wastewater treatment facility for the greater Dublin area. The GDD project will consist of the construction of a new wastewater treatment plant in the north of Dublin at Clonshaugh, with an outfall pipeline discharging into the Irish Sea around 3 kilometres to the north of Howth Head.

The GDD outfall was included in the future discharge scenarios only. The flow rate and temperature for the outfall were provided courtesy of the GDD project team (Ref. /12/).

Shanganagh Outfall

The Shanganagh wastewater treatment plant is located in County Dublin serving a suburban catchment to the south of Dublin City. The primary discharge consists of a 1.7 kilometre long sea that discharges into the Irish Sea outfall to the south of Dublin Bay.

The Shanganagh outfall was operational in both the existing environment scenario and the future discharge scenario. The flow rate and temperature were provided courtesy of the GDD project team (Ref. /12/).

Table 6.5 Flow rate, temperature, and salinity for outfalls in the water quality model for annual, summer, winter, and storm conditions.

River	Median flow rate [m^3/s]				Temperature relative to ambient levels [°C]				Salinity [PSU]			
	Annual	Summer	Winter	Storm	Annual	Summer	Winter	Storm	Annual	Summer	Winter	Storm
Shanganagh WwTP Outfall	2.1	2.1	2.1	2.1	6.0	6.0	6.0	6.0	0	0	0	0
SynerGen Power Station	6.1	6.1	6.1	6.1	6.5	6.5	6.5	6.5	Ambient			
Covanta WtE Plant	3.9	3.9	3.9	3.9	9.0	9.0	9.0	9.0	Ambient			
GDD Outfall	1.8	1..8	1.8	1.8	0	0	0	0	0	0	0	0
Poolbeg Power Station	10.0	10.0	10.0	10.0	7.6	7.6	7.6	7.6	Ambient			
Doldrum Bay Outfall	2.8×10^{-4}	2.8×10^{-4}	2.8×10^{-4}	2.8×10^{-4}	0	0	0	0	0	0	0	0

6.3.2 Infrastructure Changes

Future changes to the existing port infrastructure in and around Dublin have the potential to alter the existing hydrodynamic regime and, therefore, the dispersal and fate of dissolved or suspended substances.

The hydrodynamic model was modified to simulate the effects of two (2) envisaged infrastructure changes in order to assess the sensitivity of these developments on flow and dispersion. This included repair of the ESB cooling water channel (hydrodynamic model scenario 14) and the Alexandra Basin Redevelopment Scheme (hydrodynamic model scenario 15). Further information on these runs are described below.

ESB Cooling Water Channel

The primary Ringsend WwTP outfall discharges treated effluent into the ESB Poolbeg Power Station cooling water channel and flows into the Lower Liffey Estuary via a weir (Figure 5.17). As described in section 5.4.6, there is extensive damage to the existing cooling water channel. This damage means that treated effluent enters the Lower Liffey through gaps and holes in the walls of the cooling water channel.

In hydrodynamic model scenario 14, the damaged sections of the ESB cooling water channel was assumed to have been repaired. This was achieved in the model setup by setting the crest levels of the damaged sections above the maximum water level, thus not enabling any flow to enter the Lower Liffey via the cooling water channel (Table 6.6 and Figure 6.19). The Ringsend effluent discharged into the cooling water channel can only enter the Liffey through the weir (section D), which faces downstream of the WwTP and towards Dublin Bay.

The model run considered average annual conditions for the future discharge scenarios only. No other changes to the model setup were specified.

In all other hydrodynamic model scenarios (existing and future discharge environments) the cooling water channel and weir were modelled in the existing damaged state (Table 6.6 and Figure 6.19).

Table 6.6 Crest levels of Ringsend Weir sections specified in the hydrodynamic model in the existing (damaged) and repaired state.

Weir Section	Weir Crest Level in existing damaged state [mMSL]	Weir Crest Level in repaired state [mMSL]
Section A	1	> maximum water level
Section B	-1	> maximum water level
Section C	-1	> maximum water level
Section D	0	0

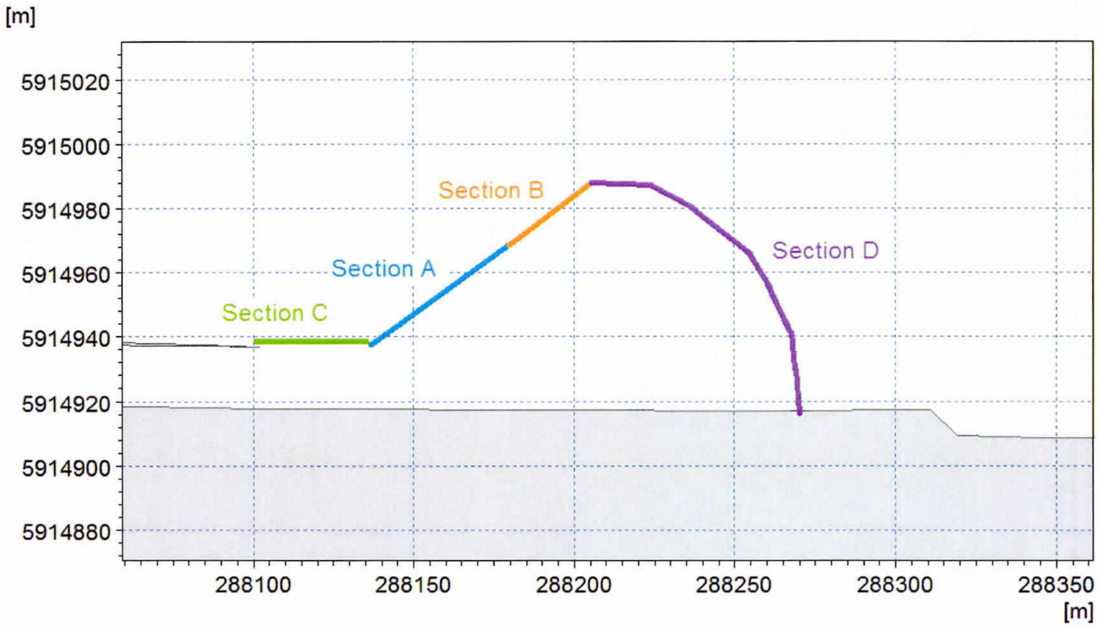


Figure 6.19 Weir sections as specified in the hydrodynamic model.

Alexandra Basin Redevelopment Project

As part of Dublin Port Company masterplan for 2040, several major infrastructure developments within the Port and entrance channel have been proposed. Amongst these developments is a capital dredging scheme to deepen the fairway and approach to Dublin Port, to increase the ruling depth from -7.8 m to -10.0 m below chart datum.

A previous modelling study performed for the EIS of the Alexandra Basin Redevelopment Project (Ref. /7/) concluded that:

- There will be no significant changes to the tidal flow regime of Dublin Bay.
- There will be no perceptible change in tidal velocity within the deepened, realigned navigation channel.

Nevertheless, the impact of the capital dredging has on flow and dispersion was simulated. This was achieved in hydrodynamic model scenario 15 by reducing the model bathymetry to -10.0 m below chart datum along the approach channel to Dublin Port and within the Alexandra Basin.

Figure 6.20 shows the model bathymetry including the Alexandra Basin Redevelopment Scheme and the change in bathymetry relative to the existing model setup.

The model run considered average annual conditions for the future discharge scenarios only. No other changes to the model setup were specified.

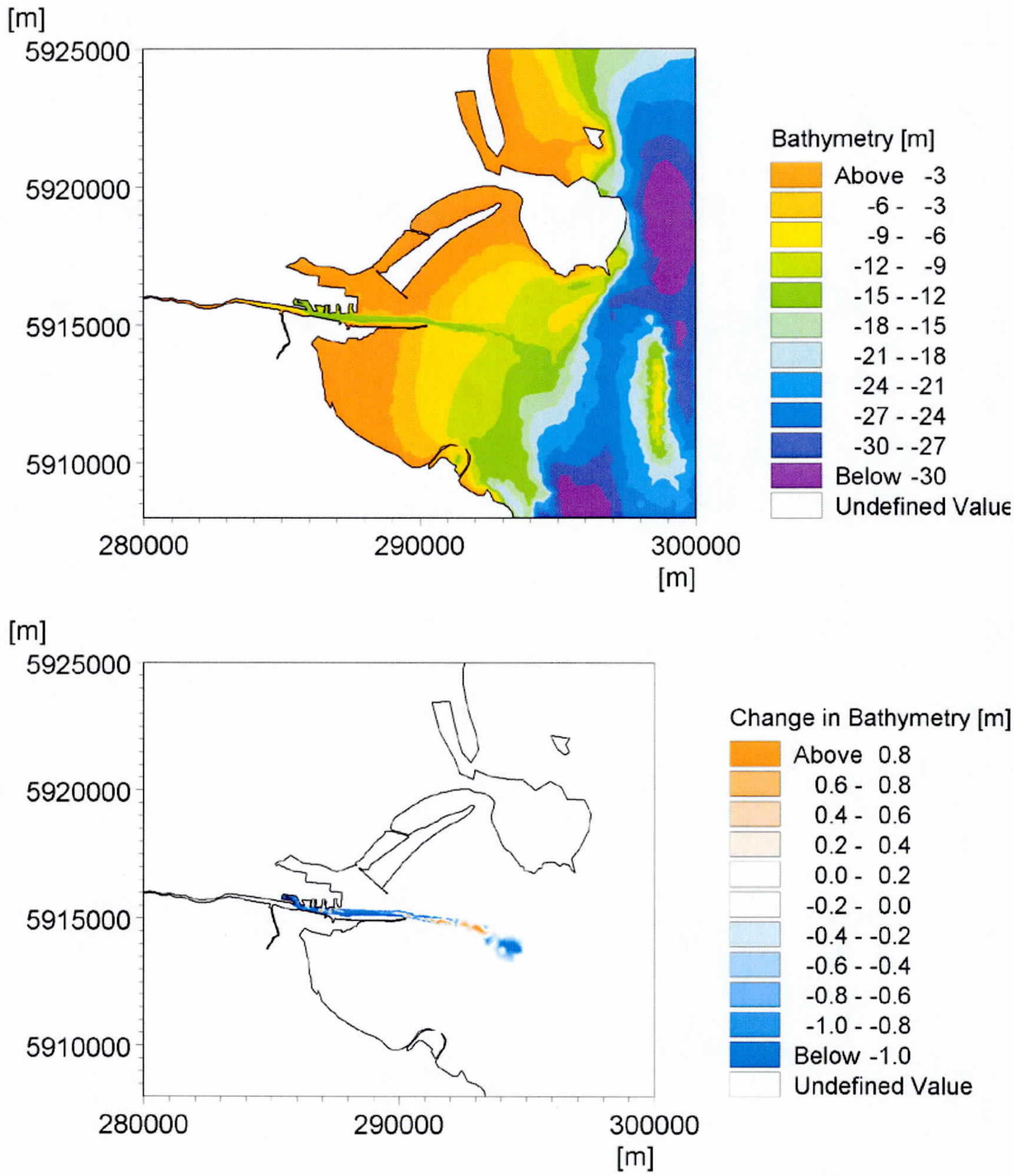


Figure 6.20 Upper panel: Hydrodynamic model bathymetry with Alexandra Basin Redevelopment Scheme included. Lower panel: difference in bathymetry between Alexandra Basin Redevelopment Scheme and existing situation (blue areas show deeper water due to dredged approach channel).

6.3.3 Meteorological Conditions

The temperature of the water in the hydrodynamic model interacts with the atmosphere through heat exchange. The atmospheric conditions were determined using data from a 5-year meteorological model (2010 – 2015).

Average conditions

The diurnal variation in air temperature and relative humidity was calculated by finding the median value at each hour of the day. This was performed for average annual, summer, and winter conditions. Figure 6.21 shows the resulting data which was specified for each day of the average condition scenarios.

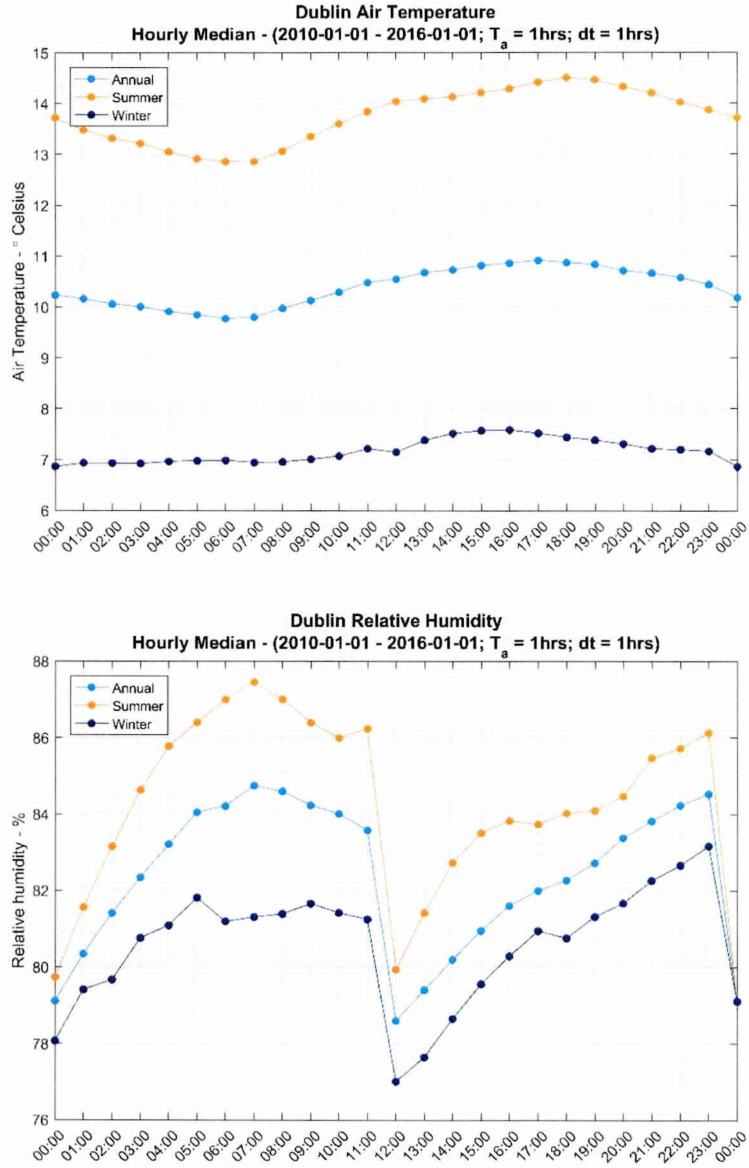


Figure 6.21 Diurnal variation in air temperature (top panel) and relative humidity (lower panel) for Dublin during average annual, summer, and winter conditions.

Storm conditions

Figure 6.22 shows the air temperature and relative humidity in Dublin during the summer storm scenario.

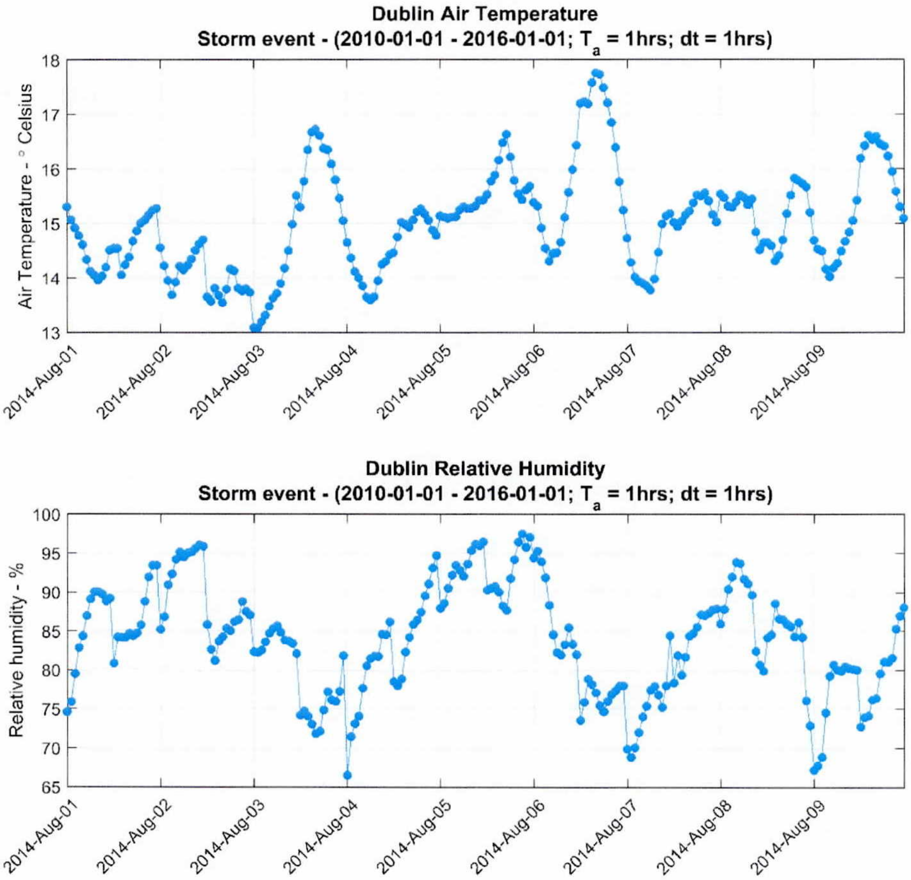


Figure 6.22 Variation in air temperature (top panel) and relative humidity (lower panel) for Dublin before, during, and after the storm scenario (2nd – 3rd August 2014).

6.3.4 Boundary Conditions

Tidal forcing was applied along the offshore open boundaries of the hydrodynamic model. The offshore boundary data were extracted from a regional model of the Irish Sea developed and maintained by DHI (Figure 5.7). The regional tidal model was in turn driven by surface elevations from a global tidal model.

The tidal data were specified as varying (spatially and temporally) along each of the open boundaries, thereby enabling the variation in water surface elevation and current speed to be captured by the model.

Table 6.7 summarises the open-boundary conditions specified for the hydrodynamic model.

Table 6.7 Offshore boundary conditions for hydrodynamic model (summer and winter).

Boundary	Temperature [°C]			Salinity [PSU]			Water Levels [m]	Current Speed [m/s]
	Annual	Summer	Winter	Annual	Summer	Winter		
Offshore Boundary	10.5	14	7	34	34	34	Time varying covering a full spring neap tidal cycle.	

6.4 Transport Model

The transport model simulates the spreading and fate of dissolved or suspended substances under the influence of the fluid transport and associated dispersion processes. The transport model was used to setup the water quality model scenarios for the Ringsend WwTP Upgrade EIAR.

A set of ninety-four (94) water quality scenarios were simulated as summarised in Table 6.2. These scenarios represented both existing environment over the baseline period (2013 – 2015, inclusive) and various permutations of the future discharge environment. The integer part of the model run number represents the hydrodynamic model scenario used as the basis for the water quality model scenario (e.g. run no. 1.05 is associated with hydrodynamic model scenario 1, and run no. 6.17 is associated with hydrodynamic model scenario 6).

The setup of the water quality model scenarios is described in this section.

6.4.1 Components

The water quality models were used to simulate six (6) different components (or pollutants), including:

- Faecal coliforms (Escherichia coli, E. coli);
- Dissolved Inorganic Nitrogen (DIN);
- Ammonia;
- Molybdate Reactive Phosphorus (MRP);
- Biochemical Oxygen Demand (BOD); and
- Total suspended solids (TSS).

For some cases, particle tracking was used instead of pollutant loads in order to investigate the transport of non-decaying substances.

6.4.2 Dispersion

Dispersion describes the transport due to non-resolved processes in the 3D hydrodynamic model. Horizontal dispersion is used to include the effects of non-resolved eddies and vertical dispersion is typically related to bed generated turbulence.

The effects of horizontal and vertical dispersion were included in the transport model using a scaled eddy viscosity formula. In this case, the dispersion coefficient was calculated as the eddy viscosity multiplied by a scaling factor. The scaling factor was set to a value of 1 (the default value) for both horizontal and vertical dispersion.

6.4.3 Decay

To simulate the time evolution of the various pollutants a decay rate was introduced. The decay rate was used to approximate the complex interactions between each pollutant and the environment within the estuary.

The decay coefficients were established based on DHI's experience of water quality modelling and previous experience in the Dublin Bay area. It is important to note that the use of an empirical constant coefficient, parameterises the processes taking place and does not specifically consider the dynamic interactions of a full ecological model.

In the model the linear decay of a component is described by:

$$dC/_{dt} = -kc \quad (1)$$

dC/dt s the decay rate (i.e. the change in concentration over time)

c is the specific concentration

k is the decay constant [s^{-1}]

Table 6.13 summarises the decay constants that were specified in the water quality model.

Note that not all substances were simulated during all conditions (annual average, summer, winter or storm).

The same decay rates were used in both the existing and future discharge scenarios.

Table 6.8 Decay constants for water quality modelling conditions.

Pollutant	Decay Rate [s^{-1}]			
	Average	Summer	Winter	Storm
BOD	1.16×10^{-6}			1.16×10^{-6}
TSS	0			
Ammonia	2.31×10^{-6}			
DIN	6.75×10^{-7}	1.16×10^{-6}	1.93×10^{-7}	
MRP	4.05×10^{-7}	8.10×10^{-7}	1.35×10^{-7}	
E. coli		1.20×10^{-4}	1.47×10^{-5}	1.20×10^{-4}

6.4.4 Source Concentrations

In the transport model, a source concentration (pollutant load) can be specified for each point source.

Figure 6.1 shows the location of all point sources in the hydrodynamic model scenarios, which include rivers, streams, canals, and inlets, as well as wastewater and industrial outfalls, in and around Dublin Bay. As stated in section 6.3.1, not all point sources were included in every scenario. Table 6.1 summarises which sources were included in each of the seventeen hydrodynamic model scenarios.

The source flux was calculated by the model as the product of the source discharge (flow rate from the hydrodynamic model) and the specified source concentration. This flux enters into the model domain, such that the inflowing mass of the pollutant is initially distributed over the element where the source is located. As a result, the concentration at the source location was often lower than the source concentration. For low source concentration and/or low source flow rates, the pollutant may be rapidly diluted.

6.4.4.1 Ringsend WwTP

There are two-point sources for the Ringsend WwTP:

- SW1, Primary Wastewater Discharge on the Lower Liffey and within the ESB Poolbeg Cooling Water Channel.
- SW2, Storm Water Overflow Discharge, located approximately 500m upstream of SW1 on the Lower Liffey Estuary.

A source concentration from SW1 was specified in each of the water quality scenarios.

A source concentration from SW2 was only active during the summer storm scenarios.

The concentrations of pollutants at SW1 and SW2 are given in Table 6.2. These concentrations were provided by J.B. Barry/Irish Water in a Microsoft Excel document (dated 27th October 2017).

Unless otherwise stated as being "Time-varying" in Table 6.2, the concentrations were set as invariant values over the simulation period.

For the summer storm conditions and the existing environment scenario, E. coli concentration were set according to measured values. These data were taken from an analysis spreadsheet Ringsend wastewater treatment works operations and maintenance report for August 2014 (Ref. /16/).

Figure 6.23 and Figure 6.24 show the E. coli concentrations as set for the primary wastewater discharge (SW1) and storm overflow discharge (SW2). Daily measured pollutant concentrations at the primary wastewater discharge (SW1) were available for week days (Monday to Friday). On days with no available data, a nearest neighbour interpolation scheme was used to infer the pollutant load. Daily measured pollutant concentrations at the storm water overflow discharge (SW2) were available for the 2nd August and 4th of August. The concentrations at SW2 during the overflow events were set according to highest value during the storm.

For the future discharge environment, the pollutant loads at the primary wastewater discharge (SW1) were set as invariant values in accordance with. For the storm water overflow discharge (SW2), the pollutant loads were the same as the baseline scenario (Figure 6.25). However, it should be noted that the occurrence of storm water overflow was reduced in the future scenarios due to the increased capacity of the upgraded WwTP (see section 6.3.1.2).

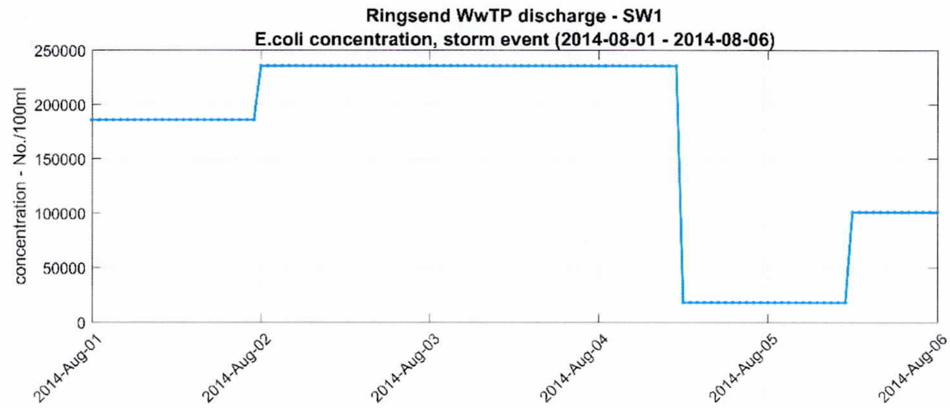


Figure 6.23 Time-series of concentration of E. coli from Ringsend WwTP outfall SW1 before, during and after the summer storm event (2nd – 3rd August 2014) for existing environment scenario.

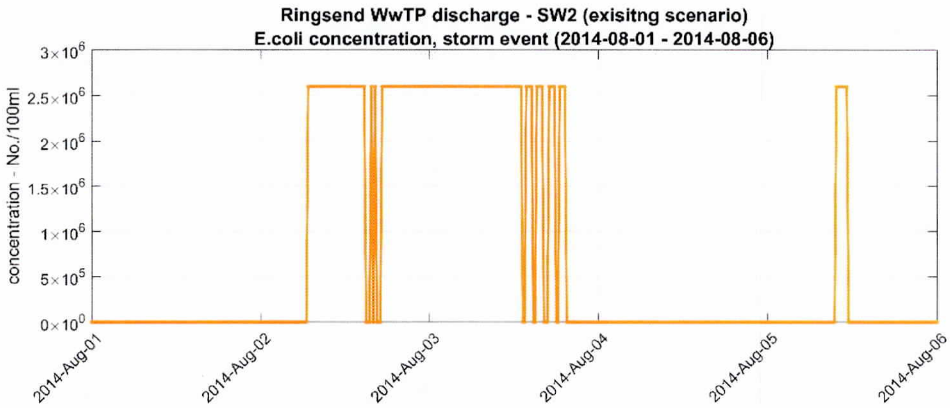


Figure 6.24 Time-series of concentration of E. coli from Ringsend WwTP storm water outfall SW2, before, during and after the summer storm event (2nd – 3rd August 2014) for existing environment scenario.

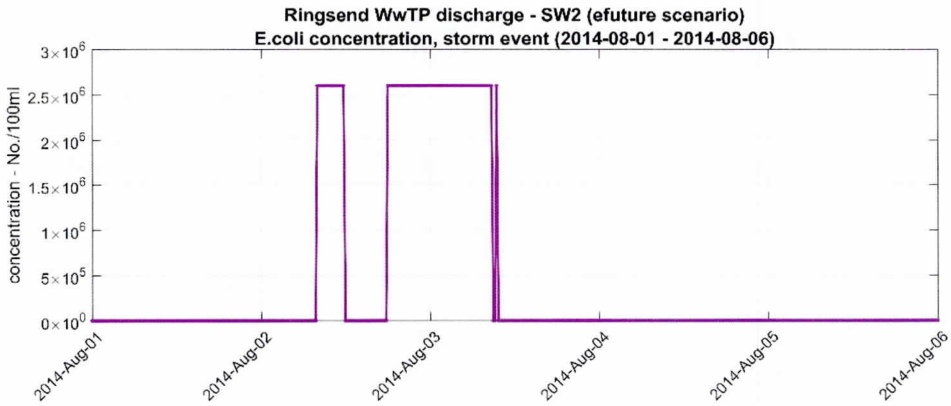


Figure 6.25 Time-series of concentration of E. coli from Ringsend WwTP storm water outfall SW2, before, during and after the summer storm event (2nd – 3rd August 2014) for future discharge scenario.

6.4.4.2 Background Concentrations

In the context of the present work, background concentrations refer to pollutant loads from the following point sources as included in the hydrodynamic model:

- Rivers, streams, and canals;
- Sewer overflows; and
- Other wastewater and industrial outfalls.

Background concentrations were not included in every water quality scenario. In order to distinguish the influence of the Ringsend WwTP outfall, background concentrations were omitted. These can be identified in Table 6.2 where the run description states "*no background concentrations*" (e.g. water quality scenario 1.09 and 3.04).

Where included, the background water quality environment was set to represent one of four conditions (three generic background conditions and one specific background condition):

- Annual average conditions;
- Typical winter conditions;
- Typical summer conditions; and
- A summer storm scenario.

Rivers, streams, and canals

Table 6.9 summarises the pollutant concentrations that were specified within the rivers, streams, and canals for the annual, summer, and winter conditions.

The concentration of pollutants from the major rivers in the model domain were determined from the monitoring efforts within the Upper Liffey Estuary and the Tolka Estuary (see section 4.3).

The source concentrations of BOD, Ammonia, DIN, and MRP in the Liffey, Dodder, Grand Canal and Tolka were derived from observed data during the period 2013-2015 at the following locations:

- DB010 – **Liffey** City, Heuston Station upstream of Cammock outfall;
- DB120 – **Dodder/Grand Canal** basin; and
- DB310 – **Tolka** downstream of Annesley Bridge.

As no water quality measurements were available from the Rivers Camac and Santry or the Royal Canal, these values were approximated. Values for the River Liffey were applied to the Camac, the Tolka was used to approximate the River Santry, and the Dodder was used for the Royal Canal. No values were available for either the Elm Park Stream or the Trimleston Stream, and the source concentrations for these sources were set to zero in all modelling scenarios (with the specific exception of the Summer Storm scenario where data on E. coli were available).

For Total Suspended Solids (TSS) only a single observation was available. This sample was taken in the Upper Liffey Estuary at Wood Quay during June 2013. The measured value of 5 mg/l was applied within all rivers specified in the model. The settling velocity for the suspended sediment was estimated to be 0.01 mm/s.

Table 6.10 summarises the concentrations of E. coli that were set for a summer, winter and summer storm scenario. For the storm scenario, the concentrations of E. coli were calculated based on summer time averages from the monitoring within the rivers of Dublin as described in section 4.3.2.

Table 6.9 River pollutant loads as specified in the water quality model scenarios for annual average, summer and winter conditions.

	BOD [mg/l]			TSS [mg/l]			Ammonia [mg/l]			DIN [mg/l N]			MRP [mg/l P]		
River	Annual	Summer	Winter	Annual	Summer	Winter	Annual	Summer	Winter	Annual	Summer	Winter	Annual	Summer	Winter
Liffey	1.5			5			0.08			2.2	2.1	2.3	0.05	0.07	0.02
Dodder	1			5			0.1			0.6	0.4	0.7	0.04	0.02	0.05
Tolka	1			5			0.04			1.7	1.1	2.4	0.02	0.03	0.02
Cammock	1.5			5			0.08			2.2	2.1	2.3	0.05	0.07	0.02
Santry	2			5			0.04			1.7	1.1	2.4	0.02	0.03	0.02
Royal Canal	1			5			0.1			0.6	0.4	0.7	0.04	0.02	0.05
Grand Canal	1			5			0.1			0.6	0.4	0.7	0.04	0.02	0.05
Sluice	3			5						2.8	2.8	2.8	0.06	0.06	0.06
Mayne	5			5						2.1	2.1	2.1	0.09	0.09	0.09
Elm Park Stream															
Trimleston Stream															

Table 6.10 River pollutant loads as specified in the water quality model scenarios for the storm scenario.

River	E. coli [No./100ml]		
	Summer	Winter	Storm
Liffey	250	250	3233
Dodder	250	250	2059
Tolka	250	250	5387
Cammock	250	250	11621
Santry	250	250	2996
Royal Canal	250	250	2059
Grand Canal	250	250	2059
Sluice	250	250	1012
Mayne	250	250	1000
Elm Park Stream	250	250	4000
Trimleston Stream	250	250	5792

Other wastewater and industrial outfalls

Table 6.11 summarises the pollutant concentrations that were specified for the various wastewater and industrial outfalls.

Table 6.12 summarises the concentrations of E. coli that were set for the summer storm scenario.

The concentrations for the Shanganagh WwTP Outfall and the GDD outfall were provided by the GDD project (Ref. /12/).

At Doldrum Bay, the concentration of pollutants in the raw sewage were based on published data and on the information available from (Ref. /14/).

The Dublin Combined Sewer Overflows were only active in the summer storm scenario. The concentration of E. coli was assumed to be half of the raw sewage value. This value was agreed between J.B. Barry and Irish Water.

For the two power stations (Synergen and Covanta), it was assumed that clean water was discharged.

Table 6.11 Outfall pollutant loads as specified in the water quality model scenarios for annual average, summer and winter conditions.

Outfall	BOD [mg/l]			TSS [mg/l]			Ammonia [mg/l]			DIN [mg/l N]			MRP [mg/l P]		
	Annual	Summer	Winter	Annual	Summer	Winter	Annual	Summer	Winter	Annual	Summer	Winter	Annual	Summer	Winter
Shanganagh WwTP Outfall	7	7	7	0	0	0	0	0	0	14.4	14.4	14.4	3	3	3
SynerGen Power Station	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Covanta WtE Plant	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
GDD Outfall	25	25	25	0	0	0	0	0	0	50	50	50	10	10	10
Doldrum Bay Outfall	350	350	350	5	5	5	45	45	45	60	60	60	10	10	10

Table 6.12 Outfall pollutant loads as specified in the water quality model scenarios for summer storm conditions.

Outfall	E. coli [No./100ml]		
	Summer	Winter	Storm
Shanganagh WwTP Outfall	1.00×10^5	1.00×10^5	1.00×10^5
SynerGen Power Station	0	0	0
Covanta WtE Plant	0	0	0
GDD Outfall	3.91×10^4	3.91×10^4	3.91×10^4
Doldrum Bay Outfall	1.00×10^7	1.00×10^7	1.00×10^7
Dublin Storm Water Overflows (SWO's)			5.00×10^6

6.4.5 Initial Concentrations

The initial conditions of the various pollutants in the wider water quality model were set according to the long-term average values from sampling locations within Dublin Bay as described in Section 4.3.

Table 6.13 shows the values set for annual average, summer, winter, and storm conditions.

Table 6.13 Initial conditions specified for water quality modelling.

Pollutant	Initial Concentrations			
	Average	Summer	Winter	Storm
BOD [mg/l]	0.75			
TSS [mg/l]	0			
Ammonia [mg/l]	0.02			
DIN [mg/l N]	0.09	0.05	0.2	
MRP [mg/l P]	0.02	0.02	0.02	
E. coli [No./100ml]		0	0	0

6.5 Validation of Existing Baseline Scenario

Validation of the water quality model was performed by comparing modelled concentrations of DIN, MRP and BOD against observed data from the monitoring efforts within Dublin Bay, the Liffey Estuary, and the Tolka Estuary (see section 4.3.1).

The hydrodynamic and transport model for summer and winter conditions were run for two consecutive spring-neap tidal cycles. The first spring-neap tidal cycle was designated as a model "warm up" period. The model results were therefore only extracted for the second spring-neap tidal cycle.

The water quality model setup represented typical conditions during the period 2013-2015 rather than specific events. On the other hand, the discrete nature of the water quality sampling represents a greater variability due to the specific conditions at the time (for example meteorological events or tidal stage). The water quality model validation was, therefore, assessed by comparing the statistical range of modelled and observed values with respect to the environmental quality standards (Table 6.14). For DIN and MRP, this was based on the median concentration. For BOD, the status was based on the concentration below which 95% of the data were found (or in other words, the concentration that is exceeded by 5% of the dataset).

Note that, in most cases, the water quality sampling was heavily biased towards the summer months. This gives greater statistical confidence in the water quality model performance during summer conditions.

Table 6.14 Environmental Quality Standard (EQS) as specified in the European Communities Environmental Objectives Surface Waters 2009 (Ref. /4/).

<table>
<tr><th>Parameter</th><th>Description</th><th>Transitional water body</th><th>Coastal water body</th></tr>
<tr><td>BOD</td><td rowspan="3">European communities environmental objectives (surface waters) regulations 2009</td><td>95 %ile concentration:
≤ 4 mg/l</td><td>N.A.</td></tr>
<tr><td>DIN</td><td>N.A.</td><td>Median concentration:
≤ 0.17 mg/l (High status)
≤ 0.25 mg/l (Good status)</td></tr>
<tr><td>MRP</td><td>Median concentration:
≤ 0.04 mg/l</td><td>N.A.</td></tr>
</table>

6.5.1 Transitional Waters

For the transitional waters (the Lower Liffey Estuary and Tolka Estuary) three locations were selected for water quality model validation.

- DB210 – Lower Liffey Estuary, downstream of East Link Toll Bridge;
- DB340 – Tolka Estuary, Clontarf Boat Club; and
- DB420 – Lower Liffey Estuary, Poolbeg Lighthouse.

These three were chosen as they represent three distinct areas within the estuary (see Figure 6.26). Location DB210 was located on the Lower Liffey, upstream of the Ringsend WwTP outfall. DB340 represents the conditions in the Tolka Estuary. Finally, DB420 was located downstream of the Ringsend WwTP outfall at the Poolbeg Lighthouse by the entrance to Dublin Harbour.

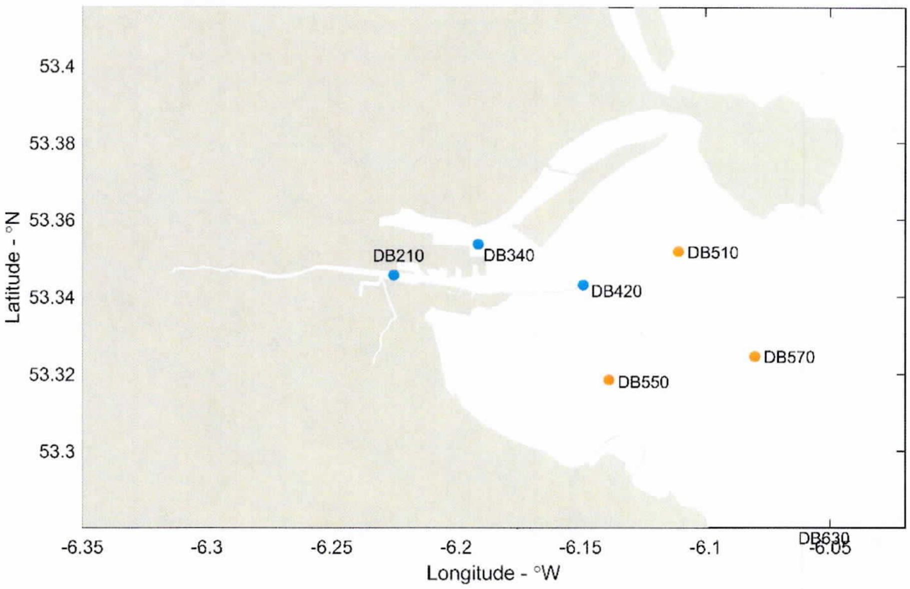


Figure 6.26 Map of Dublin Harbour and Dublin Bay showing locations of water quality monitoring stations chosen for water quality model validation. Blue dots show the location in the transitional waters. Orange dots show the locations in coastal waters.

BOD

Figure 6.27 shows observed and modelled concentration of BOD at DB210, DB340 and DB420.

In all cases the 95-percentile concentration of BOD (signified by the whiskers in Figure 6.27) were below the Environmental Quality Standards (EQS) for transitional surface waters in both the observed sampling datasets and the model predictions.

MRP

Figure 6.27 shows observed and modelled surface concentration of MRP at DB210, DB340 and DB420.

At all three locations, the modelled concentrations of MRP were found to provide a very good description of the observed concentrations.

At location DB210 and DB420, median MRP concentrations were lower than the EQS for transitional waters for both modelled and observed data and provide a very good validation.

Within the Tolka Estuary at DB340, the observed summer surface samples gave median concentration of MRP that was slightly above the EQS for transitional waters. Whereas the model gave a median concentration that was slightly below the EQS. The difference is most likely due to the discrete nature of the water quality sampling where one or two relatively high samples skew the distribution. Notwithstanding, the range of the model results show it is well matched to the 25-75% range of the observed samples. This gives confidence in the representation of MRP concentrations in coastal waters by the water quality model.

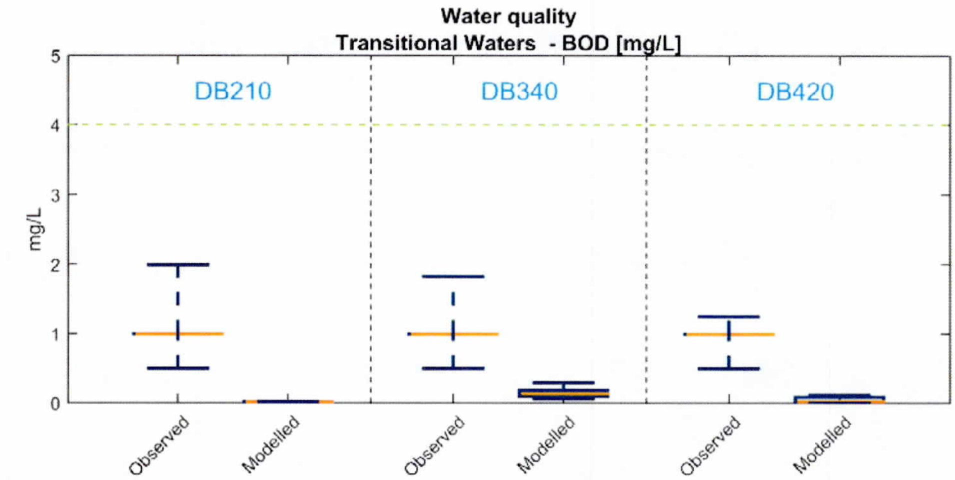


Figure 6.27 Concentration of observed and modelled BOD in the transitional waters (surface sample), representing averaging period 2013 – 2015. Horizontal orange line shows the median concentration. The blue box shows the range of the range of the 25 – 75% quantile and whiskers show the range of the 5 – 95% quantile. The dashed green lines show the environmental quality standard for good status.

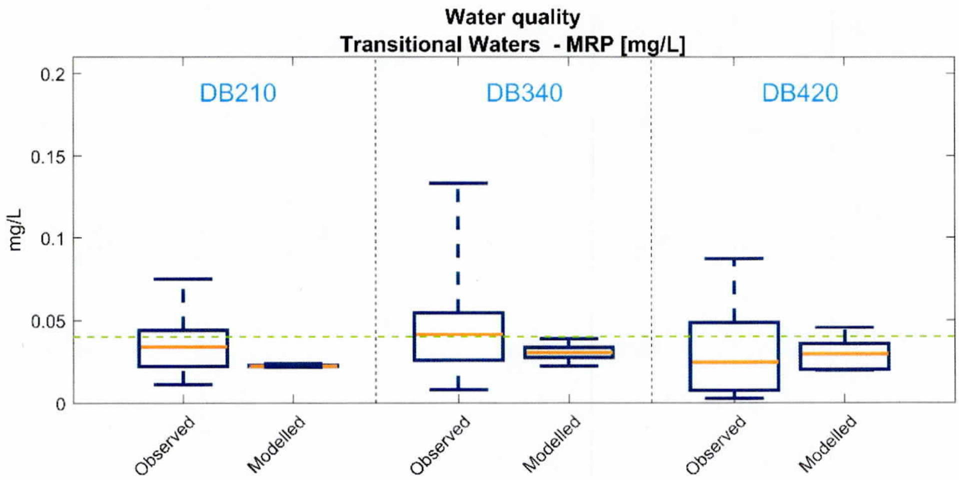


Figure 6.28 Concentration of observed and modelled MRP in the transitional waters (surface sample), representing averaging period 2013 – 2015. Horizontal orange line shows the median concentration. The blue box shows the range of the range of the 25 – 75% quantile and whiskers show the range of the 5 – 95% quantile. The dashed green lines show the environmental quality standard for good status.

6.5.2 Coastal Waters

For the coastal waters sites three locations were selected for water quality model validation.

- DB510 – 2.5 kilometres ENE of Poolbeg Lighthouse;
- DB550 – No. 4 Buoy, 2.5 kilometres E of S. Poolbeg Lighthouse; and
- DB570 – 5 kilometres ESE of Poolbeg Lighthouse.

The three locations represent the northern, southern and outer areas within Dublin Bay (see Figure 6.26).

DIN

Figure 6.29 shows observed and modelled concentration of DIN at DB510, DB550 and DB570.

The median concentration from both the observed and modelled data satisfied the EQS for high status in coastal waters.

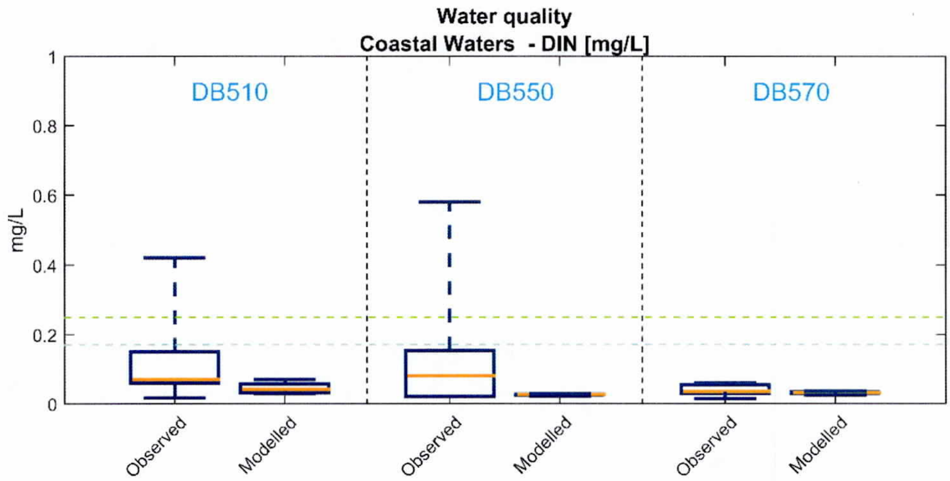


Figure 6.29 Concentration of observed and modelled DIN in the coastal waters (composite sample) during summer conditions (average over 2013 – 2015). Horizontal orange line shows the median concentration. The blue box shows the range of the range of the 25 – 75% quantile and whiskers show the range of the 5 – 95% quantile. The dashed blue and green lines show the environmental quality standard for high status and good status, respectively.

6.5.3 Summary of Water Quality Model Validation

It is apparent from the above model validation that even with the discrete nature of the sampling programme, the water quality model represented the key processes of pollutant dispersal.

The hydrodynamic and water quality models represented the decay of the measured indicators. With the previous knowledge of the model validity for the principal physical controls, it was assessed that the model was suitable for the assessment of the changes to be implemented as part of the future scenario modelling. It was considered that the modelling is relevant for producing difference plots showing the change due to the proposed scheme.

7 Scenario Modelling Results

The results of the hydrodynamic and water quality model scenarios (existing and future discharge environment) as outlined in Section 6 are presented in the following section.

The hydrodynamic and transport model were run for two (2) consecutive spring-neap tidal cycles. The first of these cycles was designated as a model spin-up period and the analysis was only performed on results from the second spring-neap tidal cycle. The exception was for the summer storm scenarios, where only the two-day storm event from 2nd – 3rd August 2014 was considered (again following a suitable model spin up period). The focus of these modelling scenarios is on understanding the changes from the existing situation to the "with scheme" situation.

7.1 Hydrodynamics

The changes in the hydrodynamics as described in section 6.2 and summarised in Table 6.1. The principal changes to the sources and structures that may impact on the flow in the estuary and Dublin Bay were:

- Increase in the discharge water volumes from the Ringsend WwTP;
- Discharge of relatively high temperature water from the Covanta WtE plant outfall; and
- Repair of the ESB cooling water channel and weir at the Ringsend WwTP outfall.

As the effluent from Ringsend WwTP is discharged to the surface waters of the Lower Liffey Estuary, changes in the surface currents were identified as the most pertinent hydrodynamic receptor. The information below summarises modification to the surface currents between the baseline and future discharge hydrodynamic modelling scenarios.

7.1.1 Existing and Future Discharge Environments - Average Conditions

The surface current speed during average conditions for the existing (hydrodynamic scenario 1) and future discharge environment (hydrodynamic scenario 6) are shown for near-spring ebb and flood conditions in Figure 7.1 and Figure 7.2, respectively. The difference in surface current speeds are also shown, to identify changes between the scenarios.

During ebb tide, there were some localised areas of increased surface current speed along the South Poolbeg Wall, downstream of the Ringsend WwTP and in Dublin Bay, just beyond the terminus of the Poolbeg wall. However, the magnitude of these current speed changes (0.02 – 0.04 m/s) were small in comparison to the background conditions (up to 0.5 m/s).

During flood tide, there were no identified areas of increased/decreased surface current speeds.

The density at the water surface during average conditions for the existing (hydrodynamic scenario 1) and future discharge environment (hydrodynamic scenario 6) are shown for near-spring ebb and flood conditions in Figure 7.3 and Figure 7.4, respectively. The difference in water density at the surface are also shown.

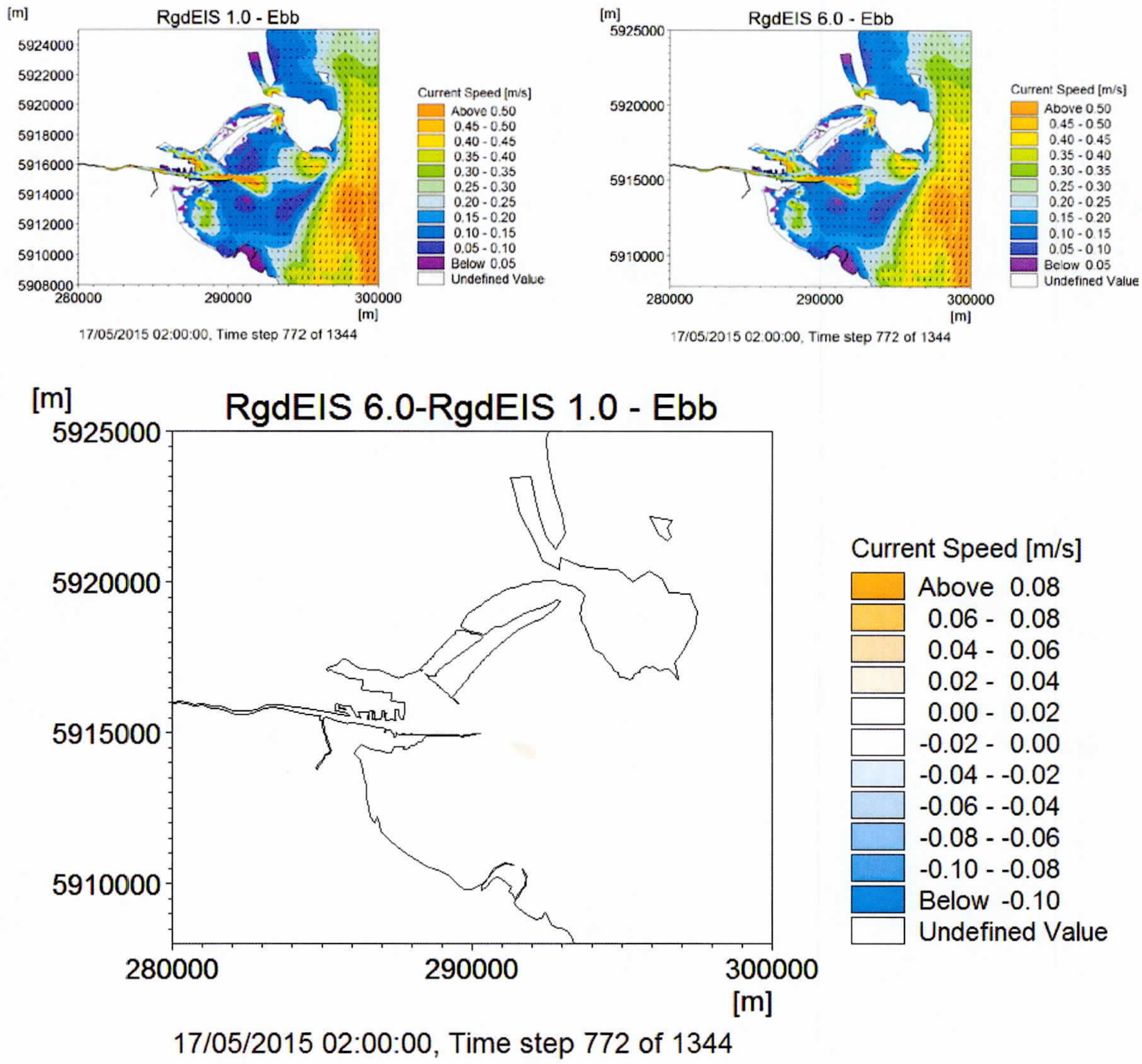


Figure 7.1 Surface current speeds during near-spring ebb tide. Upper-left panel: hydrodynamic model scenario 1 – existing environment, average conditions. Upper-right panel: hydrodynamic model scenario 6 – future discharge, average conditions. Lower panel: difference between future discharge and existing environment. Orange (blue) shaded areas show increased (decreased) surface current speed.

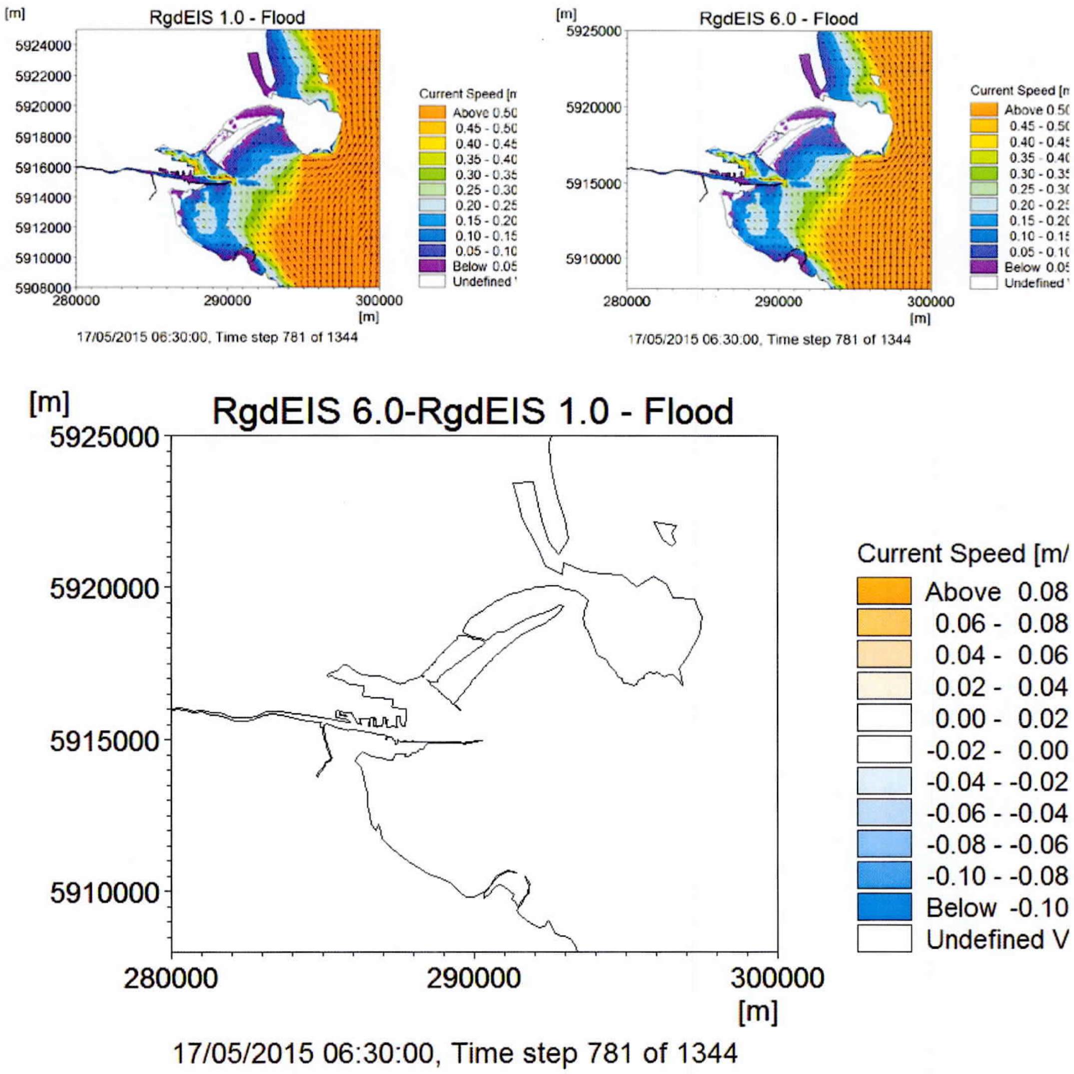


Figure 7.2 Surface current speeds during near-spring flood tide. Upper-left panel: hydrodynamic model scenario 1 – existing environment, average conditions. Upper-right panel: hydrodynamic model scenario 6 – future discharge, average conditions. Lower panel: difference between future discharge and existing environment. Orange (blue) shaded areas show increased (decreased) surface current speed.

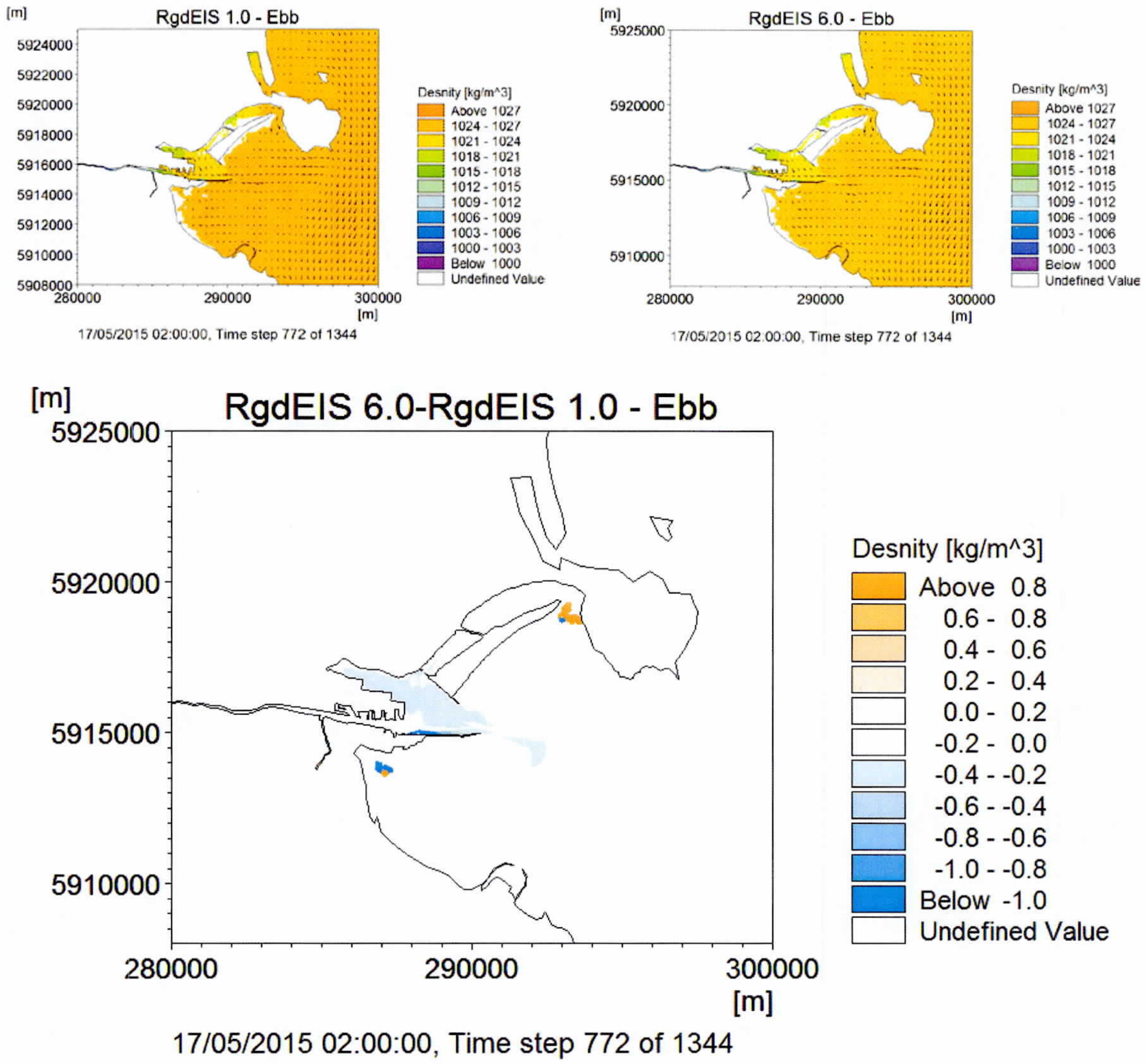


Figure 7.3 Density of surface waters during near-spring ebb tide. Upper-left panel: hydrodynamic model scenario 1 – existing environment, average conditions. Upper-right panel: hydrodynamic model scenario 6 – future discharge, average conditions. Lower panel: difference between future discharge and existing environment. Orange (blue) shaded areas show increased (decreased) water density at the surface.

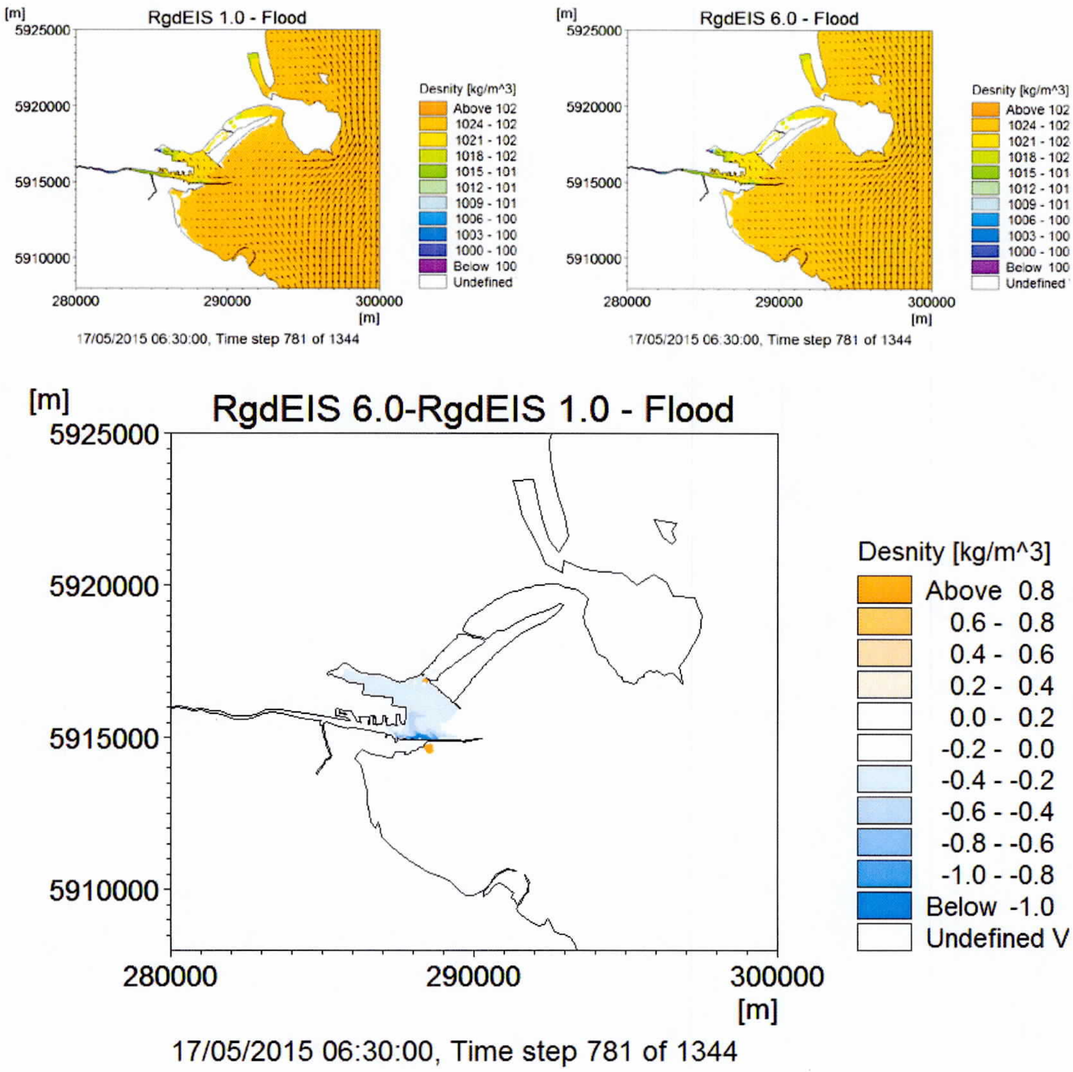


Figure 7.4 Density of surface waters during near-spring flood tide. Upper-left panel: hydrodynamic model scenario 1 – existing environment, average conditions. Upper-right panel: hydrodynamic model scenario 6 – future discharge, average conditions. Lower panel: difference between future discharge and existing environment. Orange (blue) shaded areas show increased (decreased) water density at the surface.

7.2 Water Quality Scenarios

The output from the water quality model scenarios are presented as maps showing the concentration and fate of various pollutants in Dublin Bay and its estuaries. For some scenarios, maps were also produced to show the change in concentration between existing and future discharge environments.

Result maps were produced for all the water quality mode scenarios listed in Table 6.2.

A subset of these results is included in the following sections. The selection of which 'water quality model runs' to include was provided by JB Barry in consultation with Irish water and are summarised in Table 7.1.

The outputs from all water quality model simulations are supplied in a digital format as described in Appendix C.

Table 7.1 Water quality model runs included in results presentation.

Water quality model scenario		Water quality model scenario for comparison		Analysis	
Run No.	**Description**	**Run No.**	**Comparison**	**Type**	**Section**
1.01	BOD – average, existing environment	6.01	BOD – average, future discharge environment	Existing and future discharge environments	7.2.1.1
1.02	BOD – peak concentration, existing environment	6.02	BOD – peak concentration, future discharge environment		7.2.1.1
1.03	TSS – average, existing environment	6.03	TSS – average, future discharge environment		7.2.1.2
1.04	TSS – peak concentration, existing environment	6.04	TSS – peak concentration, future discharge environment		7.2.1.2
1.06	Ammonia (total and un-ionised) – existing environment	6.06	Ammonia (total and un-ionised) – future discharge environment		7.2.1.3
1.07	DIN – average, existing environment	6.07	DIN – average, future discharge environment		7.2.1.4
1.08	MRP – average, existing environment	6.08	MRP– average, future discharge environment		7.2.1.5
2.01	BOD – peak discharge, existing environment	7.01	BOD – peak discharge, future discharge environment		7.2.1.1
2.02	TSS – peak discharge, existing environment	7.02	TSS – peak discharge, future discharge environment		7.2.1.2
3.01	DIN – winter, existing environment	8.01	DIN – winter, future discharge environment		7.2.1.4
3.02	MRP – winter, existing environment	8.02	MRP – winter, future discharge environment		7.2.1.5
4.01	DIN – summer, existing environment	9.01	DIN – summer, future discharge environment		7.2.1.4
4.02	MRP – summer, existing environment	9.02	MRP – summer, future discharge environment		7.2.1.5
4.05	E. coli – summer, existing environment	9.05	E. coli – summer, future discharge environment		7.2.1.6
5.01	E. coli – storm, existing environment	10.01	E. coli – storm, future discharge environment		7.2.1.6
1.01	BOD – average, existing environment	1.02	BOD – peak concentration, existing environment	Construction impacts	7.2.2
1.03	TSS – average, existing environment	1.04	TSS – peak concentration, existing environment		7.2.2
6.09	Conservative tracer – average, future discharge environment	11.07	Conservative tracer – average, future Discharge (Poolbeg Power Station On)	Cumulative impacts	7.2.4.1
6.09	Conservative tracer – average, future discharge environment	14.01	Conservative tracer – average, future Discharge (ESB channel repaired)		7.2.4.2
6.09	Conservative tracer – average, future discharge environment	15.01	Conservative tracer – average, future Discharge (Alexandra Basin Redeveloped)		7.2.4.3

7.2.1 Existing and Future Discharge Environment

Representative concentrations

The mapped concentrations were determined statistically based on the entire simulation period for each water quality model run. For example, the pollutant levels are at or below the 95 percentiles 95% of the time (and are conversely exceeded 5% of the time). Similarly, the load is equal to or below the 50-percentile concentration 50% of the time, and exceeded 50% of the time (this is the definition of the median concentration).

The representative concentration for each of the modelled pollutant were as follows:

- BOD, the 95-percentile concentration over a spring-neap tidal cycle;
- TSS, the 95-percentile concentration over a spring-neap tidal cycle;
- Ammonia (total), the 95-percentile concentration over a spring-neap tidal cycle;
- Ammonia (un-ionised), the 50-percentile (i.e. median) concentration over a spring-neap tidal cycle;
- DIN, the 50-percentile (i.e. median) concentration over a spring-neap tidal cycle;
- MRP, the 50-percentile (i.e. median) concentration over a spring-neap tidal cycle; and
- E. coli. the 95-percentile concentration over a spring-neap tidal cycle.

The list above distinguishes between total ammonia and un-ionised ammonia. It is the un-ionised form that is toxic to marine life such as fish and, therefore, has been considered for water quality. The concentration of un-ionised ammonia was determined from the concentration of total ammonia. The precise relationship between these two forms is difficult to quantity and is dependent on pH and temperature. However, it was agreed with Irish Water that as a conservative estimate, un-ionised ammonia concentrations can be approximated as 2.5% of total ammonia.

For each water quality run, results maps were produced for three (3) different vertical reference levels:

- Concentration at water surface level;
- Depth-average concentration; and
- Concentration at mid-layer of the water column.

Environmental Quality Standard (EQS)

The maps have been colour coded to show the areas that attain (or otherwise exceed) the relevant Environmental Quality Standards (EQS). These values and their representative colour codes are summarised in Table 7.2.

The EQS values for BOD, DIN, MRP and E.coli were set according to criteria specified within the European Communities Environmental Objectives for Surface Waters (Ref. /4/) and Bathing Waters (Ref. /5/) (see Table 3.2 and Table 3.3).

For total ammonia, there are no EQS specified for transitional or coastal water bodies in the European Communities Environmental Objectives for Surface Waters (Ref. /4/). Instead the criteria for river water bodies and lakes is applied. This states that concentrations should be below 0.09 mg/l (high status) and 0.140 mg/l (good status) based on 95% of samples.

For un-ionised ammonia, the EQS was based on those proposed by SEPA (REF) of 0.021 mg/l as an annual mean for estuarine and coastal waters for the protection of saltwater fish and shellfish.

For total suspended solids, no quantitative EQS are specified within the European Communities Environmental Objectives for Surface Waters (Ref. /4/). The results are shown on a scale between 5 mg/l and 35 mg/l. The following general criteria may be used to assess the clarity of the water: clear (< 20 mg/l), cloudy (> 35 mg/l).

Table 7.2 Environmental Quality Standard (EQS) and representative colours used for water quality model results presentation.

Pollutant	Environmental Quality Standard (EQS)				
	White	Blue	Green	Yellow	Orange
BOD [mg/l]	≤4	≤8	>8	N/A	N/A
TSS [mg/l]	≤5	≤10	≤25	≤35	>35
Ammonia (total) [mg N/l]	≤0.09	≤0.14	≤0.28	>0.28	N/A
Ammonia (un-ionised)	≤0.005	≤0.01	≤0.021	>0.021	N/A
DIN [mg N/l]	≤0.17	≤0.25	≤1.4	≤2.6	>2.6
MRP [mg P/l]	≤0.04	≤0.08	≤0.16	>0.16	N/A
E. coli [No./100ml]	≤250	≤500	≤1000	>1000	N/A

7.2.1.1 Biochemical Oxygen Demand (BOD)

For the existing and future discharge environment scenarios, depth-average concentration of BOD exceeded the EQS of 4 mg/l for transitional waters during annual average, peak discharge, and peak flow conditions (see upper panel of Figure 7.5, Figure 7.6, and Figure 7.7, respectively). The area of exceedance above the EQS was limited to the vicinity of the Ringsend WwTP outfall and immediately downstream adjacent to the South Poolbeg Wall. Concentrations within the Upper Liffey Estuary and the Tolka Estuary were within the EQS for transitional waters.

The difference between the future discharge and existing environments showed a reduction in depth-average BOD concentrations within the estuaries. For the annual average conditions, this reduction was seen along the South Poolbeg Wall, downstream of the WwTP outfall (see lower panel of Figure 7.5). For both the peak discharge and peak flow scenarios, the results also show a reduction in BOD concentration within the Tolka Estuary (lower panels of Figure 7.6, and Figure 7.7).

7.2.1.2 Total Suspended Solids (TSS)

For the existing and future discharge environment scenarios, depth-average concentration of TSS were largest in the immediate vicinity of the Ringsend WwTP outfall (see upper panel of Figure 7.8, Figure 7.9, and Figure 7.10 respectively). The maximal concentration was higher in the existing environment (up to 35 mg/l) than for the future discharge scenario (up to c. 25 mg/l).

The difference between the future discharge and existing environments showed a reduction in depth-average TSS concentrations within the Liffey and Tolka estuaries (see lower panel of Figure 7.8-Figure 7.10). The largest reduction was along the South Poolbeg Wall, downstream from the Ringsend WwTP outfall.

7.2.1.3 Ammonia

In the existing environment, depth-average concentration of total ammonia shows values that exceed 0.14 mg/l in much of the Lower Liffey Estuary and the whole of the Tolka Estuary (see upper-left panel of Figure 7.11). In the future discharge scenario, however, the areas of high total ammonia concentration were restricted to the area of the Lower Liffey Estuary around the Ringsend WwTP outfall and the South Poolbeg Wall (see upper-right panel of Figure 7.11).

The change in the water quality environment was an overall reduction in the concentration of total ammonia in the estuaries (see lower panel of Figure 7.11)

For un-ionised form of ammonia, concentration of above 0.01 mg/l were modelled downstream of the Ringsend WwTP (upper-left panel of Figure 7.12). For the future discharge environment, there were no areas with concentration above 0.005 mg/l outside of the Ringsend WwTP outfall channel (upper-right panel of Figure 7.12).

The change in the water quality environment was an overall reduction in the concentration of un-ionised ammonia in the estuaries (see lower panel of Figure 7.12).

7.2.1.4 Dissolved Inorganic Nitrogen (DIN)

DIN is the principal limiting factor in coastal waters and the impact of exceeding the EQS could lead to conditions with the potential to be eutrophic. It is noted that the EQS for DIN do not apply within the transitional water bodies (i.e. the estuaries).

During average conditions, the concentration of DIN in the coastal waters achieved the EQS for high status (median concentration ≤ 0.17 mg/l) in both the existing and future discharge environment (see upper panels of Figure 7.13).

During winter conditions, the concentration of DIN in the coastal waters achieved the EQS for high status (median concentration ≤ 0.17 mg/l) in the south of Dublin Bay in both the existing and future discharge environment. In the north of Dublin Bay, the EQS for good status (median concentration ≤ 0.25 mg/l) was achieved.

During summer conditions, the concentration of DIN in the coastal waters achieved the EQS for high status (median concentration ≤ 0.17 mg/l) in both the existing and future discharge environment (see upper panels of Figure 7.15).

There was no overall significant change in the coastal waters with respect to concentrations of DIN during average, winter or summer conditions (see lower panels of Figure 7.13, Figure 7.14 and Figure 3.1Figure 7.15).

7.2.1.5 Molybdate Reactive Phosphate (MRP)

MRP is a limiting nutrient in transitional water bodies. It is noted that the EQS for MRP does not apply in the coastal water bodies.

During average, winter and summer conditions, the concentration of MRP in the existing environment scenario exceeded the EQS of 0.04 mg/l along the South Poolbeg Wall and within Tolka Estuary (upper-left panels of Figure 7.16, Figure 7.17 and Figure 7.18).

In the future discharge scenario, the areas with MRP concentration above the EQS were restricted to the area downstream of the Ringsend WwTP outfall and adjacent to the South Poolbeg Wall Figure 7.17 (upper-right panels of Figure 7.16, Figure 7.17 and Figure 7.18).

There was an overall decrease in the concentration of MRP for the future discharge scenario within the transitional waters during average, winter and summer conditions (see lower panel of Figure 7.16, Figure 7.17 and Figure 7.18).

7.2.1.6 E.coli

During summer conditions, there was an overall increase in E.coli concentration in the Lower Liffey and Tolka estuaries in the future discharge environment (lower panel of Figure 7.19). The predicted increase was since the volume of effluent discharged during summer conditions was ~40% larger in the future discharge environment, whereas the concentration of E.coli in the treated effluent was invariant at 1.00×10^5 per 100 ml. As a result, the total pollutant load discharged in the future scenario was larger, and this is reflected in the elevated concentrations in the Liffey Estuary.

Bathing Waters

There are three EU designated beaches within Dublin Bay: Dollymount Strand, Sandymount Strand, and Merrion Strand (see Figure 3.2).

The results of the water quality modelling scenarios show that there was no deterioration in the water quality at the three bathing waters and that excellent quality is predicted at each of the beaches.

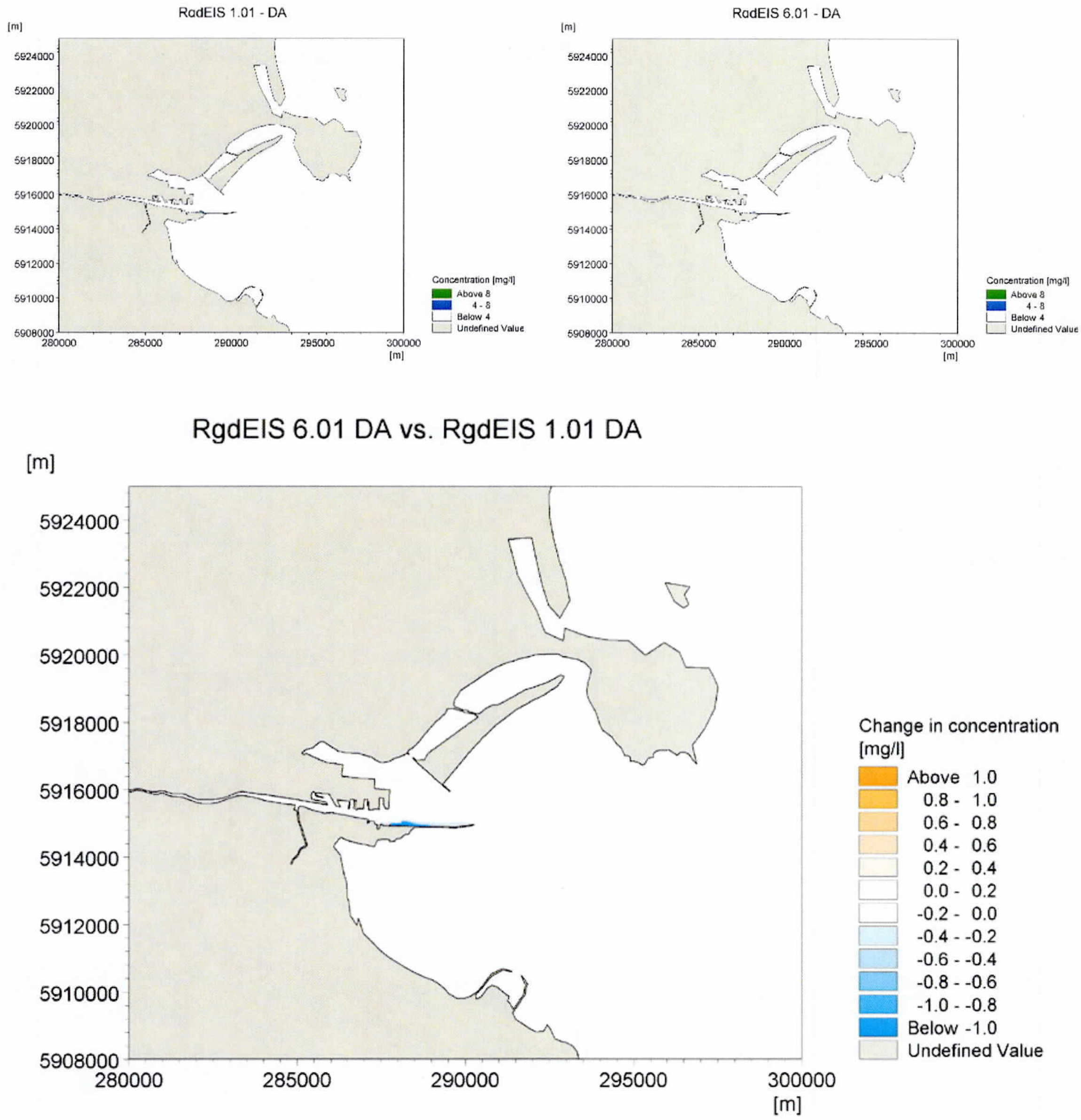


Figure 7.5 Concentration of BOD [mg/l, 95%ile, depth-average]. Upper-left panel: water-quality model scenario 1.01 – existing environment, average conditions. Upper-right panel: water-quality model scenario 6.01 – future discharge, average conditions. Lower panel: difference between scenario 6.01 and 1.01 with orange (blue) shaded areas show increased (decreased) in concentration.

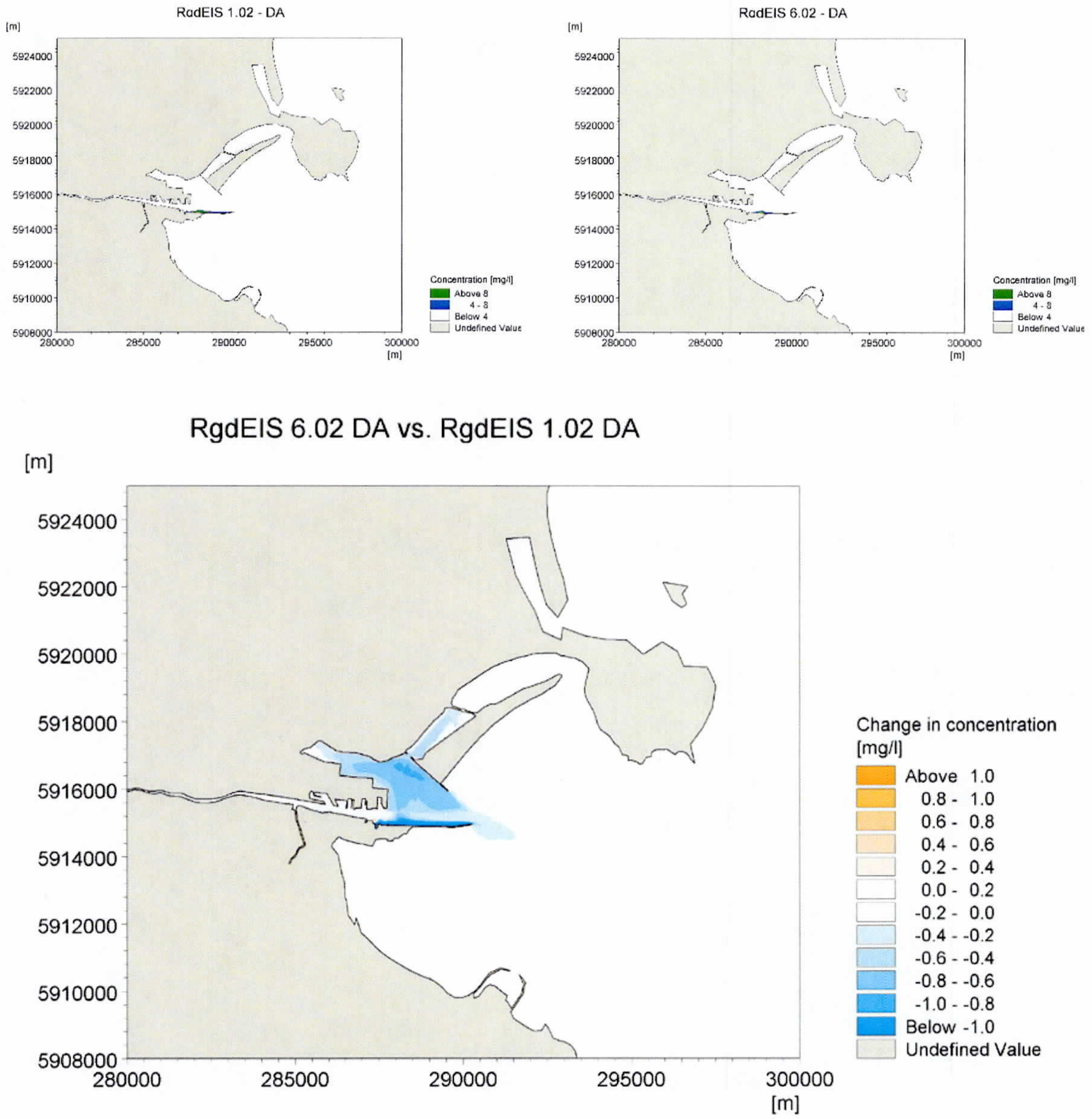


Figure 7.6 Concentration of BOD [mg/l, 95%ile, depth-average]. Upper-left panel: water-quality model scenario 1.02 – existing environment, average conditions, peak discharge. Upper-right panel: water-quality model scenario 6.02 – future discharge, average conditions, peak discharge. Lower panel: difference between scenario 6.02 and 1.02 with orange (blue) shaded areas show increased (decreased) in concentration.

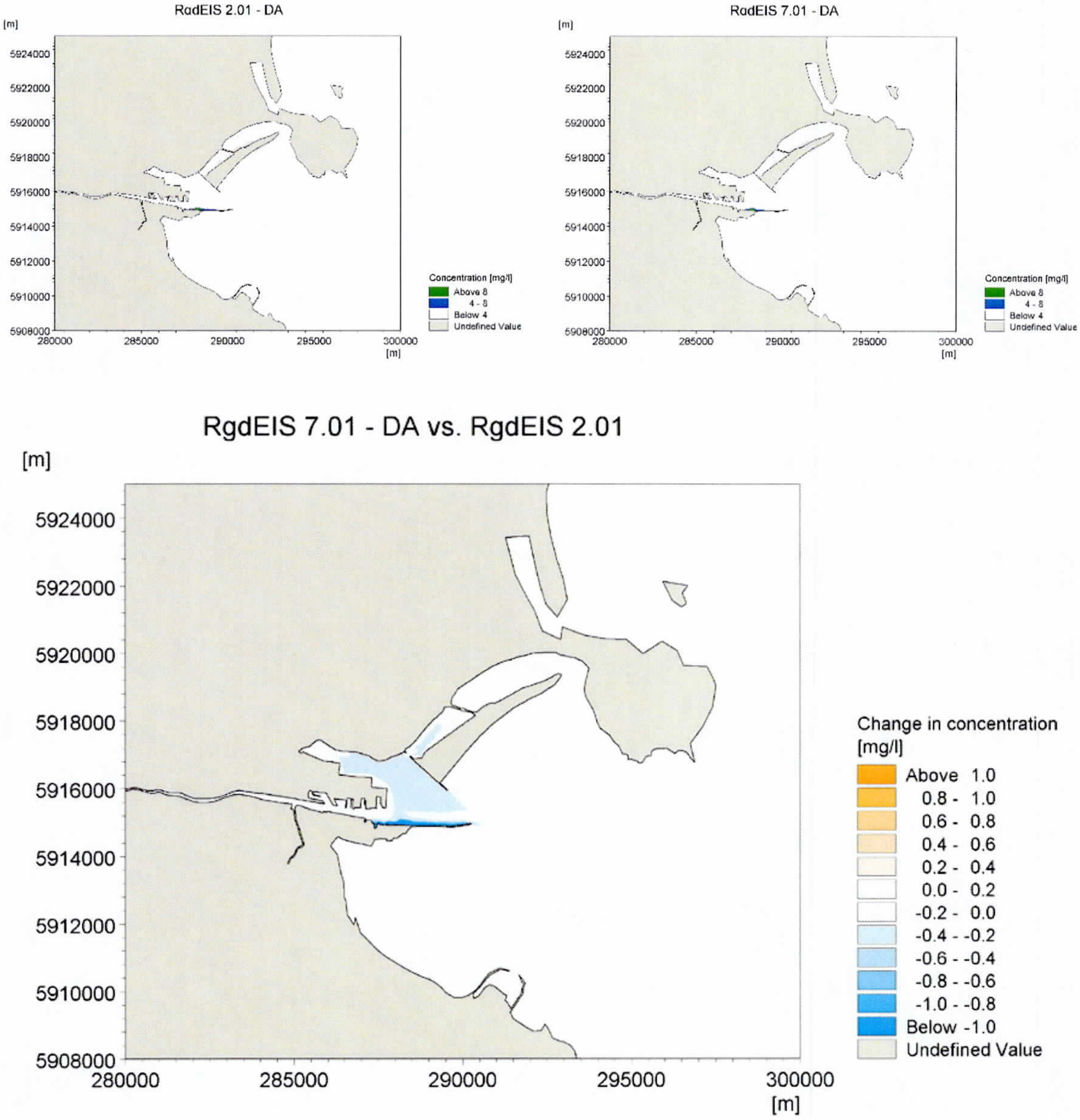


Figure 7.7 Concentration of BOD [mg/l, 95%ile, depth-average]. Upper-left panel: water-quality model scenario 2.01 – existing environment, peak flow conditions. Upper-right panel: water-quality model scenario 7.01 – future discharge, peak flow conditions. Lower panel: difference between scenario 7.01 and 2.01 with orange (blue) shaded areas show increased (decreased) in concentration.

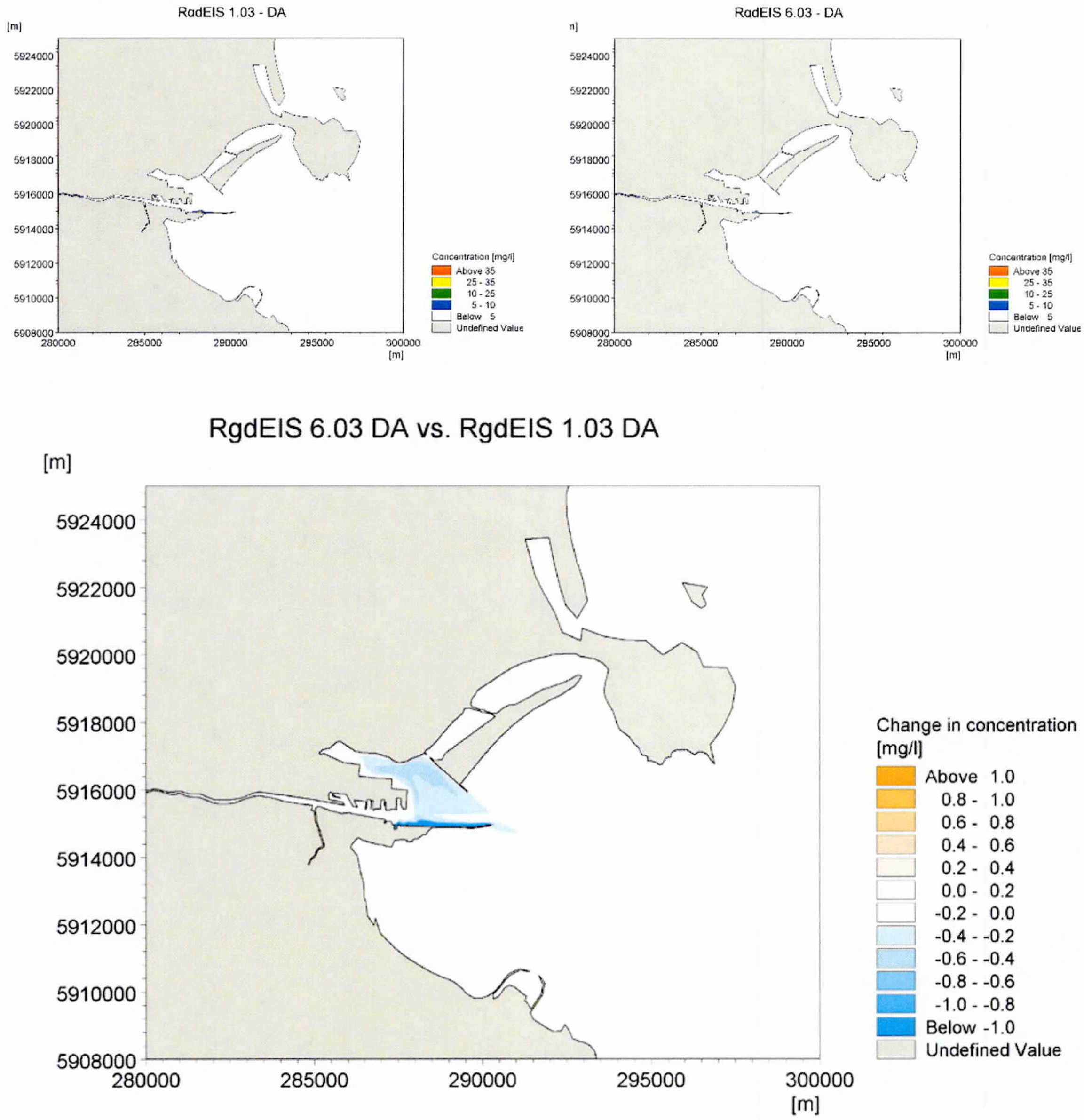


Figure 7.8 Concentration of TSS [mg/l, 95%ile, depth-average]. Upper-left panel: water-quality model scenario 1.03 – existing environment, average conditions. Upper-right panel: water-quality model scenario 6.03 – future discharge, average conditions. Lower panel: difference between scenario 6.01 and 1.01 with orange (blue) shaded areas show increased (decreased) in concentration.

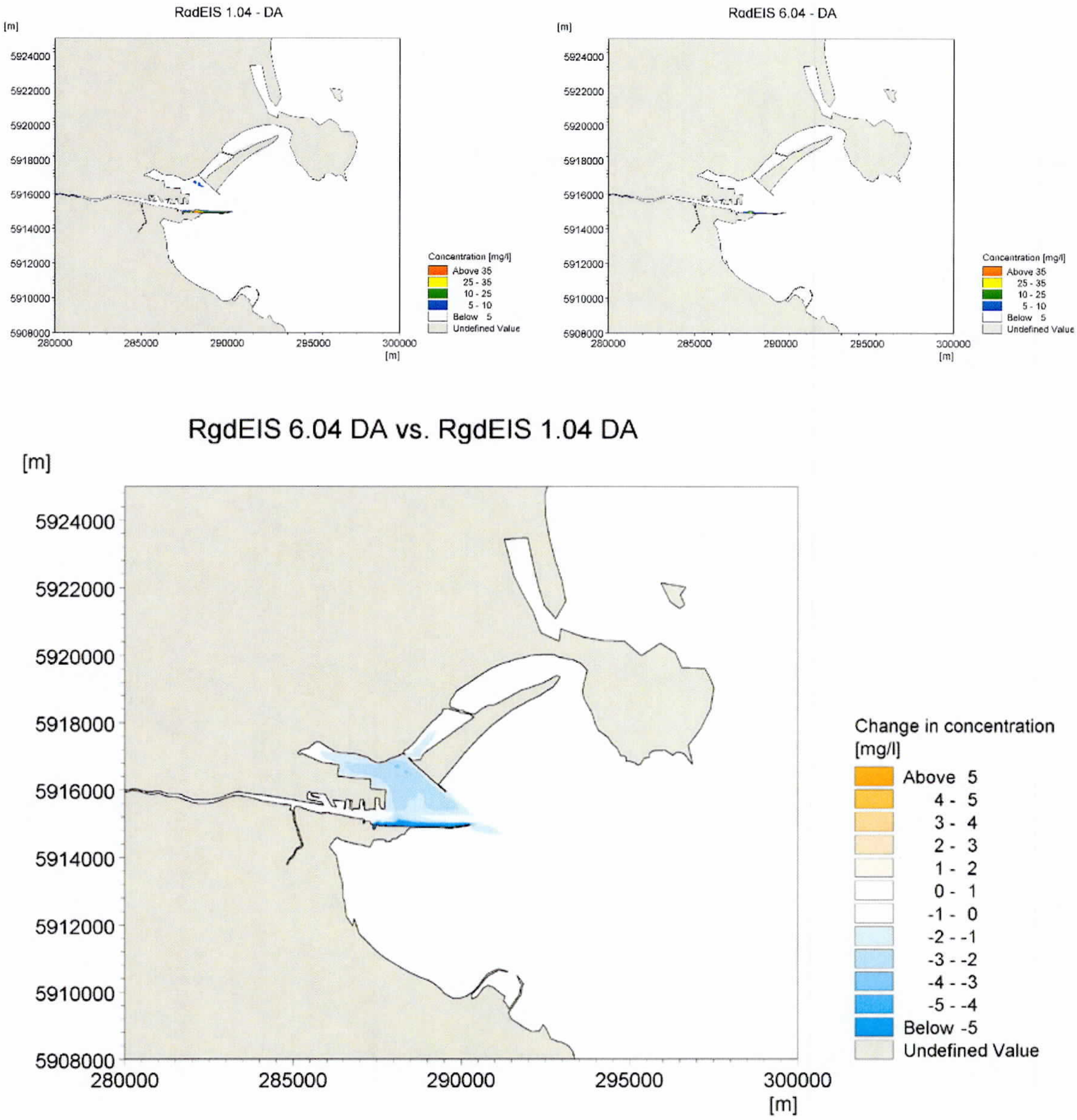


Figure 7.9 Concentration of TSS [mg/l, 95%ile, depth-average]. Upper-left panel: water-quality model scenario 1.04 – existing environment, average conditions, peak discharge. Upper-right panel: water-quality model scenario 6.04 – future discharge, average conditions, peak discharge. Lower panel: difference between scenario 6.04 and 1.04 with orange (blue) shaded areas show increased (decreased) in concentration.

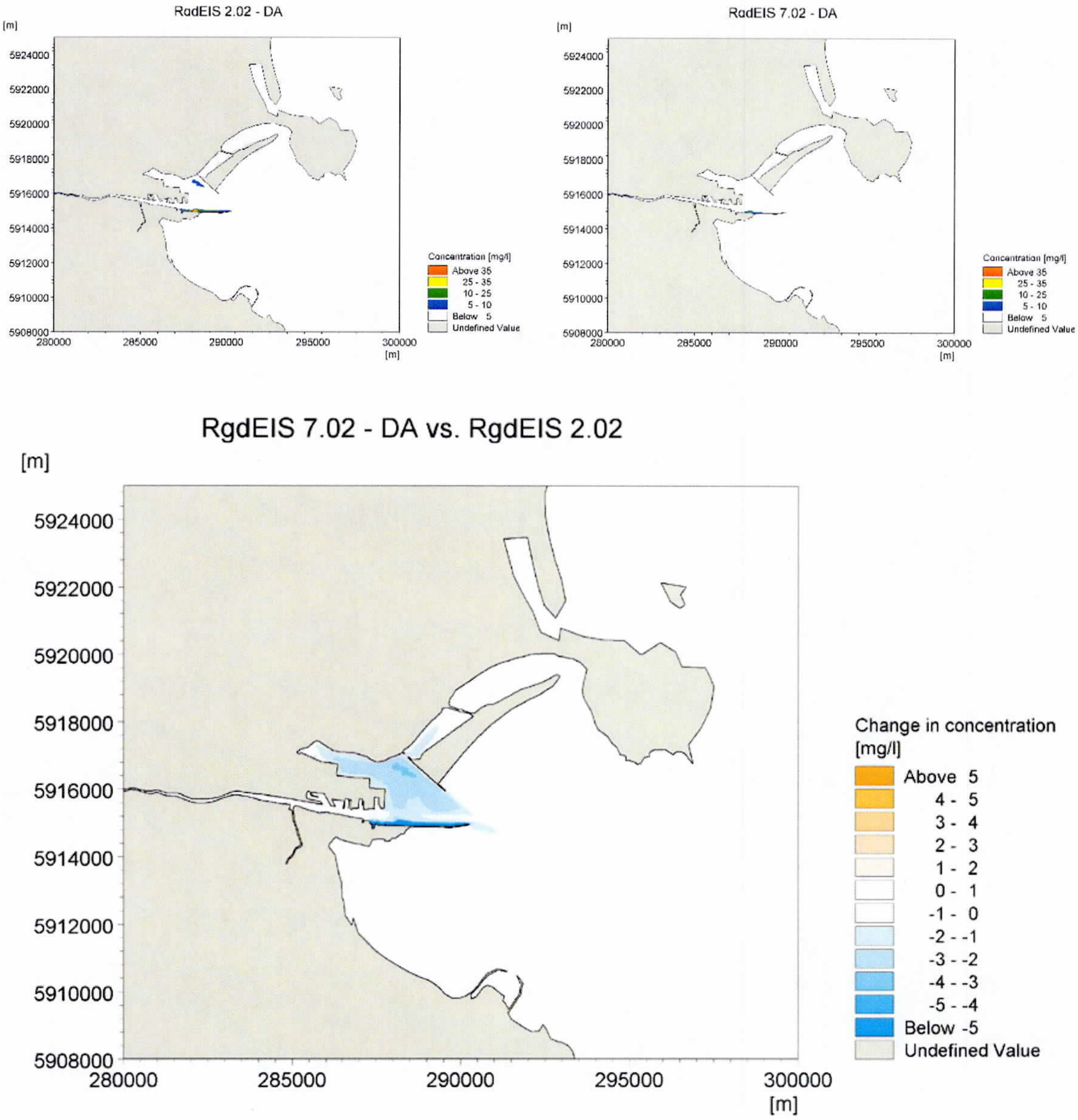


Figure 7.10 Concentration of TSS [mg/l, 95%ile, depth-average]. Upper-left panel: water-quality model scenario 2.01 – existing environment, peak flow conditions. Upper-right panel: water-quality model scenario 7.01 – future discharge, peak flow conditions. Lower panel: difference between scenario 7.01 and 2.01 with orange (blue) shaded areas show increased (decreased) in concentration.

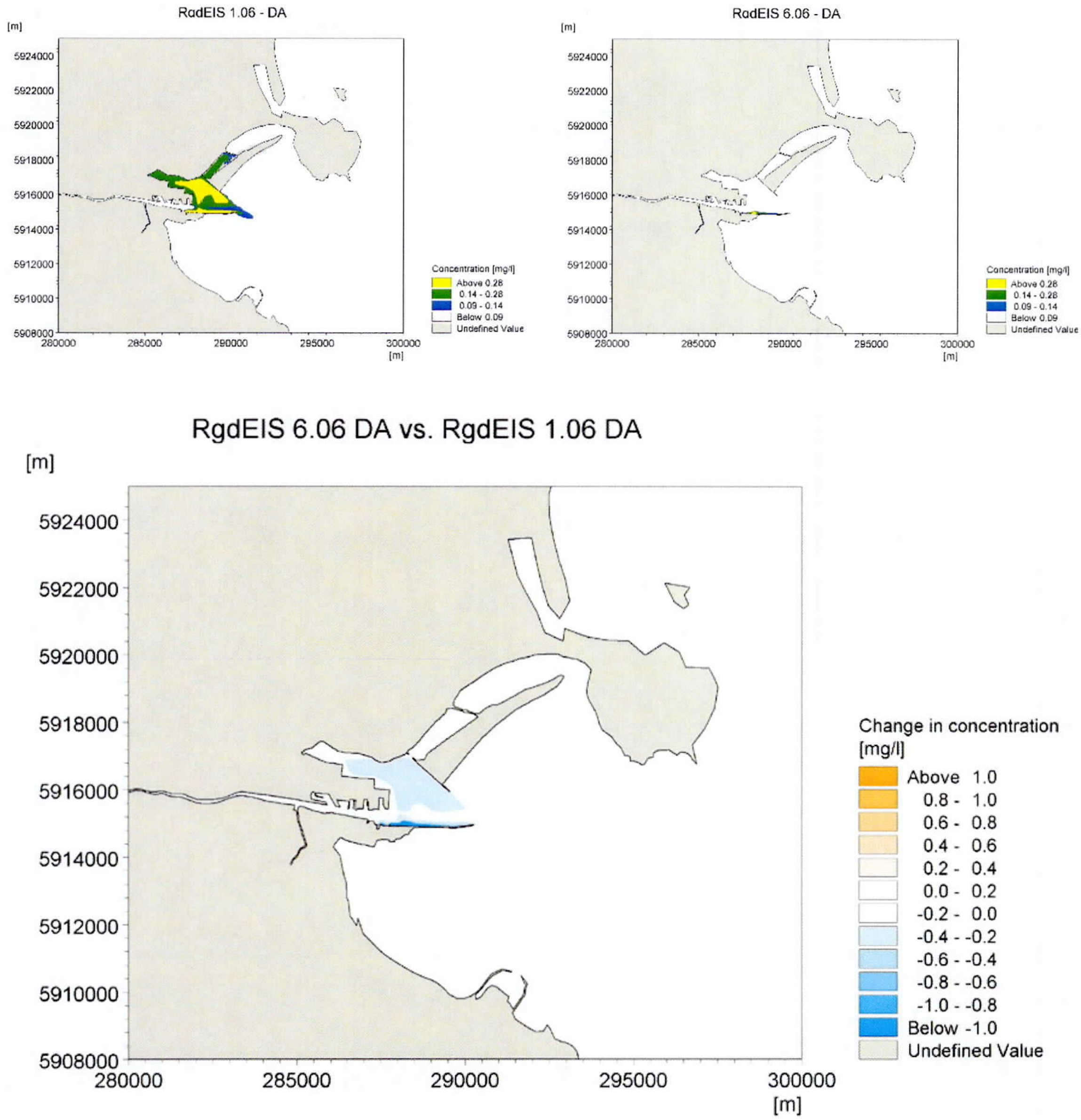


Figure 7.11 Concentration of total ammonia [mg/l, 95%ile, depth-average]. Upper-left panel: water-quality model scenario 1.06 – existing environment, average conditions. Upper-right panel: water-quality model scenario 6.06 – future discharge, average conditions. Lower panel: difference between scenario 6.06 and 1.06 with orange (blue) shaded areas show increased (decreased) in concentration.

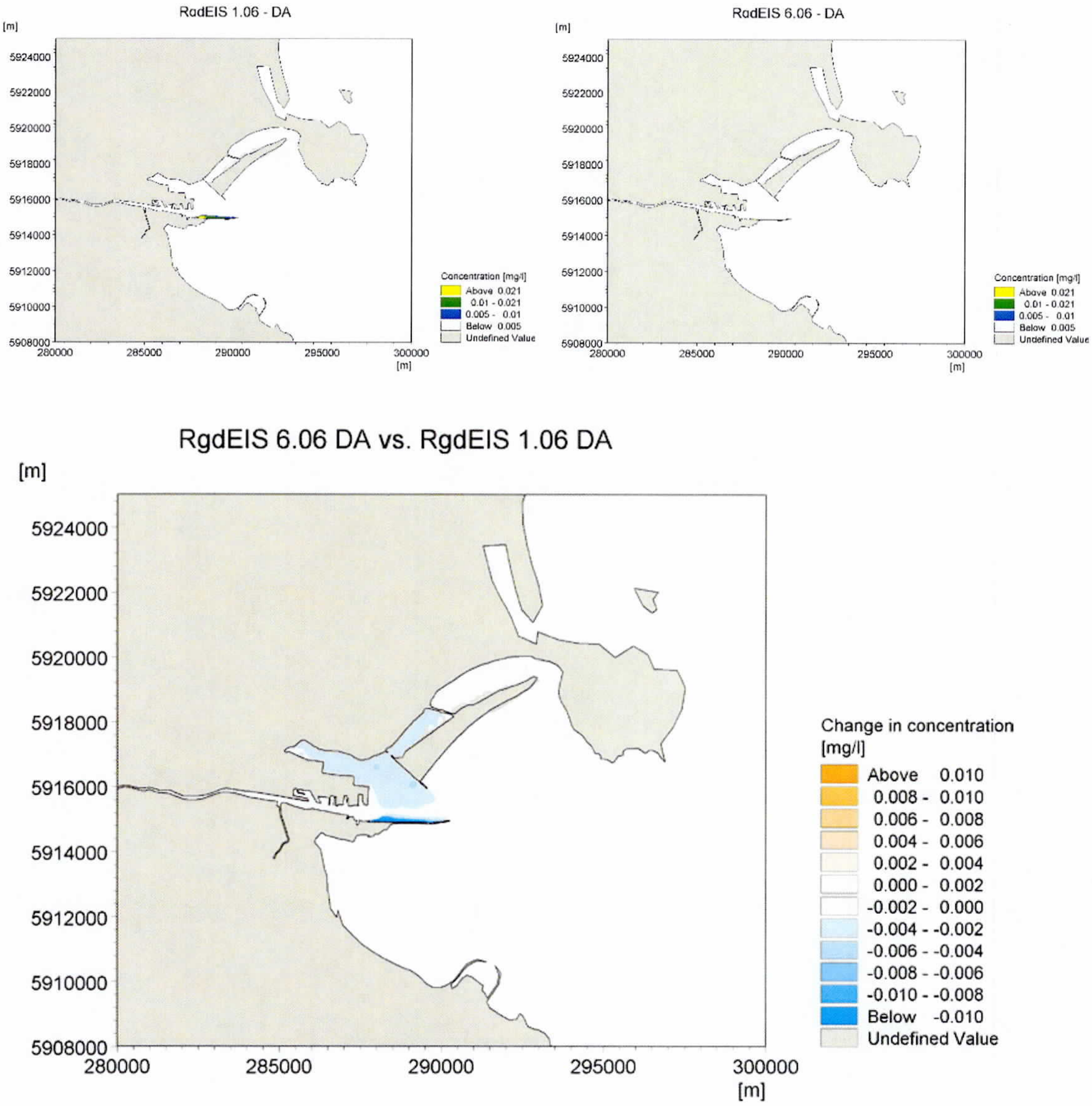


Figure 7.12 Concentration of un-ionised ammonia [mg/l, 50%ile, depth-average]. Upper-left panel: water-quality model scenario 1.06 – existing environment, average conditions. Upper-right panel: water-quality model scenario 6.06 – future discharge, average conditions. Lower panel: difference between scenario 6.06 and 1.06 with orange (blue) shaded areas show increased (decreased) in concentration.

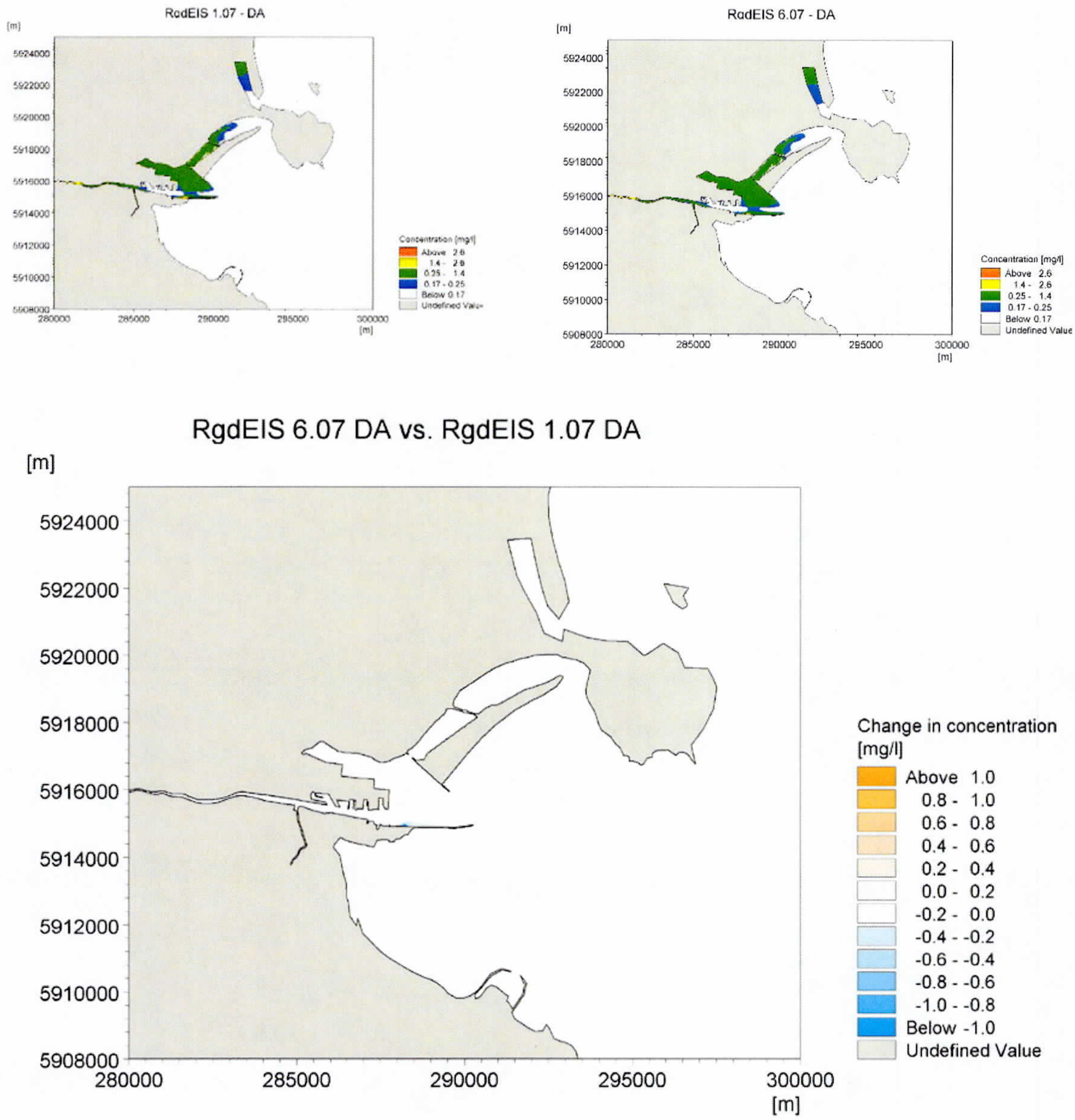


Figure 7.13 Concentration of DIN [mg/l, 50%ile, depth-average]. Upper-left panel: water-quality model scenario 1.07 – existing environment, average conditions. Upper-right panel: water-quality model scenario 6.07 – future discharge, average conditions. Lower panel: difference between scenario 6.07 and 1.07 with orange (blue) shaded areas show increased (decreased) in concentration.

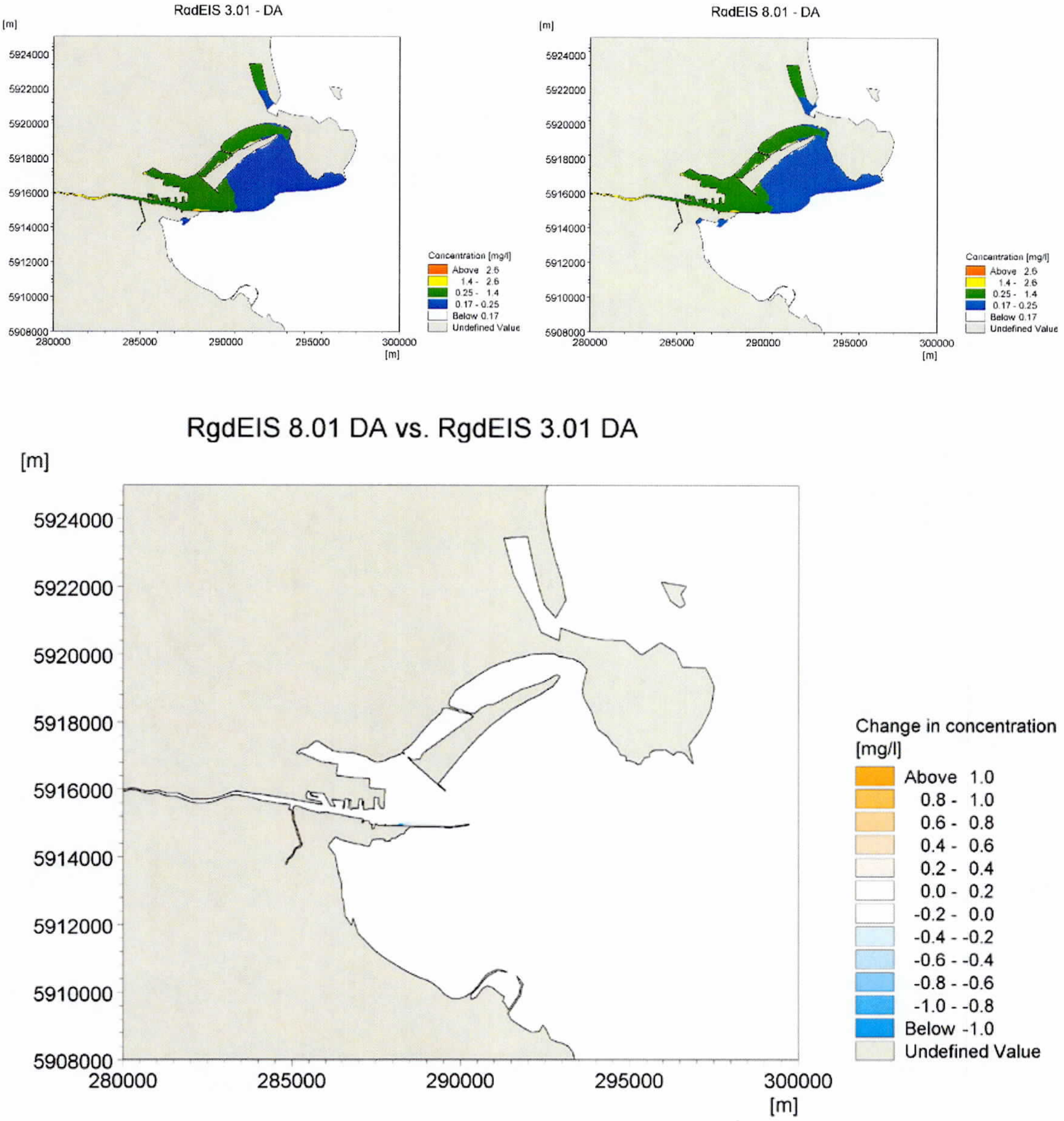


Figure 7.14 Concentration of DIN [mg/l, 50%ile, depth-average]. Upper-left panel: water-quality model scenario 3.01 – existing environment, winter conditions. Upper-right panel: water-quality model scenario 8.01 – future discharge, winter conditions. Lower panel: difference between scenario 8.01 and 3.01 with orange (blue) shaded areas show increased (decreased) in concentration.

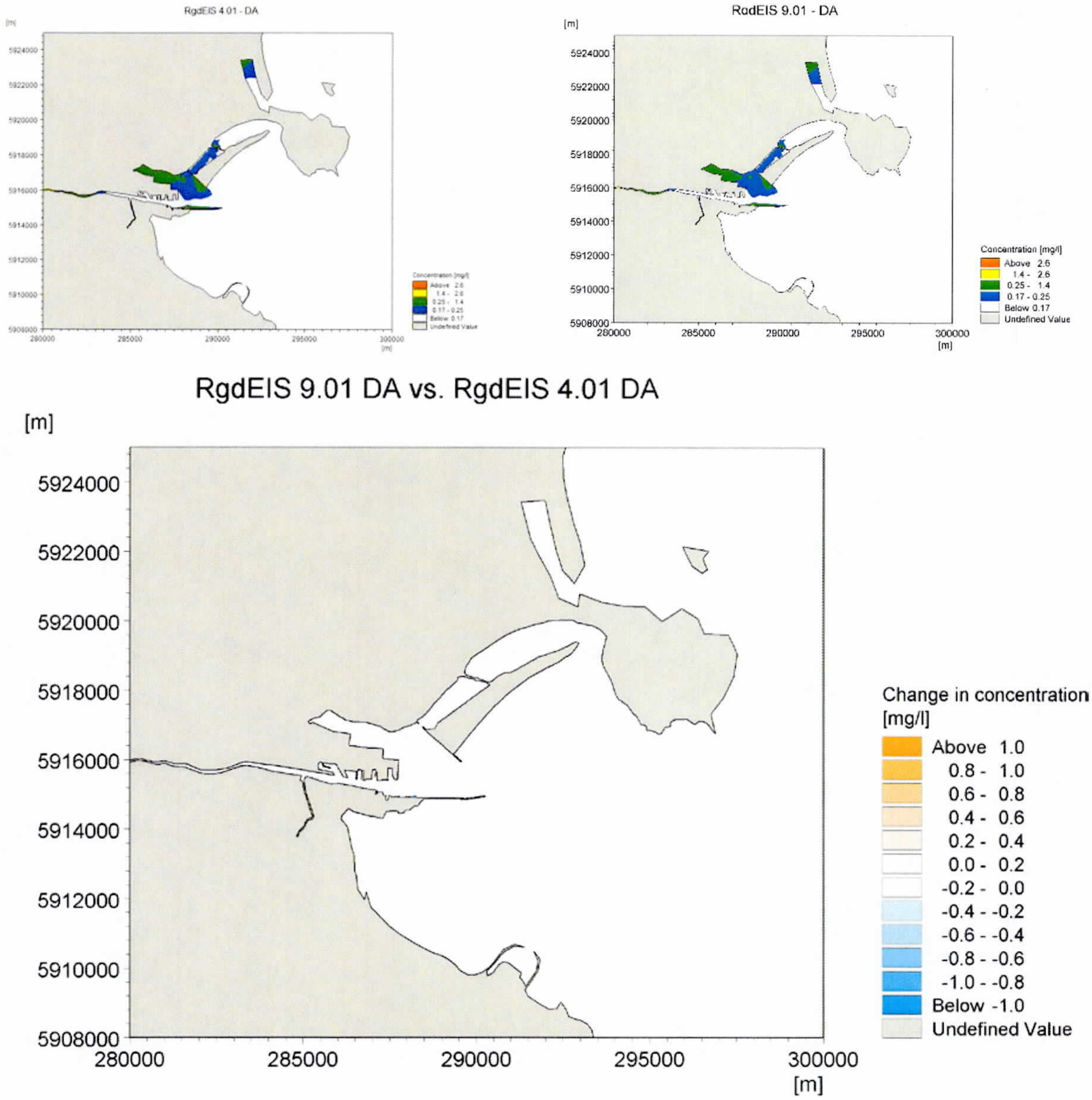


Figure 7.15 Concentration of DIN [mg/l, 50%ile, depth-average]. Upper-left panel: water-quality model scenario 4.01 – existing environment, summer conditions. Upper-right panel: water-quality model scenario 9.01 – future discharge, summer conditions. Lower panel: difference between scenario 9.01 and 4.01 with orange (blue) shaded areas show increased (decreased) in concentration.

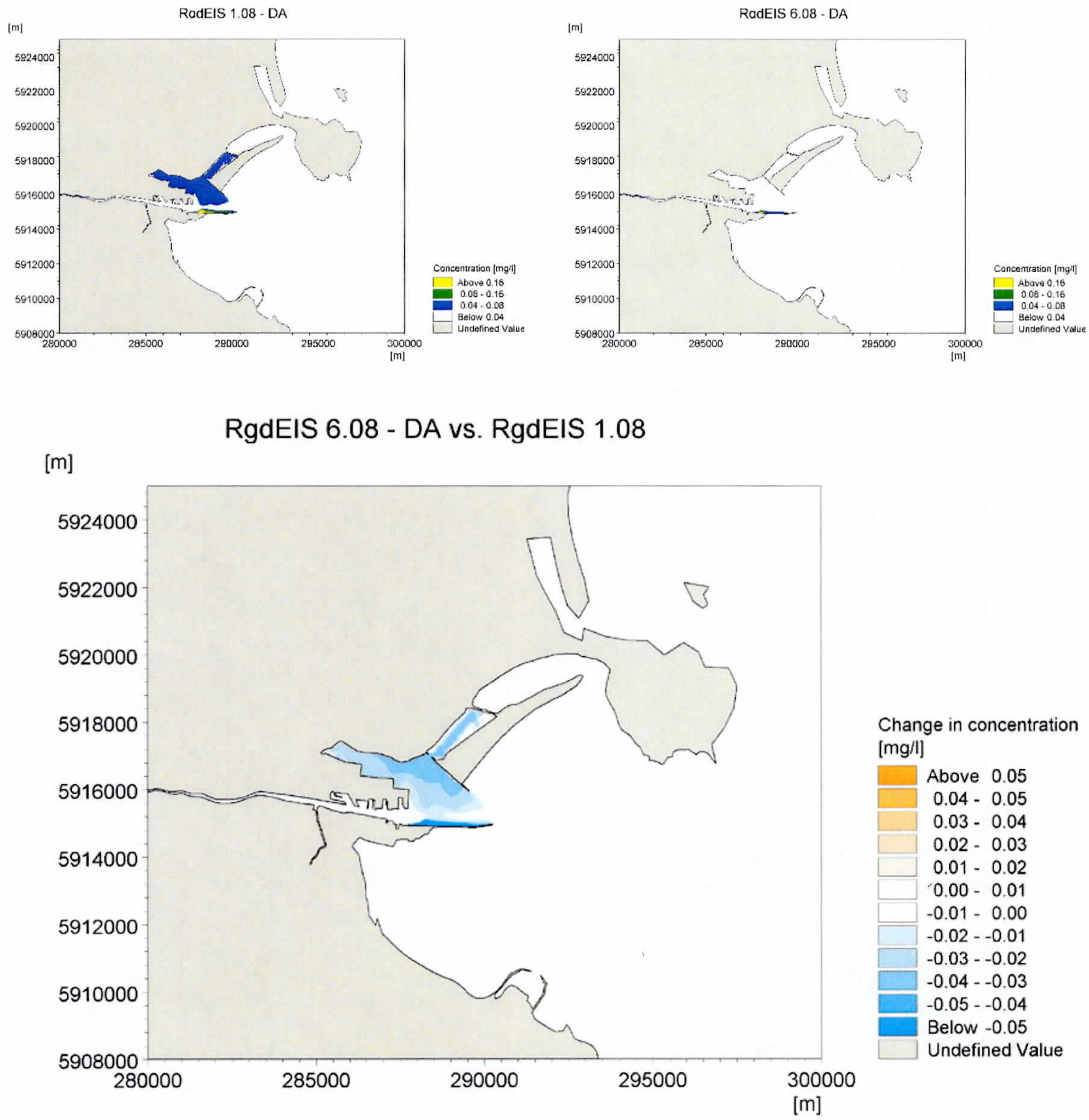


Figure 7.16 Concentration of MRP [mg/l, 50%ile, depth-average]. Upper-left panel: water-quality model scenario 1.08 – existing environment, average conditions. Upper-right panel: water-quality model scenario 6.08 – future discharge, average conditions. Lower panel: difference between scenario 6.08 and 1.08 with orange (blue) shaded areas show increased (decreased) in concentration.

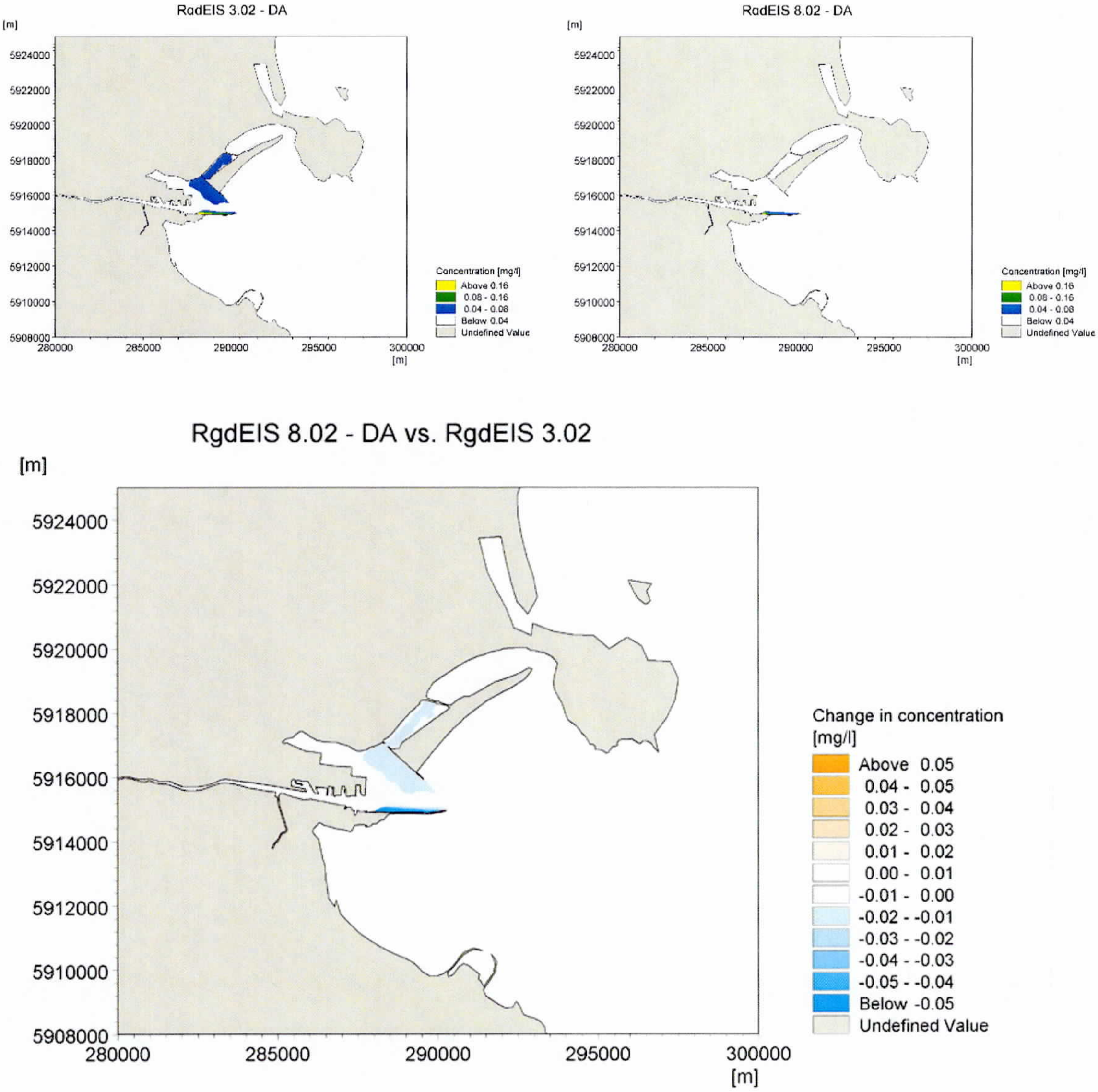


Figure 7.17 Concentration of MRP [mg/l, 50%ile, depth-average]. Upper-left panel: water-quality model scenario 8.02 – existing environment, winter conditions. Upper-right panel: water-quality model scenario 8.02 – future discharge, winter conditions. Lower panel: difference between scenario 8.02 and 3.02 with orange (blue) shaded areas show increased (decreased) in concentration.

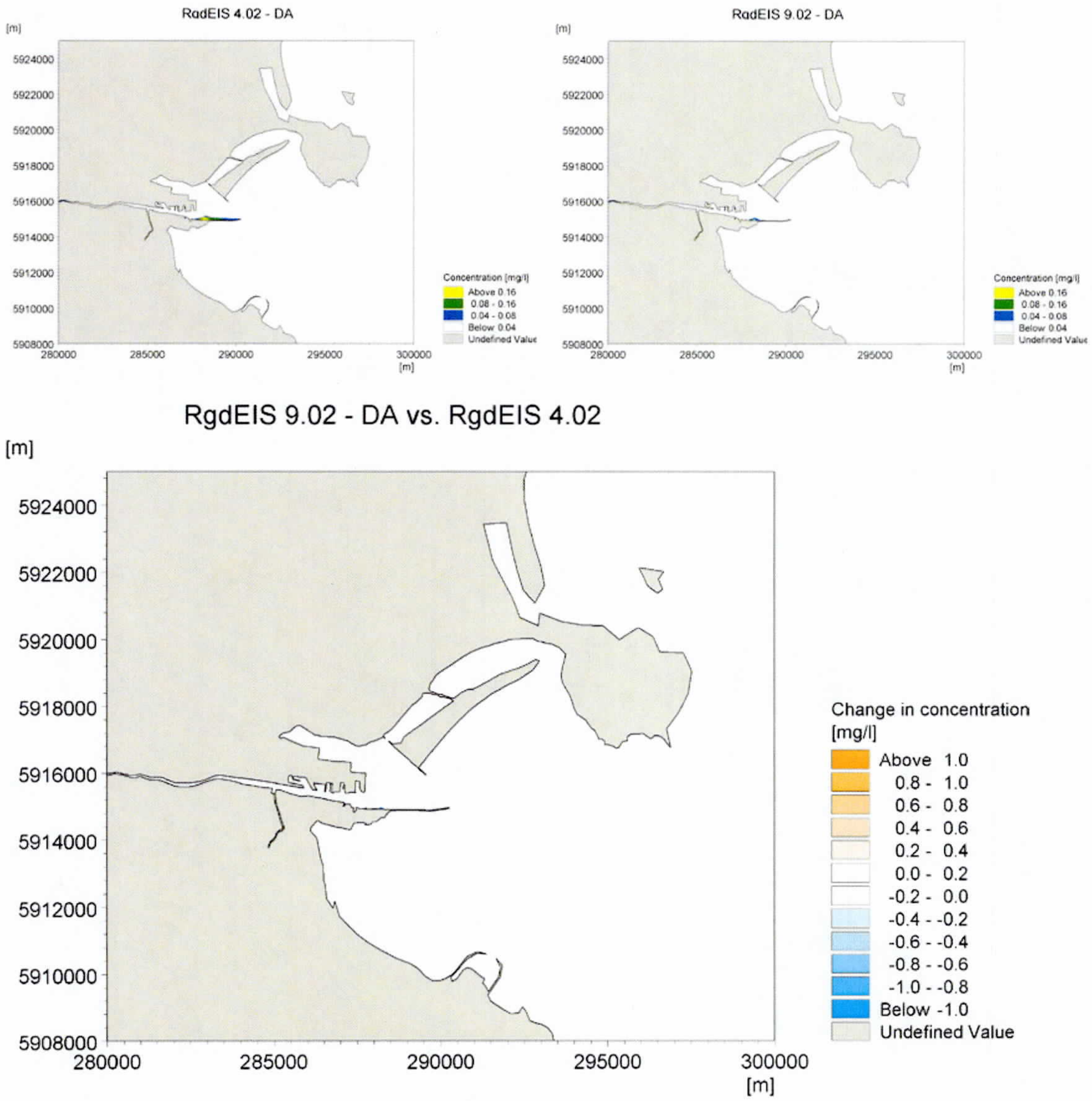


Figure 7.18 Concentration of MRP [mg/l, 50%ile, depth-average]. Upper-left panel: water-quality model scenario 4.02 – existing environment, summer conditions. Upper-right panel: water-quality model scenario 9.02 – future discharge, summer conditions. Lower panel: difference between scenario 9.02 and 4.02 with orange (blue) shaded areas show increased (decreased) in concentration.

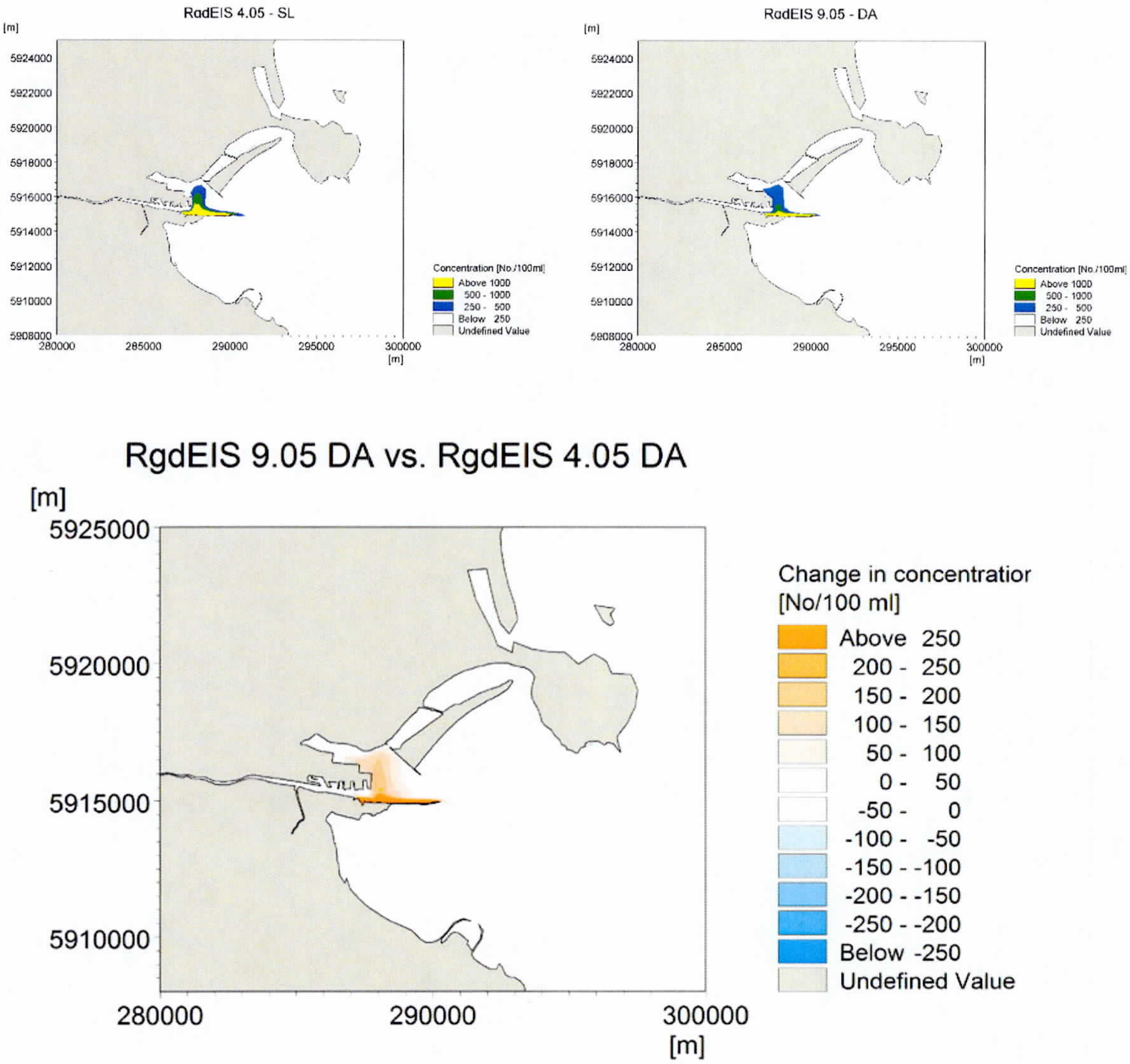


Figure 7.19 Concentration of E. coli [No/100 m/l, 95%ile, surface]. Upper-left panel: water-quality model scenario 4.05 – existing environment, summer conditions. Upper-right panel: water-quality model scenario 9.05 – future discharge, summer conditions. Lower panel: difference between scenario 4.05 and 9.05 with orange (blue) shaded areas show increased (decreased) in concentration.

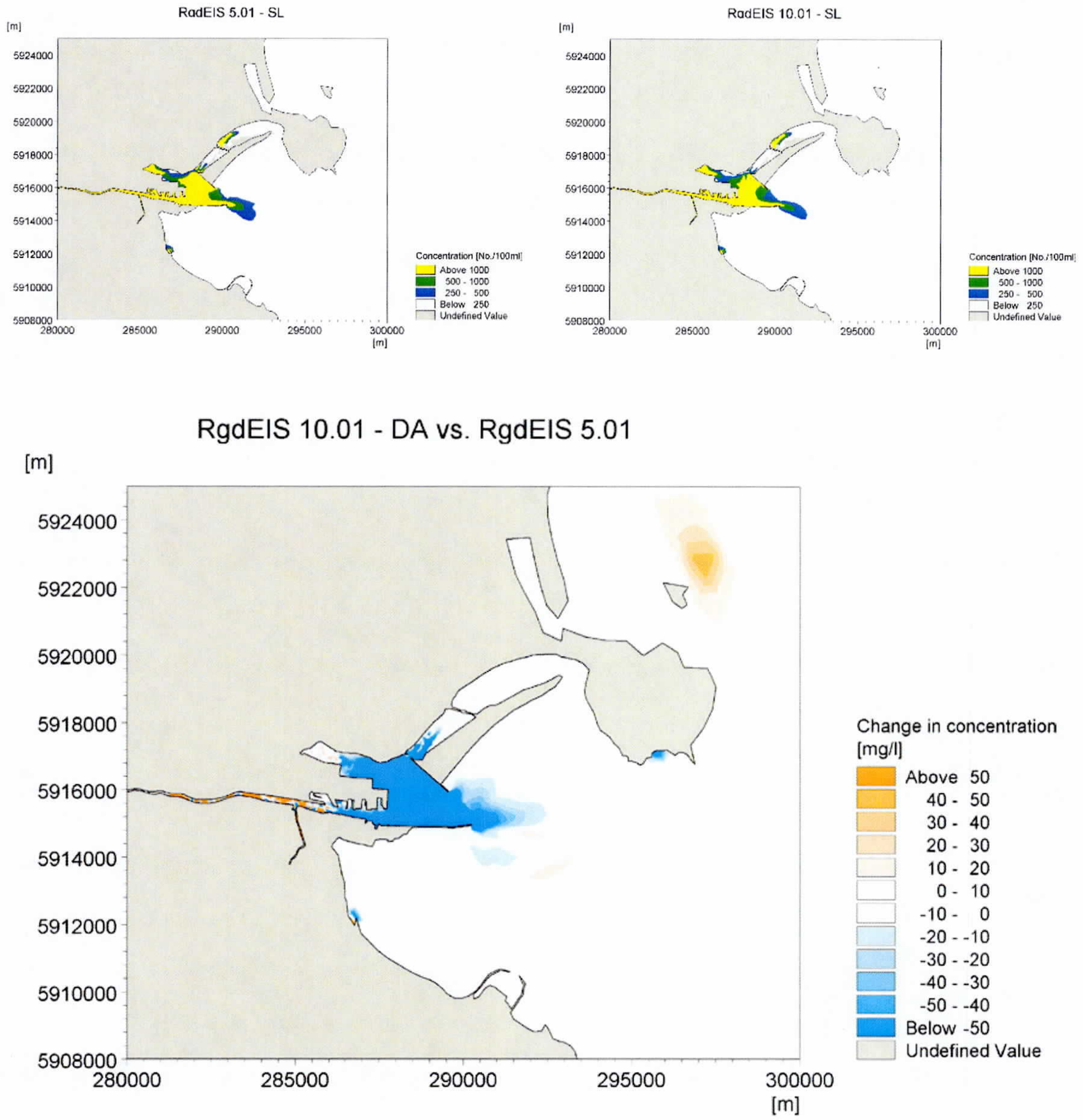


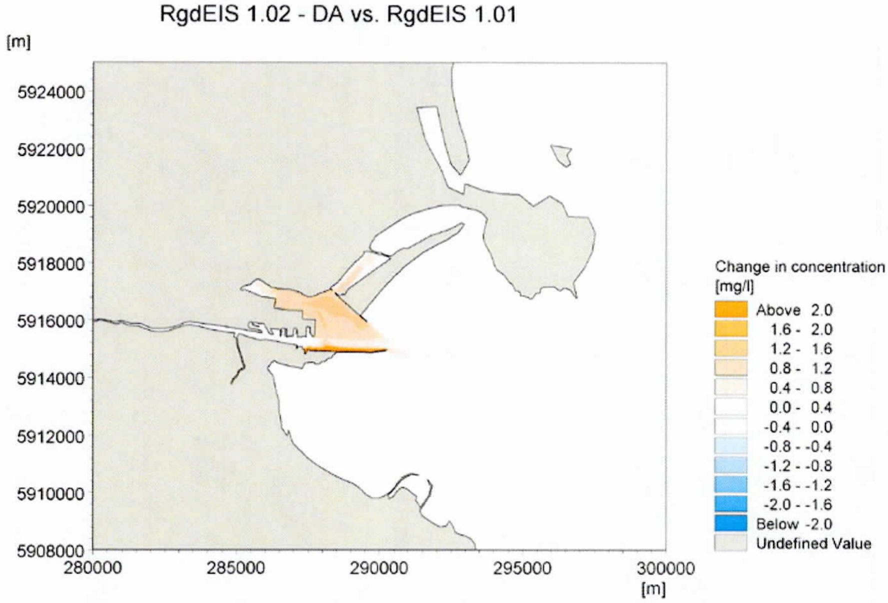
Figure 7.20 Concentration of E. coli [No/100 m/l, 95%ile, surface]. Upper-left panel: water-quality model scenario 5.01 – existing environment, storm conditions. Upper-right panel: water-quality model scenario 10.01 – future discharge, storm conditions. Lower panel: absolute difference between scenario 10.01 and 5.01 with orange (blue) shaded areas show increased (decreased) in concentration.

7.2.2 Construction Impacts

As it is anticipated that the upgrade and refit of the Ringsend WwTP will overlap, Irish Water requested that consideration be given to the effects of peak events during this phase (so called construction impacts). The potential effects of construction impacts were predicted by comparing the peak and average flow scenarios during the existing environmental conditions (see construction impacts in Table 7.1).

Figure 7.21 shows absolute and percentage change in the 95-percentile depth-average concentration of BOD.

Figure 7.22 shows absolute and percentage change in the 95-percentile depth-average concentration of TSS.



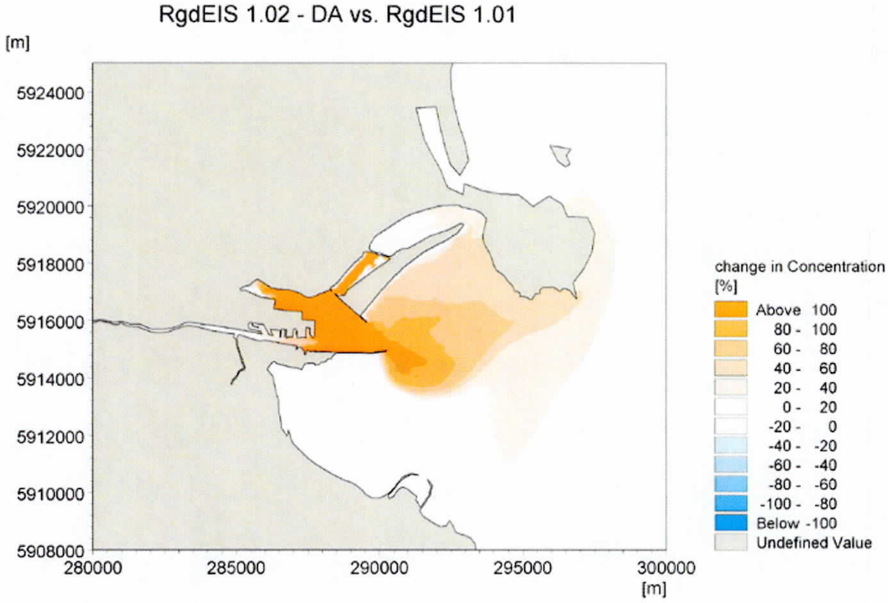
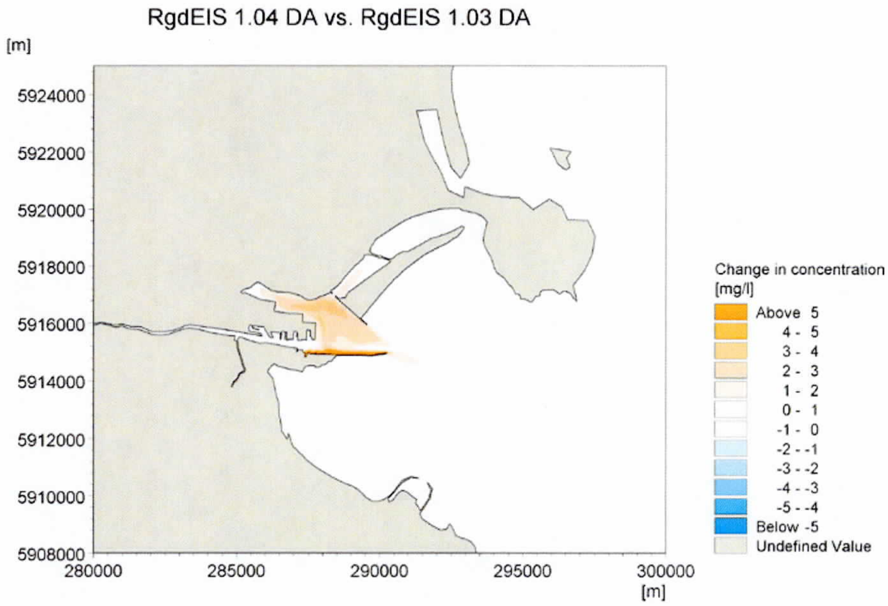


Figure 7.21 Construction impact. Difference in concentration of BOD based on water quality model scenario 1.02 – existing environment, peak discharge, and 1.01 – existing environment, average conditions. Upper panel: absolute difference in concentration [mg/l, 95%ile, depth-average]. Lower panel: percentage change [95%ile, depth-average]



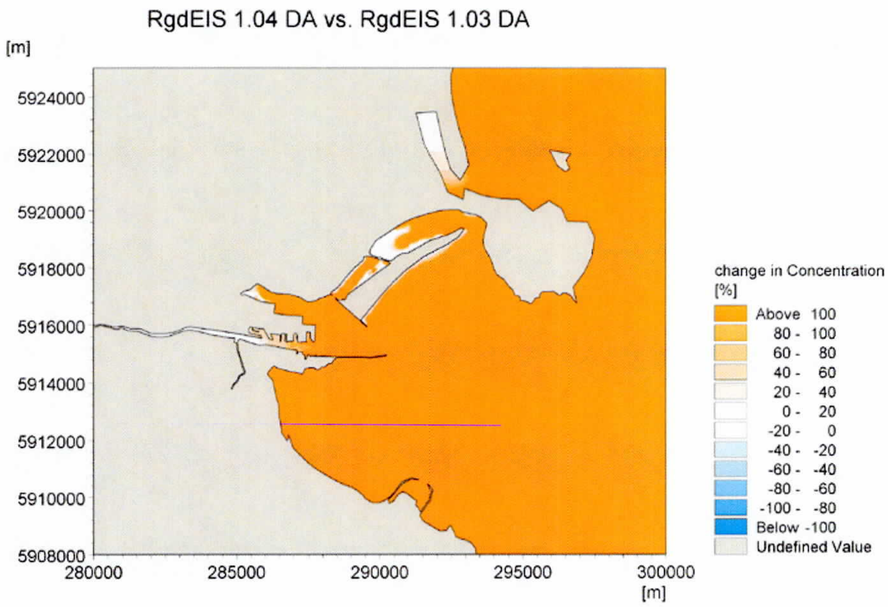


Figure 7.22 Construction impact. Difference in concentration of TSS based on water quality model scenario 1.04 – existing environment, peak discharge, and 1.03 – existing environment, average conditions. Upper panel: absolute difference in concentration [mg/l, 95%ile, depth-average]. Lower panel: percentage change [95%ile, depth-average]

7.2.3 Risk Assessment

The risk assessment considers the effects on the water quality environment of a 3-day continuous discharge from the Ringsend WwTP with no background concentrations. This was simulated by water quality mode scenario 6.18 (see Table 6.2). The pollutant selected for this scenario was BOD with a concentration of 240 mg/l (untreated) and the release coincided with start of spring-tide conditions in Dublin.

Figure 7.23 shows snapshots of the instantaneous concentration of BOD in the Dublin Bay and its estuaries every 6-hours during the 3-day continuous discharge.

Figure 7.24 shows snapshots of the instantaneous concentration of BOD in Dublin Bay and its estuaries after the end of the 3-day continuous discharge. It can be observed that 24-hours after the spill only area in the upper Tolka Estuary and behind Bull Island show elevated levels of BOD. All the BOD has dispersed/decayed 66-hours after the end of the spill.

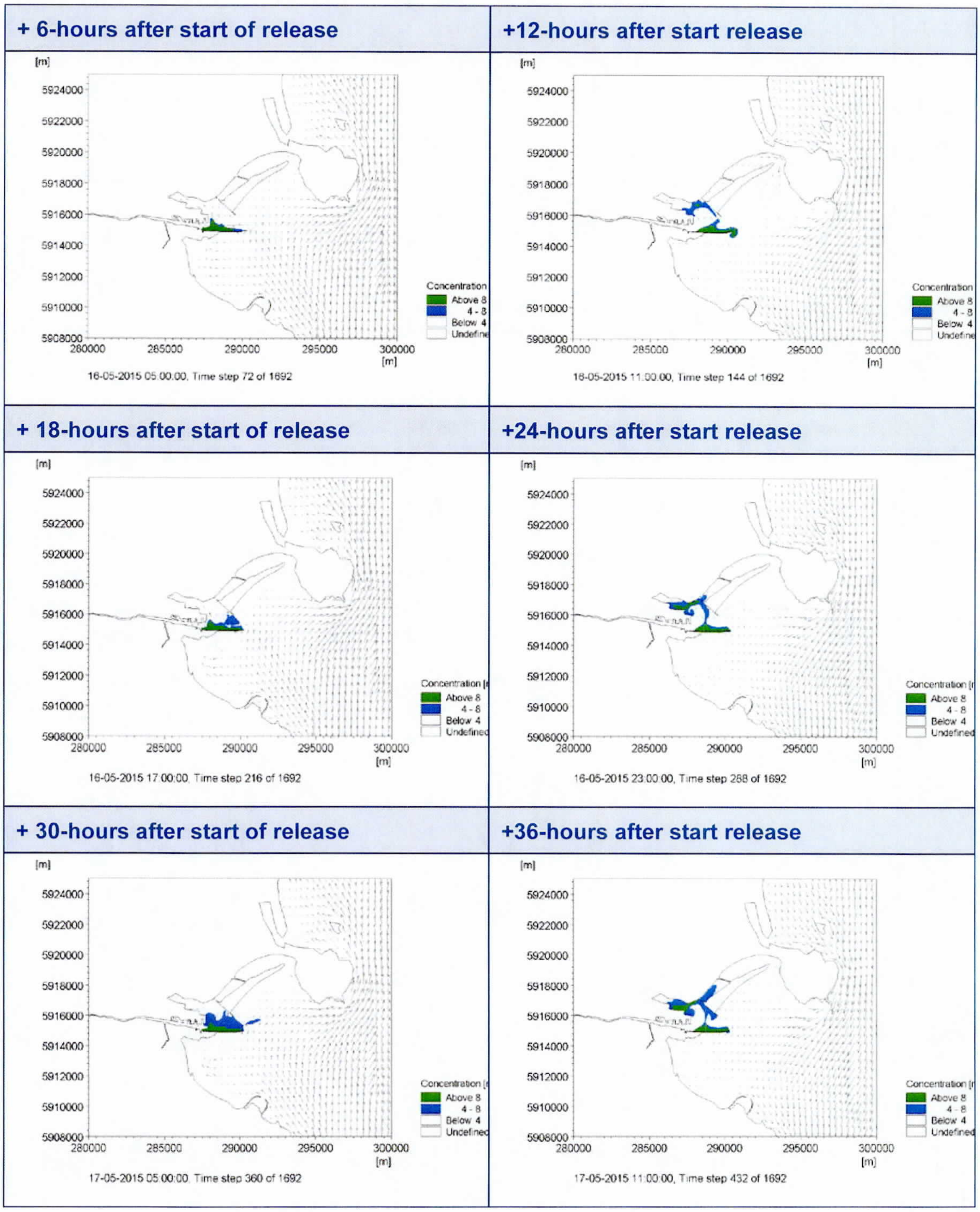


Figure 7.23 Snapshot of concentration of BOD [mg/l, surface] every 6-hours during a 72-hour period - water quality model scenario 6.18, BOD – 3 Day Untreated Discharge). Vectors show the magnitude and direction of the depth-average current velocity.

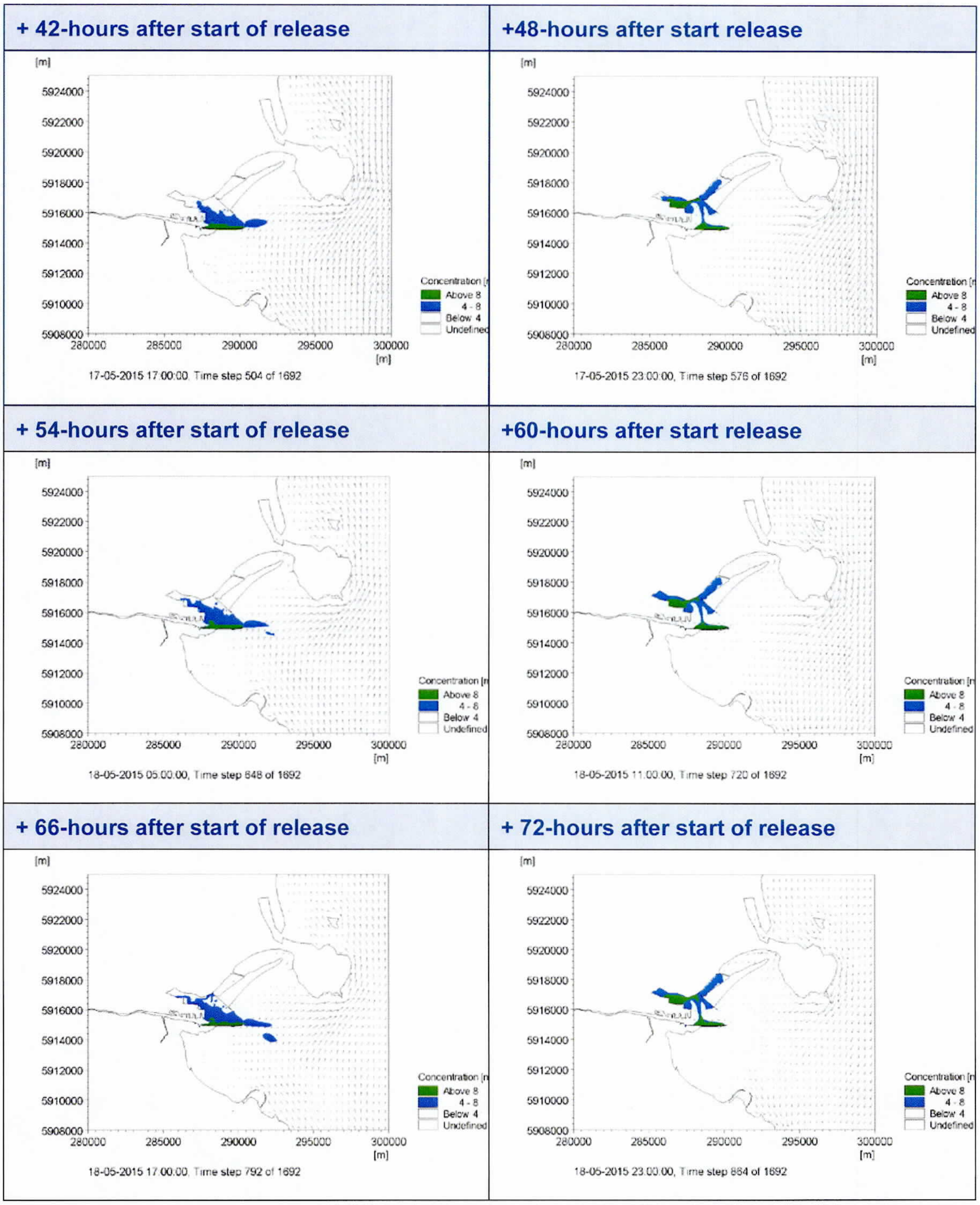


Figure 7.34 (continued) Snapshot of concentration of BOD [mg/l, surface] every 6-hours during a 72-hour period - water quality model scenario 6.18, BOD – 3 Day Untreated Discharge). Vectors show the magnitude and direction of the depth-average current velocity.

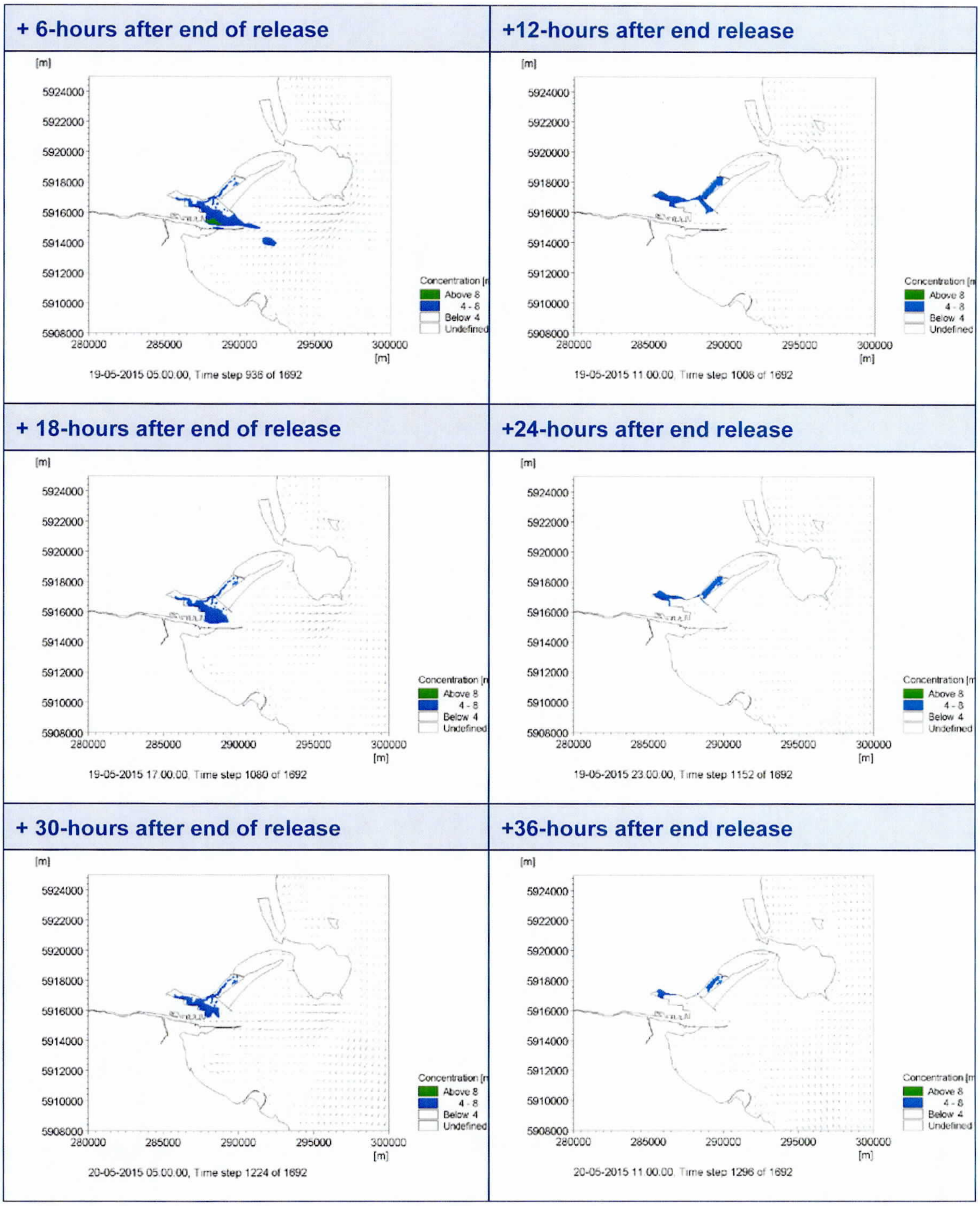
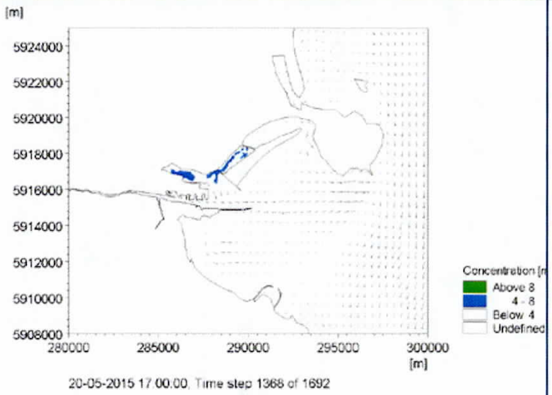
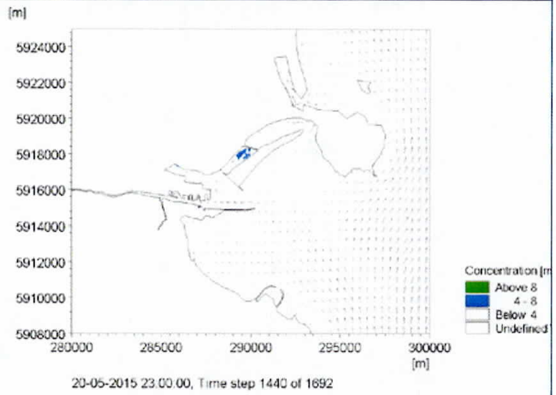


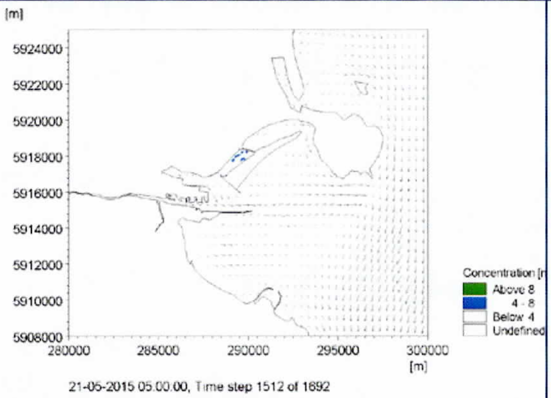
Figure 7.24 Snapshot of concentration of BOD [mg/l, surface] every 6-hours after end of 72-hour period - water quality model scenario 6.18, BOD – 3 Day Untreated Discharge). Vectors show the magnitude and direction of the depth-average current velocity.

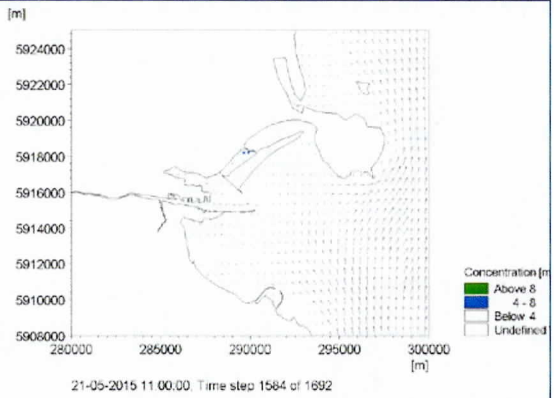
+ 42-hours after end of release	+48-hours after end release





+ 54-hours after end of release	+60-hours after end release





+ 66-hours after end of release

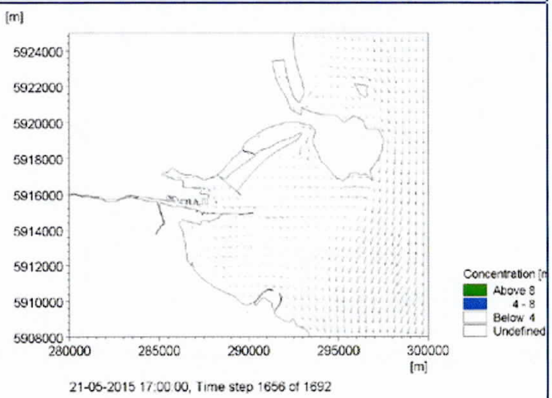


Figure 7.35 (continued) Snapshot of concentration of BOD [mg/l, surface] every 6-hours after end of 72-hour period - water quality model scenario 6.18, BOD – 3 Day Untreated Discharge). Vectors show the magnitude and direction of the depth-average current velocity.

7.2.4 Cumulative Impacts

Cumulative impacts seek to investigate the effects of other future infrastructure changes that may impact on the water quality environment of Dublin Bay and its estuaries. These include the following:

- Scenario 11.07 – Future Discharge – Average Conditions with Poolbeg Power Station running (see Section 6.3.1.4);
- Scenario 11.07 – Future Discharge – Repair of the ESB Cooling Water Channel (see Section 6.3.2); and
- Scenario 11.07 – Future Discharge – Alexandra Basin Redevelopment Scheme (see Section 6.3.2).

The fate of substances released from the Ringsend WwTP were modelled as a conservative tracer; passive, non-decaying particles. The trajectories of particles released from the above scenario are plotted alongside the particles from scenario 6.09 (Future discharge – average conditions) to understand the cumulative impact effects.

For these runs, six passive particles are released at the top of every hour for 24 hours on four (4) separate days throughout the spring-neap tidal cycle. The release days reflect a range of tidal and conditions (see Table 7.3).

The particles do not decay but are only tracked for 48 hours (2 days) after the time of release. Horizontal and vertical diffusion are included (the dispersion describes the transport due to molecular diffusion and due to non-resolved turbulence or eddies).

Table 7.3 Tidal stage during particle release days for the cumulative impacts assessment.

Day	Tide Stage
3	Intermediate (near-spring)
5	Spring
11	Neap
13	Intermediate (neap-neap)

7.2.4.1 Poolbeg Power Station

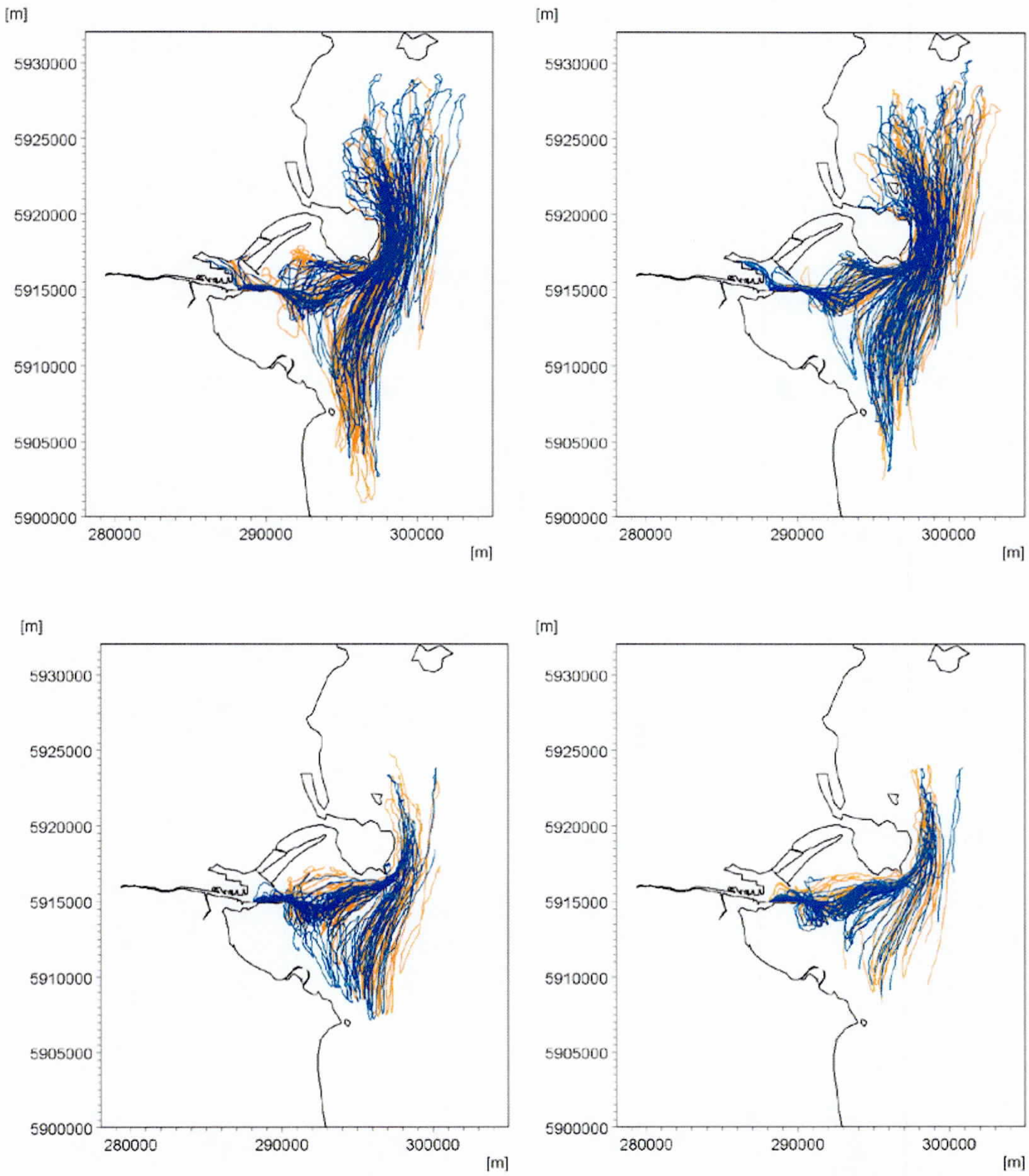


Figure 7.25 Conservative tracer particles released to surface waters at Ringsend WwTP outfall on Lower Liffey. Particles tracks show position over a period of 48-hours from time of release. Blue tracks show particles from water quality model run 11.07 – Future discharge environment with Poolbeg Power Station On. Orange tracks show water quality model run 6.09 – Future discharge, average conditions. The four plots show particles released during day 3 (upper panel, left), day 5 (upper panel, right), day 11 (lower panel, left) and day 13 (lower panel, right).

7.2.4.2 ESB Cooling Water Channel Repair

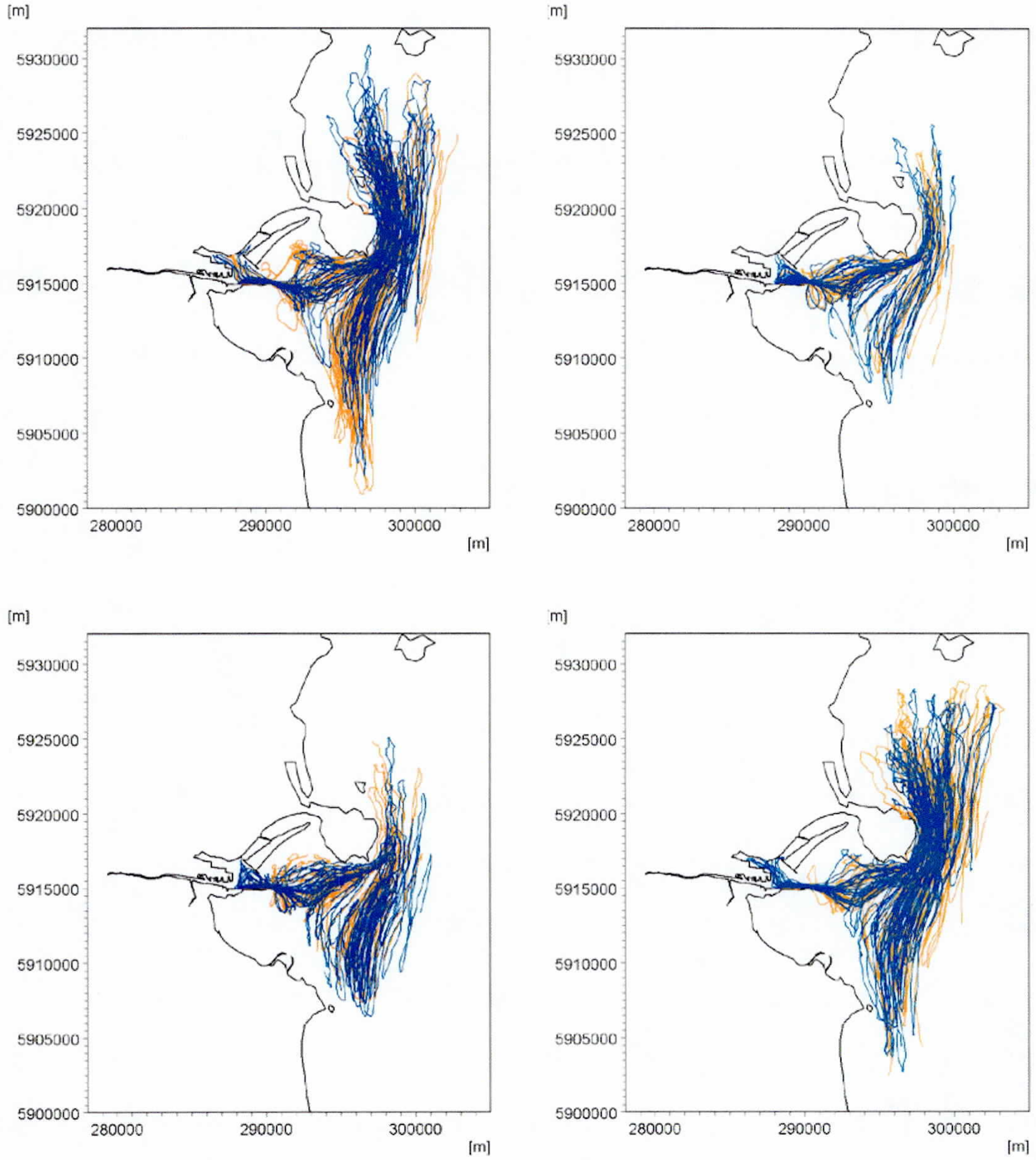


Figure 7.26 Conservative tracer particles released to surface waters at Ringsend WwTP outfall on Lower Liffey. Particles tracks show position over a period of 48-hours from time of release. Blue tracks show particles from water quality model run 14.01 – Future discharge environment, with ESB cooling water channel repaired. Orange tracks show water quality model run 6.09 – Future discharge environment, average conditions. The four plots show particles released during day 3 (upper panel, left), day 5 (upper panel, right), day 11 (lower panel, left) and day 13 (lower panel, right).

7.2.4.3 Alexandra Basin Redevelopment

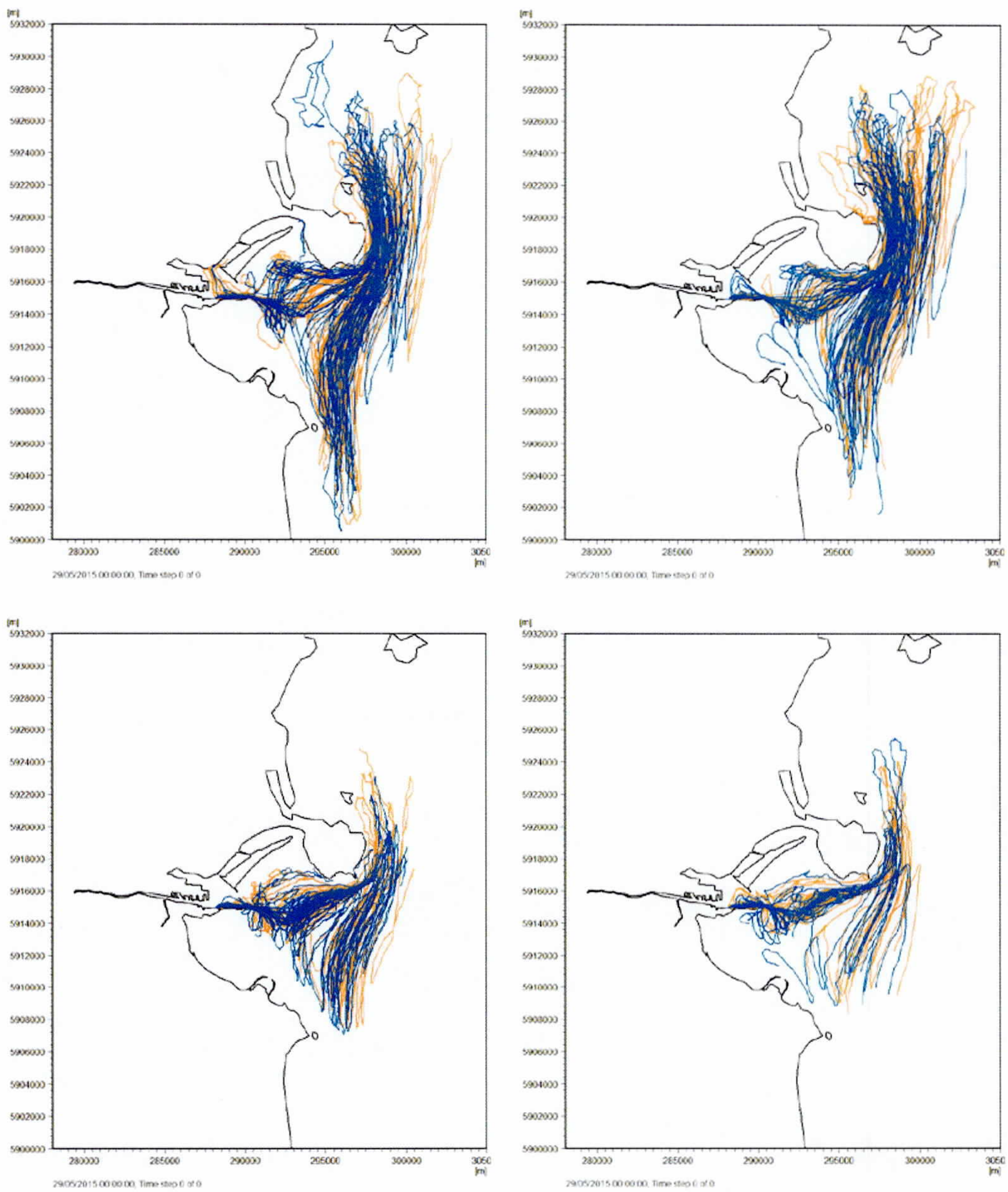


Figure 7.27 Conservative tracer particles released to surface waters at Ringsend WwTP outfall on Lower Liffey. Particles tracks show position over a period of 48-hours from time of release. Blue tracks show particles from water quality model run 15.01 – Future discharge environment with Alexandra Basin Redeveloped. Orange tracks show water quality model run 6.09 – Future discharge, average conditions. The four plots show particles released during day 3 (upper panel, left), day 5 (upper panel, right), day 11 (lower panel, left) and day 13 (lower panel, right).

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8 Conclusions

This report details the investigations undertaken to assess the potential changes to the water environment due to the proposed alterations to the Ringsend WwTP.

8.1 Changes to the Hydrodynamic Conditions

As the principal control on the results of the modelling, the impact of the hydrodynamics is critical to the representativeness of the tested water quality scenarios. As detailed in the calibration stage of the study, the model generally showed a good comparison to the measured data in Dublin Bay and the Lower Liffey Estuary.

This study has shown that overall, tidal currents are relatively weak in the Liffey and the Tolka, with the ability for freshwater flow and other discharges to the estuaries to either dominate or play an important part in the dynamics. This is visibly evidenced around the Ringsend outfall by frontal features delineating fresher/saltier water at various stages of the tide as shown in Figure 8.1. It is also noted from this figure that the model captures this variability.

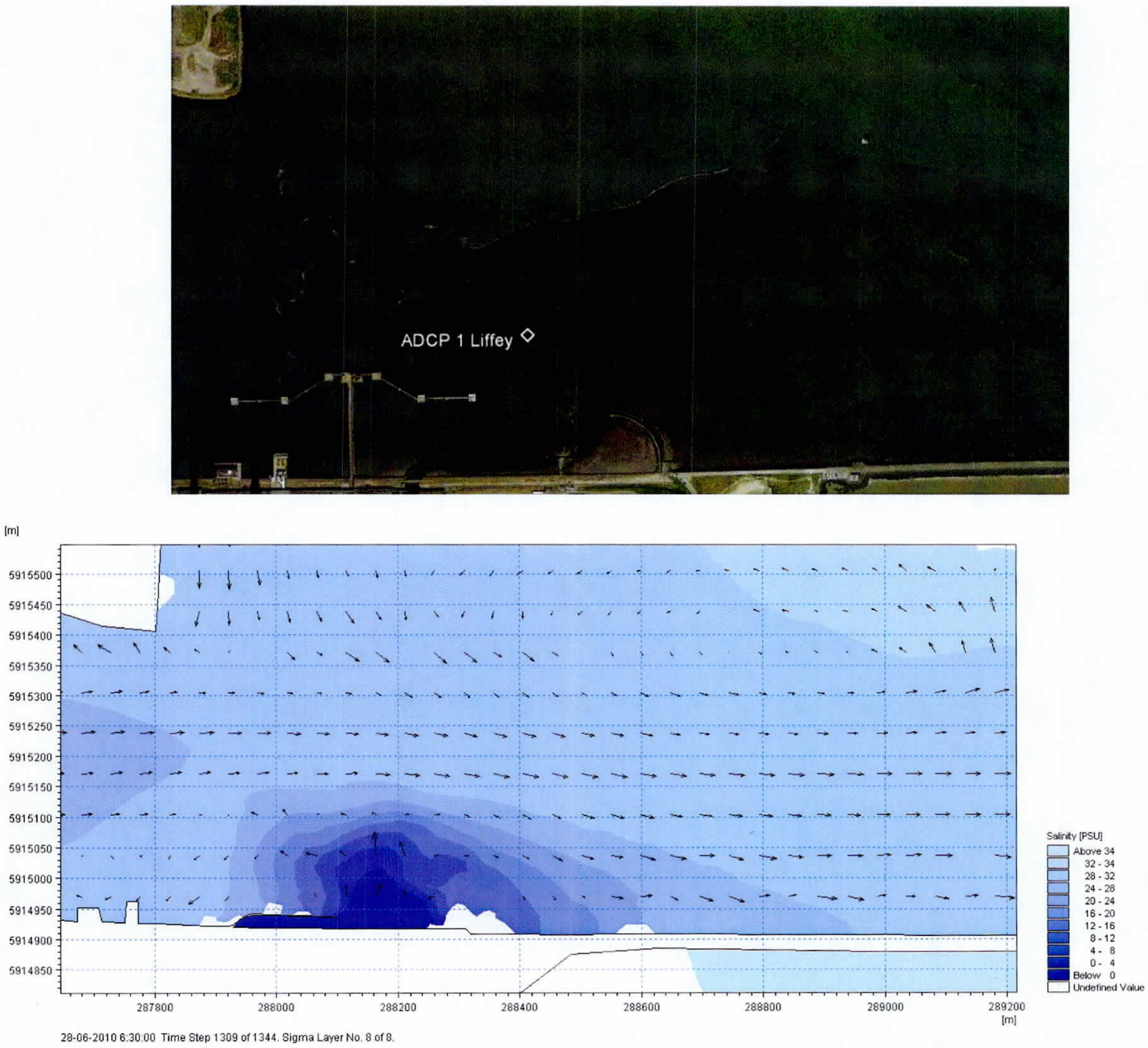

Figure 8.1 Example of surface frontal features observed in the Lower Liffey mixing fresh water from a range of sources (top panel) and the model representation of salinity for an example timestep (bottom panel).

Therefore, when considering the 3D structure of the water column of the Lower Liffey it is noted that there is a high degree of complexity, with a pronounced salinity stratification passing the Ringsend outfall on the rise and fall of the tide.

The difference between surface and at depth flow magnitude and directions is noticeable in the area around Ringsend, with the possibility for the two to be opposed at particular states of the tide. For the interaction with the plume from Ringsend, it is noted that most of the interaction will be at the surface, as the freshwater from the WwTP will typically be less dense than the surrounding estuary.

In addition, the location at the confluence of the Liffey and the Tolka leads to additional complexity. The wider mouth to the Tolka leading to surface flow tending to pass Ringsend and enter the Tolka on the flood tide, rather than flow up the Liffey.

Under the flood flow conditions tested in the storm scenarios, it has been seen that the combined flow of the rivers can dominate the lower estuary with freshwater flows.

Comparing the pre- and post-scheme changes in hydrodynamics, the dominant change is that caused by implementing the proposed repairs to the sheetpiles and weir in the ESB outfall channel. In its current dilapidated state, it can be seen that flow exits in the direction of the Liffey on the flood tide and remains constrained towards the South Bull Wall typically. Post remediation, the flow over the easterly end of the weir leads to a slight change in the position of the surface water flows, which is sufficient to lead to a small increase in water from the vicinity or Ringsend into the lower Tolka.

8.2 Changes to the Water Quality Conditions

With respect to water quality, the model results show that there can be seen to be a very slight increase in BOD in the lower Tolka. However, it is considered likely that the model changes seen will be below the level of measured detectability for BOD and appears a significant change with respect to the % difference and not the 95%ile values. It is noted that in the future scenario BOD coming from Ringsend will be half of the existing situation. It is considered that the primary reason for this difference is due to the changes in how the flood tide operates with the repaired weir structure at the ESB outfall.

For TSS, it is apparent that overall there is an improvement in the future as the levels coming from Ringsend will reduce. In addition, it is noted that limited background information was available to this study for TSS. Therefore it is considered likely that the background concentrations due to wave stirring and from rivers is likely to be greater than that seen in the model tests.

Ammonia and MRP can be seen to be an improvement in most locations following the WwTP upgrade. Of note for Ammonia is that within the Bay/Coastal waters it can be seen to be below the EQS (for river and lake environments) in the future scenario.

The results for DIN illustrate a slight worsening in summer, which again appears at odds with the reduction in DIN planned for the new WwTP. The principal control on this, is again considered to be the upgrade to the sheetpiles around the ESB outfall, leading to the changes. It is noted that the status of the estuary for DIN is generally on the threshold between poor and good. However the EQS for DIN used herein is prescribed for Coastal Waters, not Transitional Waters.

The outputs for E. coli are specific to the storm events and show a general reduction, primarily due to the improved control of the storm water. It is important to note that none of the bathing water monitoring locations were seen to be negatively impacted by the proposed changes, with the results highlighting that the existing failures at beaches is likely to be due to the localised outfalls in the immediate proximity of the bathing water beaches.

8.3 Remaining Uncertainties

The detailed modelling study undertaken here is considered robust for the EIAR. However the process has highlighted areas of residual uncertainty.

- Further representation of the flows in the rivers – gauging of the rivers is undertaken at some distance from the areas of impact and the large number of unmonitored freshwater flows could influence the dynamics of the estuary.
- As part of any future design studies for the ESB outfall, there could be a need for further consideration of the potential impact of any alterations made to the structure.

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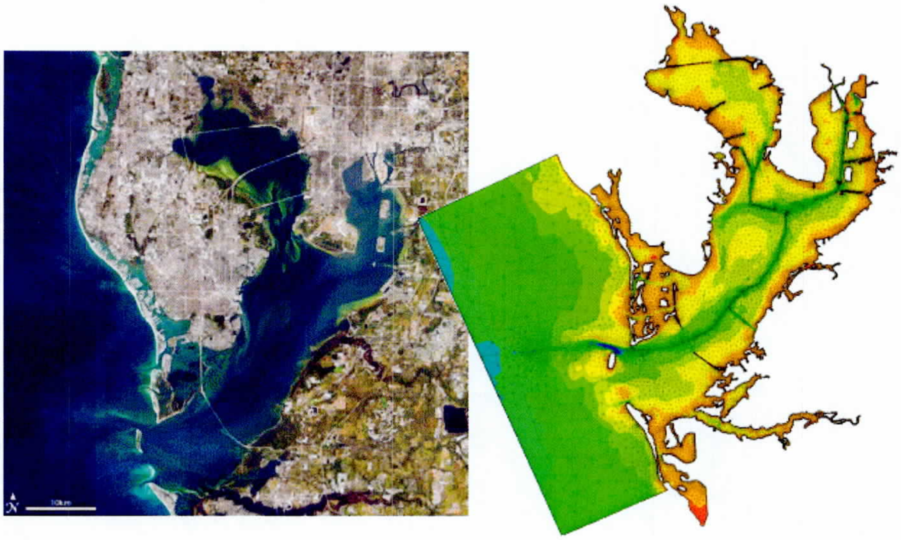
APPENDIX A – MIKE 3 FM, Short Description

MIKE
Powered by DHI



MIKE 21 & MIKE 3 Flow Model FM

Hydrodynamic Module

Short Description

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MIKE213_HD_FM_Short_Description_.docx / AJS/EBR/NHP / 2015-11-18

MIKE 21 & MIKE 3 Flow Model FM

The Flow Model FM is a comprehensive modelling system for two- and three-dimensional water modelling developed by DHI. The 2D and 3D models carry the same names as the classic DHI model versions MIKE 21 & MIKE 3 with an 'FM' added referring to the type of model grid - Flexible Mesh.

The modelling system has been developed for complex applications within oceanographic, coastal and estuarine environments. However, being a general modelling system for 2D and 3D free-surface flows it may also be applied for studies of inland surface waters, e.g. overland flooding and lakes or reservoirs.

MIKE 21 & MIKE 3 Flow Model FM is a general hydrodynamic flow modelling system based on a finite volume method on an unstructured mesh

The Modules of the Flexible Mesh Series

DHI's Flexible Mesh (FM) series includes the following modules:

Flow Model FM modules

- Hydrodynamic Module, HD
- Transport Module, TR
- Ecology Module, ECO Lab
- Oil Spill Module, ELOS
- Sand Transport Module, ST
- Mud Transport Module, MT
- Particle Tracking Module, PT

Wave module

- Spectral Wave Module, SW

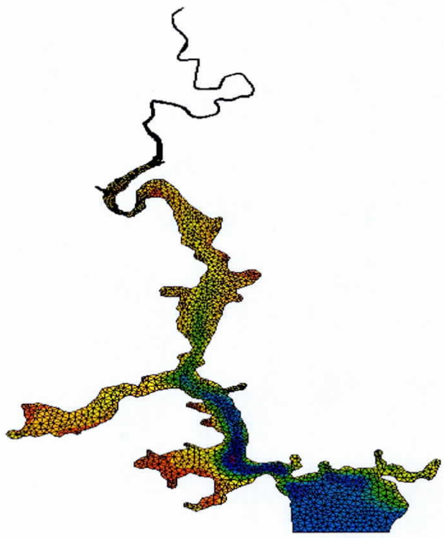
The FM Series meets the increasing demand for realistic representations of nature, both with regard to 'look alike' and to its capability to model coupled processes, e.g. coupling between currents, waves and sediments. Coupling of modules is managed in the Coupled Model FM.

All modules are supported by advanced user interfaces including efficient and sophisticated tools for mesh generation, data management, 2D/3D visualization, etc. In combination with comprehensive documentation and support, the FM series forms a unique professional software tool for consultancy services related to design, operation and maintenance tasks within the marine environment.

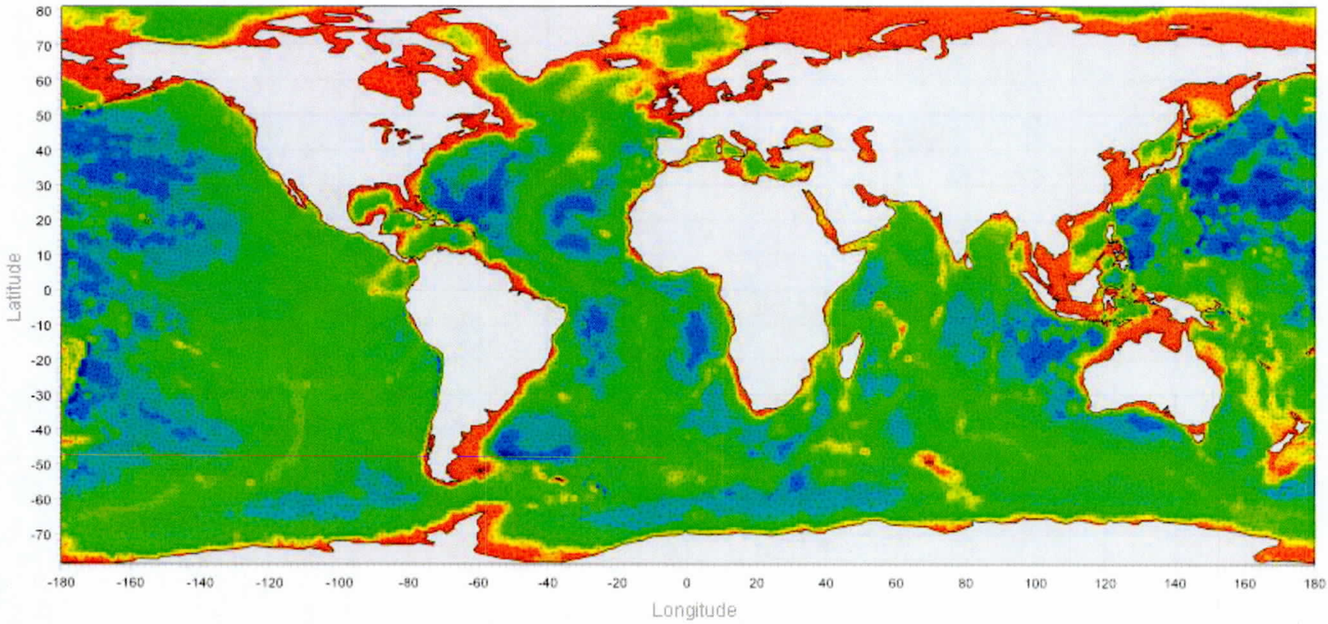
An unstructured grid provides an optimal degree of flexibility in the representation of complex geometries and enables smooth representations of boundaries. Small elements may be used in areas where more detail is desired, and larger elements used where less detail is needed, optimising information for a given amount of computational time.

The spatial discretisation of the governing equations is performed using a cell-centred finite volume method. In the horizontal plane an unstructured grid is used while a structured mesh is used in the vertical domain (3D).

This document provides a short description of the Hydrodynamic Module included in MIKE 21 & MIKE 3 Flow Model FM.



Example of computational mesh for Tamar Estuary, UK



MIKE 21 & MIKE 3 FLOW MODEL FM supports both Cartesian and spherical coordinates. Spherical coordinates are usually applied for regional and global sea circulation applications. The chart shows the computational mesh and bathymetry for the planet Earth generated by the MIKE Zero Mesh Generator

MIKE 21 & MIKE 3 Flow Model FM - Hydrodynamic Module

The Hydrodynamic Module provides the basis for computations performed in many other modules, but can also be used alone. It simulates the water level variations and flows in response to a variety of forcing functions on flood plains, in lakes, estuaries and coastal areas.

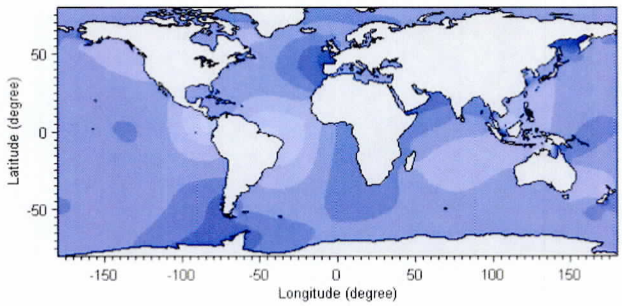
Application Areas

The Hydrodynamic Module included in MIKE 21 & MIKE 3 Flow Model FM simulates unsteady flow taking into account density variations, bathymetry and external forcings.

The choice between 2D and 3D model depends on a number of factors. For example, in shallow waters, wind and tidal current are often sufficient to keep the water column well-mixed, i.e. homogeneous in salinity and temperature. In such cases a 2D model can be used. In water bodies with stratification, either by density or by species (ecology), a 3D model should be used. This is also the case for enclosed or semi-enclosed waters where wind-driven circulation occurs.

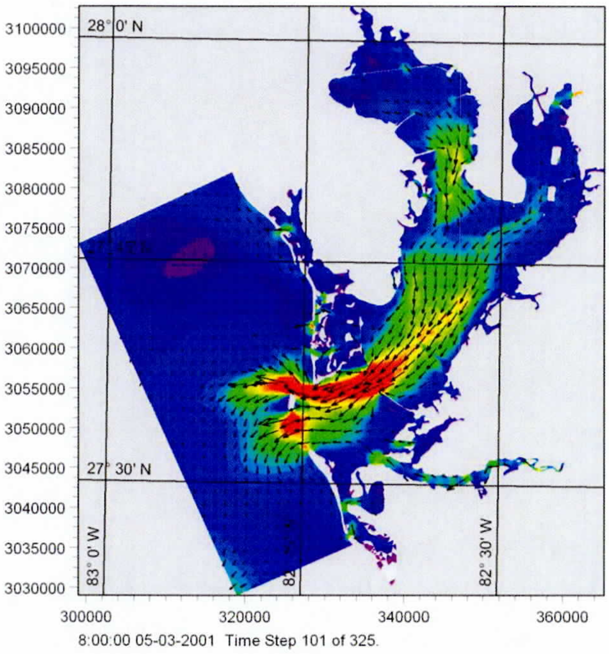
Typical application areas are

- Assessment of hydrographic conditions for design, construction and operation of structures and plants in stratified and non-stratified waters
- Environmental impact assessment studies
- Coastal and oceanographic circulation studies
- Optimization of port and coastal protection infrastructures
- Lake and reservoir hydrodynamics
- Cooling water, recirculation and desalination
- Coastal flooding and storm surge
- Inland flooding and overland flow modelling
- Forecast and warning systems

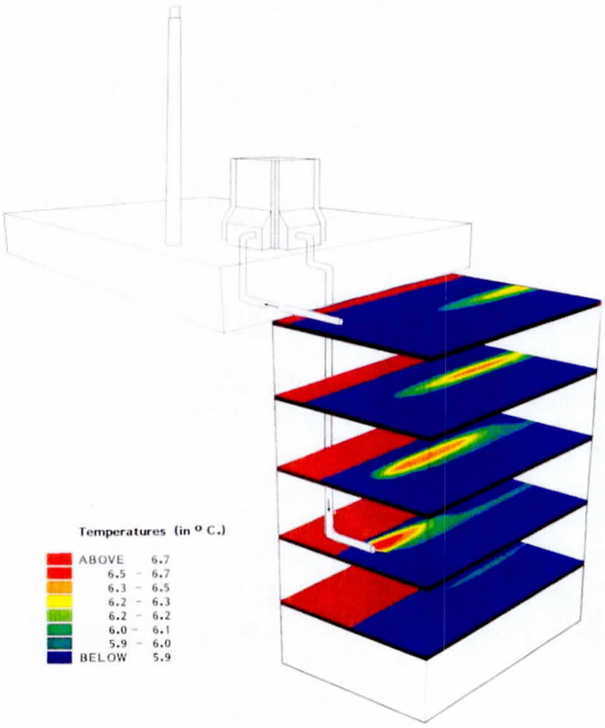


Example of a global tide application of MIKE 21 Flow Model FM. Results from such a model can be used as boundary conditions for regional scale forecast or hindcast models

The MIKE 21 & MIKE 3 Flow Model FM also support spherical coordinates, which makes both models particularly applicable for global and regional sea scale applications.



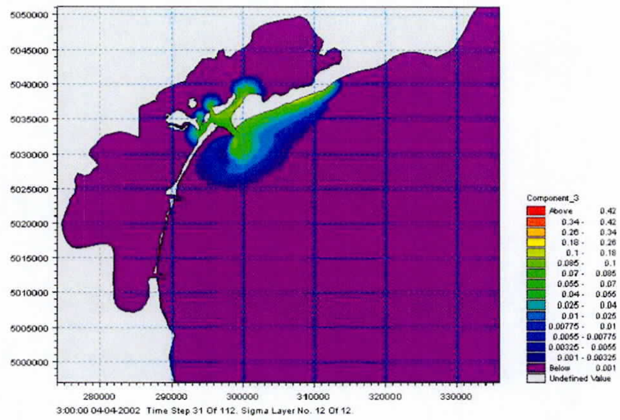
Example of a flow field in Tampa Bay, FL, simulated by MIKE 21 Flow Model FM



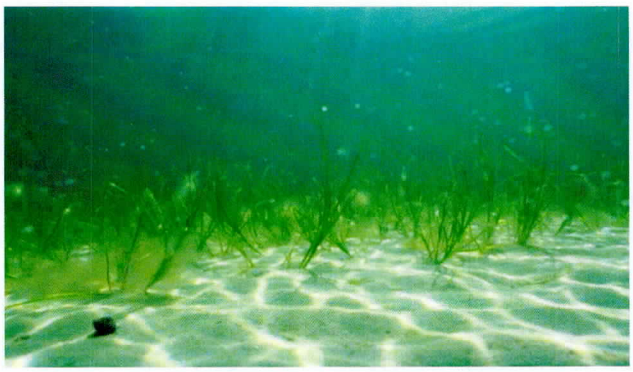
Study of thermal recirculation

Typical applications with the MIKE 21 & MIKE 3 Flow Model FM include cooling water recirculation and ecological impact assessment (eutrophication)

The Hydrodynamic Module is together with the Transport Module (TR) used to simulate the spreading and fate of dissolved and suspended substances. This module combination is applied in tracer simulations, flushing and simple water quality studies.



Tracer simulation of single component from outlet in the Adriatic, simulated by MIKE 21 Flow Model FM HD+TR

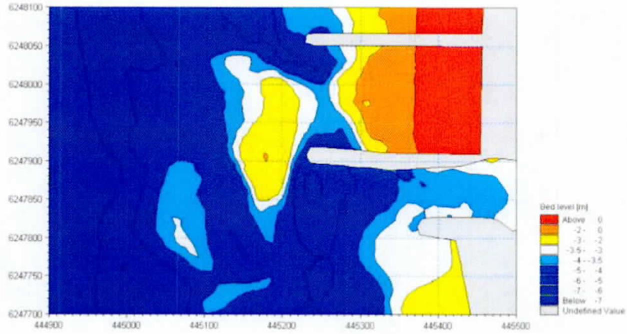


Prediction of ecosystem behaviour using the MIKE 21 & MIKE 3 Flow Model FM together with ECO Lab

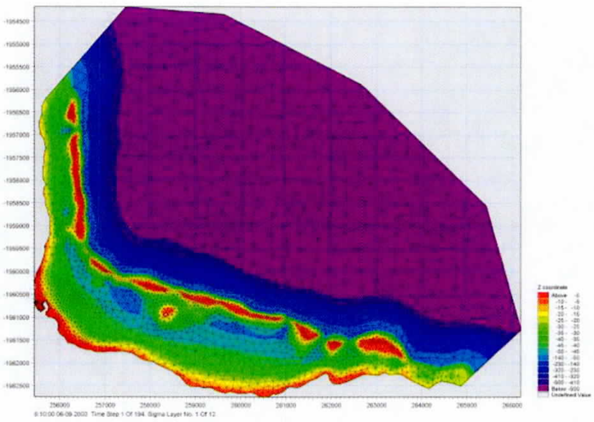
The Hydrodynamic Module can be coupled to the Ecological Module (ECO Lab) to form the basis for environmental water quality studies comprising multiple components.

Furthermore, the Hydrodynamic Module can be coupled to sediment models for the calculation of sediment transport. The Sand Transport Module and Mud Transport Module can be applied to simulate transport of non-cohesive and cohesive sediments, respectively.

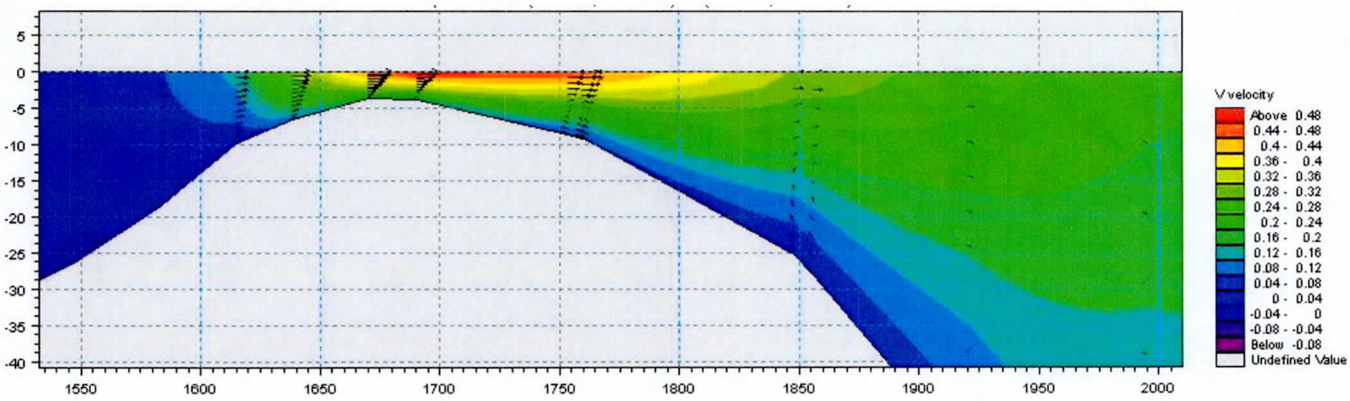
In the coastal zone the transport is mainly determined by wave conditions and associated wave-induced currents. The wave-induced currents are generated by the gradients in radiation stresses that occur in the surf zone. The Spectral Wave Module can be used to calculate the wave conditions and associated radiation stresses.



Coastal application (morphology) with coupled MIKE 21 HD, SW and ST, Torsminde harbour Denmark



Model bathymetry of Taravao Bay, Tahiti



Example of Cross reef currents in Taravao Bay, Tahiti simulated with MIKE 3 Flow Model FM. The circulation and renewal of water inside the reef is dependent on the tides, the meteorological conditions and the cross reef currents, thus the circulation model includes the effects of wave induced cross reef currents

Computational Features

The main features and effects included in simulations with the MIKE 21 & MIKE 3 Flow Model FM – Hydrodynamic Module are the following:

- Flooding and drying
- Momentum dispersion
- Bottom shear stress
- Coriolis force
- Wind shear stress
- Barometric pressure gradients
- Ice coverage
- Tidal potential
- Precipitation/evaporation
- Wave radiation stresses
- Sources and sinks

Model Equations

The modelling system is based on the numerical solution of the two/three-dimensional incompressible Reynolds averaged Navier-Stokes equations subject to the assumptions of Boussinesq and of hydrostatic pressure. Thus, the model consists of continuity, momentum, temperature, salinity and density equations and it is closed by a turbulent closure scheme. The density does not depend on the pressure, but only on the temperature and the salinity.

For the 3D model, the free surface is taken into account using a sigma-coordinate transformation approach or using a combination of a sigma and z-level coordinate system.

Below the governing equations are presented using Cartesian coordinates.

The local continuity equation is written as

$$\frac{\partial u}{\partial x}+\frac{\partial v}{\partial y}+\frac{\partial w}{\partial z}=S$$

and the two horizontal momentum equations for the x- and y-component, respectively

$$\frac{\partial u}{\partial t}+\frac{\partial u^2}{\partial x}+\frac{\partial vu}{\partial y}+\frac{\partial wu}{\partial z}=fv-g\frac{\partial \eta}{\partial x}-\frac{1}{\rho_0}\frac{\partial p_a}{\partial x}-\frac{g}{\rho_0}\int_z^{\eta}\frac{\partial \rho}{\partial x}dz+F_u+\frac{\partial}{\partial z}\left(\nu_t\frac{\partial u}{\partial z}\right)+u_sS$$

$$\frac{\partial v}{\partial t}+\frac{\partial v^2}{\partial y}+\frac{\partial uv}{\partial x}+\frac{\partial wv}{\partial z}=-fu-g\frac{\partial \eta}{\partial y}-\frac{1}{\rho_0}\frac{\partial p_a}{\partial y}-\frac{g}{\rho_0}\int_z^{\eta}\frac{\partial \rho}{\partial y}dz+F_v+\frac{\partial}{\partial z}\left(\nu_t\frac{\partial v}{\partial z}\right)+v_sS$$

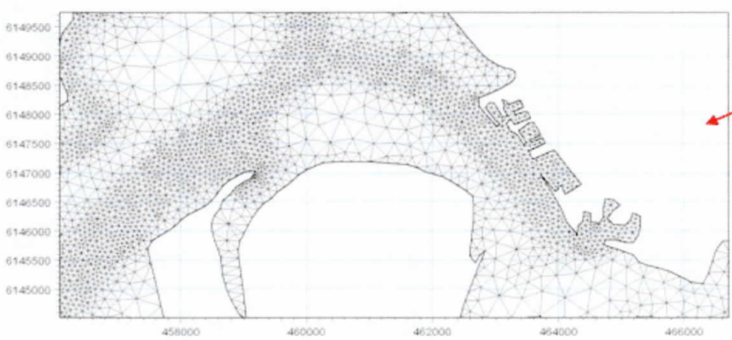
Temperature and salinity

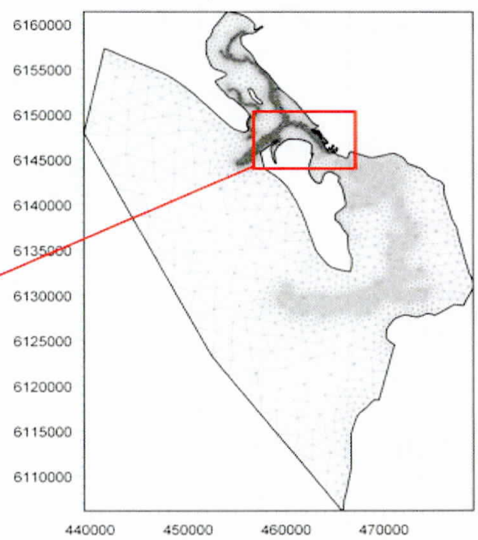
In the Hydrodynamic Module, calculations of the transports of temperature, *T*, and salinity, *s* follow the general transport-diffusion equations as

$$\frac{\partial T}{\partial t}+\frac{\partial uT}{\partial x}+\frac{\partial vT}{\partial y}+\frac{\partial wT}{\partial z}=F_T+\frac{\partial}{\partial z}\left(D_v\frac{\partial T}{\partial z}\right)+\hat{H}+T_sS$$

$$\frac{\partial s}{\partial t}+\frac{\partial us}{\partial x}+\frac{\partial vs}{\partial y}+\frac{\partial ws}{\partial z}=F_s+\frac{\partial}{\partial z}\left(D_v\frac{\partial s}{\partial z}\right)+s_sS$$

Unstructured mesh technique gives the maximum degree of flexibility, for example: 1) Control of node distribution allows for optimal usage of nodes 2) Adoption of mesh resolution to the relevant physical scales 3) Depth-adaptive and boundary-fitted mesh. Below is shown an example from Ho Bay Denmark with the approach channel to the Port of Esbjerg





The horizontal diffusion terms are defined by

$$(F_T, F_s) = \left[\frac{\partial}{\partial x}\left(D_h \frac{\partial}{\partial x} \right) + \frac{\partial}{\partial y}\left(D_h \frac{\partial}{\partial y} \right) \right](T, s)$$

The equations for two-dimensional flow are obtained by integration of the equations over depth.

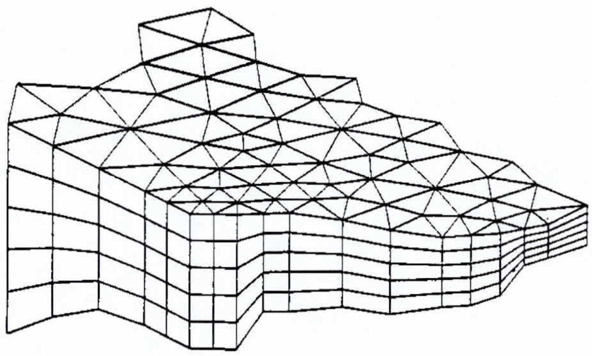
Heat exchange with the atmosphere is also included.

Symbol list

t	time
x, y, z	Cartesian coordinates
u, v, w	flow velocity components
T, s	temperature and salinity
D_v	vertical turbulent (eddy) diffusion coefficient
$\hat{H}$	source term due to heat exchange with atmosphere
S	magnitude of discharge due to point sources
T_s, s_s	temperature and salinity of source
F_T, F_s, F_c	horizontal diffusion terms
D_h	horizontal diffusion coefficient
h	depth

Solution Technique

The spatial discretisation of the primitive equations is performed using a cell-centred finite volume method. The spatial domain is discretised by subdivision of the continuum into non-overlapping elements/cells.



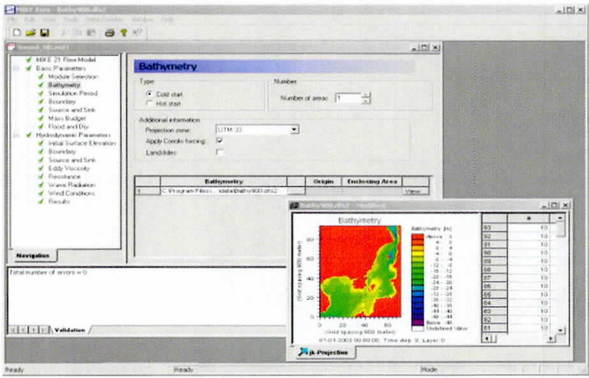
Principle of 3D mesh

In the horizontal plane an unstructured mesh is used while a structured mesh is used in the vertical domain of the 3D model. In the 2D model the elements can be triangles or quadrilateral elements. In the 3D model the elements can be prisms or bricks whose horizontal faces are triangles and quadrilateral elements, respectively.

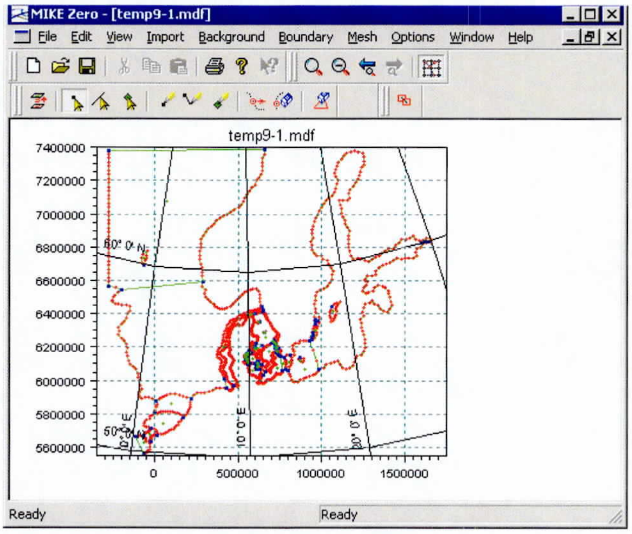
Model Input

Input data can be divided into the following groups:

- Domain and time parameters:
  - computational mesh (the coordinate type is defined in the computational mesh file) and bathymetry
  - simulation length and overall time step
- Calibration factors
  - bed resistance
  - momentum dispersion coefficients
  - wind friction factors
- Initial conditions
  - water surface level
  - velocity components
- Boundary conditions
  - closed
  - water level
  - discharge
- Other driving forces
  - wind speed and direction
  - tide
  - source/sink discharge
  - wave radiation stresses



View button on all the GUIs in MIKE 21 & MIKE 3 FM HD for graphical view of input and output files

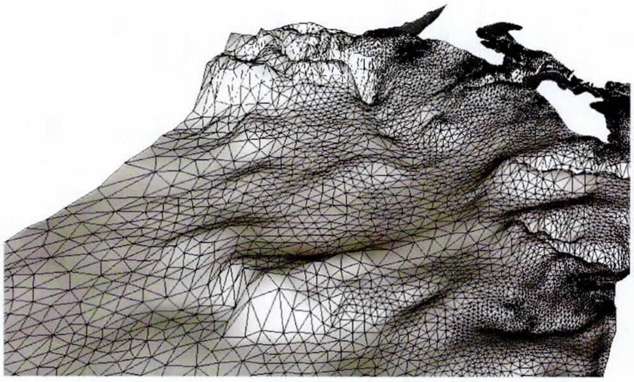


The Mesh Generator is an efficient MIKE Zero tool for the generation and handling of unstructured meshes, including the definition and editing of boundaries

Providing MIKE 21 & MIKE 3 Flow Model FM with a suitable mesh is essential for obtaining reliable results from the models. Setting up the mesh includes the appropriate selection of the area to be modelled, adequate resolution of the bathymetry, flow, wind and wave fields under consideration and definition of codes for defining boundaries.

2D visualization of a computational mesh (Odense Estuary)

Bathymetric values for the mesh generation can e.g. be obtained from the MIKE by DHI product MIKE C-Map. MIKE C-Map is an efficient tool for extracting depth data and predicted tidal elevation from the world-wide Electronic Chart Database CM-93 Edition 3.0 from Jeppesen Norway.

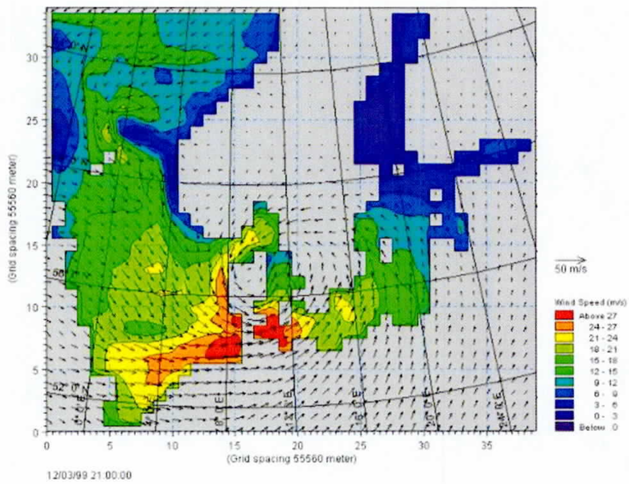


3D visualization of a computational mesh

If wind data is not available from an atmospheric meteorological model, the wind fields (e.g. cyclones) can be determined by using the wind-generating programs available in MIKE 21 Toolbox.

Global winds (pressure & wind data) can be downloaded for immediate use in your simulation. The sources of data are from GFS courtesy of NCEP, NOAA. By specifying the location, orientation and grid dimensions, the data is returned to you in the correct format as a spatial varying grid series or a time series. The link is:

http://waterdata.dhigroup.com/octopus/home



The chart shows a hindcast wind field in the North Sea and Baltic Sea as wind speed and wind direction

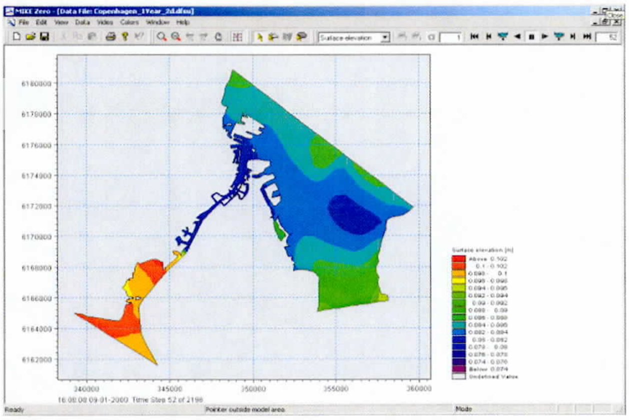
Model Output

Computed output results at each mesh element and for each time step consist of:

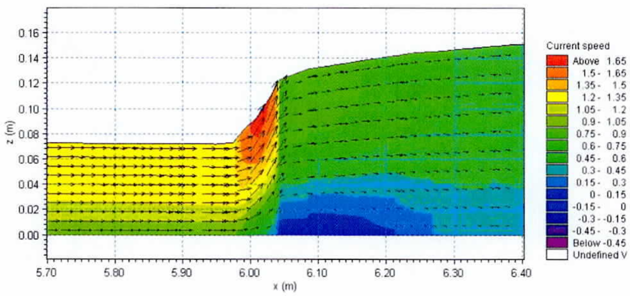
- Basic variables
  - water depth and surface elevation
  - flux densities in main directions
  - velocities in main directions
  - densities, temperatures and salinities
- Additional variables
  - Current speed and direction
  - Wind velocities
  - Air pressure
  - Drag coefficient
  - Precipitation/evaporation
  - Courant/CFL number
  - Eddy viscosity
  - Element area/volume

The output results can be saved in defined points, lines and areas. In the case of 3D calculations the results are saved in a selection of layers.

Output from MIKE 21 & MIKE 3 Flow Model FM is typically post-processed using the Data Viewer available in the common MIKE Zero shell. The Data Viewer is a tool for analysis and visualization of unstructured data, e.g. to view meshes, spectra, bathymetries, results files of different format with graphical extraction of time series and line series from plan view and import of graphical overlays.



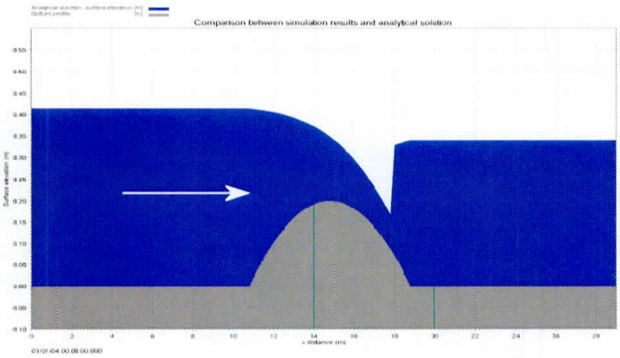
The Data Viewer in MIKE Zero – an efficient tool for analysis and visualization of unstructured data including processing of animations. Above screen dump shows surface elevations from a model setup covering Port of Copenhagen



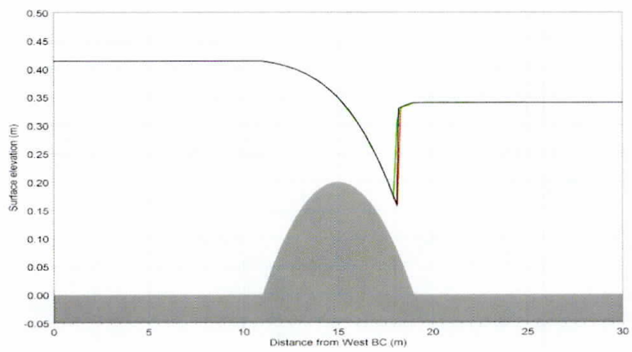
Vector and contour plot of current speed at a vertical profile defined along a line in Data Viewer in MIKE Zero

Validation

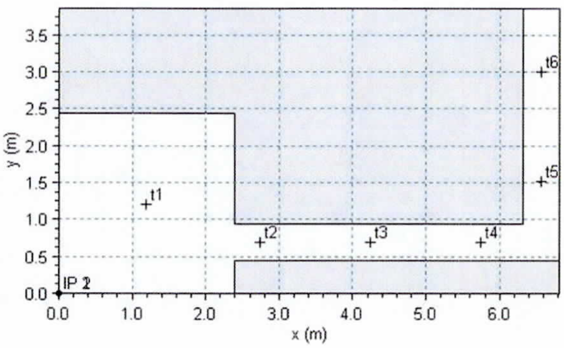
Prior to the first release of MIKE 21 & MIKE 3 Flow Model FM the model has successfully been applied to a number of rather basic idealized situations for which the results can be compared with analytical solutions or information from the literature.



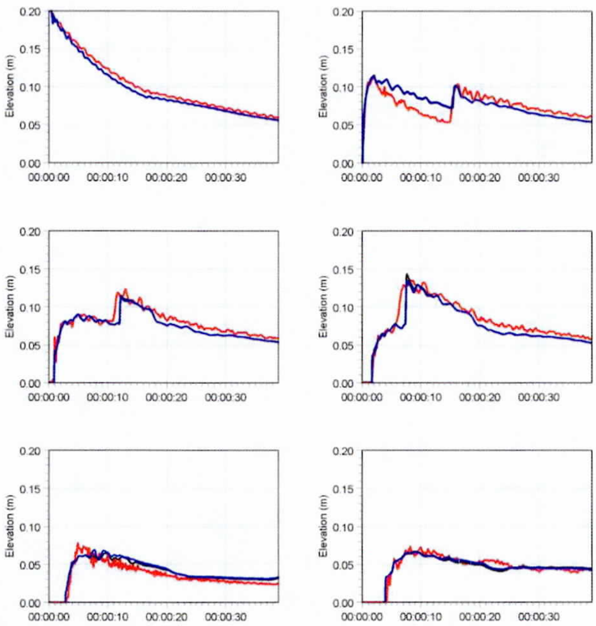
The domain is a channel with a parabola-shaped bump in the middle. The upstream (western) boundary is a constant flux and the downstream (eastern) boundary is a constant elevation. Below: the total depths for the stationary hydraulic jump at convergence. Red line: 2D setup, green line: 3D setup, black line: analytical solution



A dam-break flow in an L-shaped channel (a, b, c):

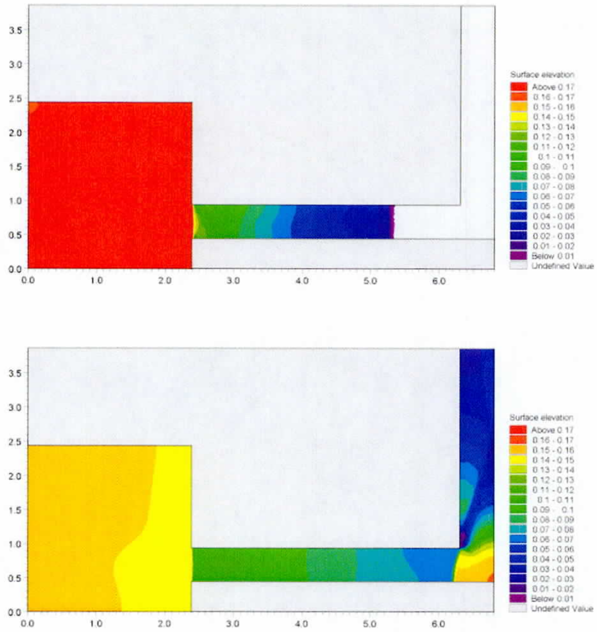


a) Outline of model setup showing the location of gauging points



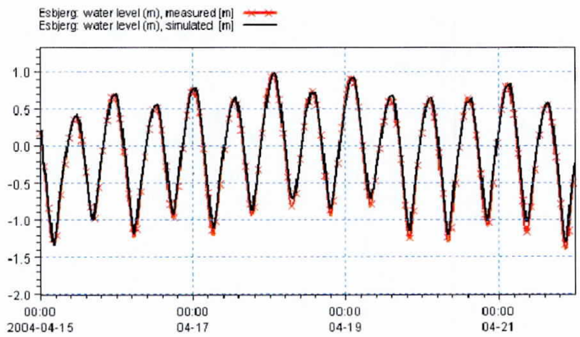
b) Comparison between simulated and measured water levels at the six gauge locations.
(Blue) coarse mesh (black) fine mesh and (red) measurements

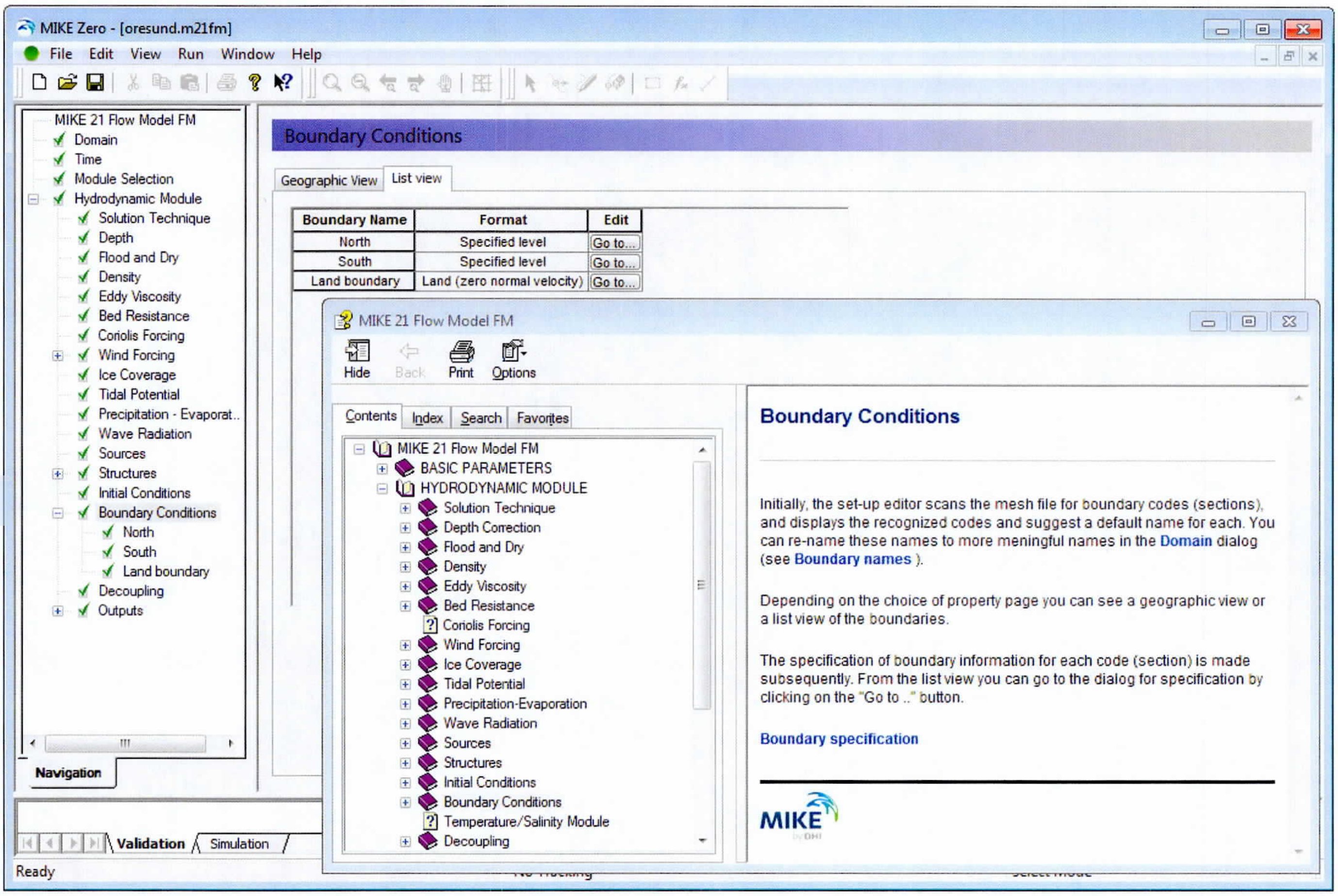
The model has also been applied and tested in numerous natural geophysical conditions; ocean scale, inner shelves, estuaries, lakes and overland, which are more realistic and complicated than academic and laboratory tests.



c) Contour plots of the surface elevation at T = 1.6 s (top) and T = 4.8 s (bottom)

Example from Ho Bay, a tidal estuary (barrier island coast) in South-West Denmark with access channel to the Port of Esbjerg. Below: Comparison between measured and simulated water levels



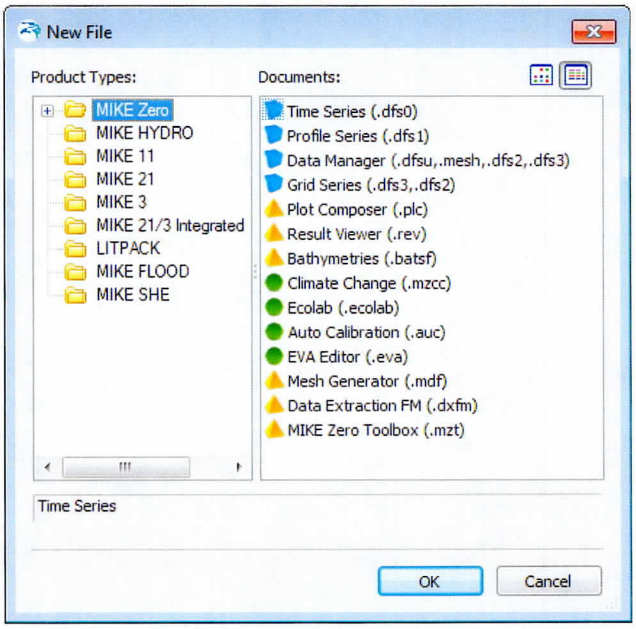


The user interface of the MIKE 21 and MIKE 3 Flow Model FM (Hydrodynamic Module), including an example of the extensive Online Help system

aphical User Interface

The MIKE 21 & MIKE 3 Flow Model FM Hydrodynamic Module is operated through a fully Windows integrated graphical user interface (GUI). Support is provided at each stage by an Online Help system.

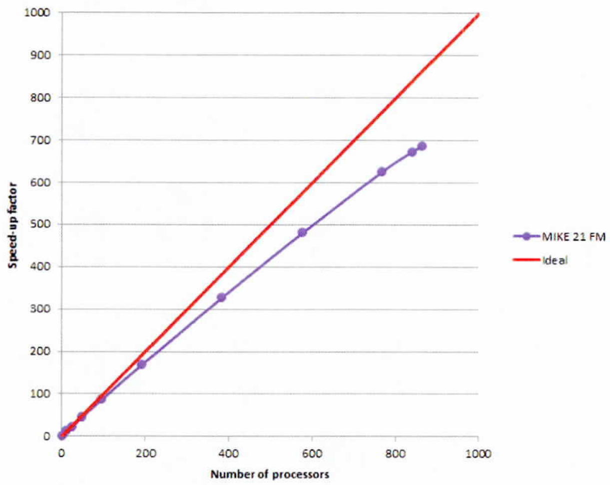
The common MIKE Zero shell provides entries for common data file editors, plotting facilities and utilities such as the Mesh Generator and Data Viewer.



Overview of the common MIKE Zero utilities

Parallelisation

The computational engines of the MIKE 21/3 FM series are available in versions that have been parallelised using both shared memory (OpenMP) as well as distributed memory architecture (MPI). The result is much faster simulations on systems with many cores.



MIKE 21 FM speed-up using a HPC Cluster with distributed memory architecture (purple)

Hardware and Operating System Requirements

The MIKE 21 and MIKE 3 Flow Model FM Hydrodynamic Module supports Microsoft Windows 7 Professional Service Pack 1 (32 and 64 bit), Windows 8.1 Pro (64 bit), Windows 10 Pro (64 bit) and Windows Server 2012 R2 Standard (64 bit). Microsoft Internet Explorer 9.0 (or higher) is required for network license management as well as for accessing the Online Help.

The recommended minimum hardware requirements for executing the MIKE 21 and MIKE 3 Flow Model FM Hydrodynamic Module are:

Processor:	3 GHz PC (or higher)
Memory (RAM):	4 GB (or higher)
Hard disk:	160 GB (or higher)
Monitor:	SVGA, resolution 1024x768
Graphic card:	64 MB RAM (256 MB RAM or higher is recommended)

Support

News about new features, applications, papers, updates, patches, etc. are available here:

www.mikepoweredbydhi.com/Download/DocumentsAndTools.aspx

For further information on MIKE 21 and MIKE 3 Flow Model FM software , please contact your local DHI office or the support centre:

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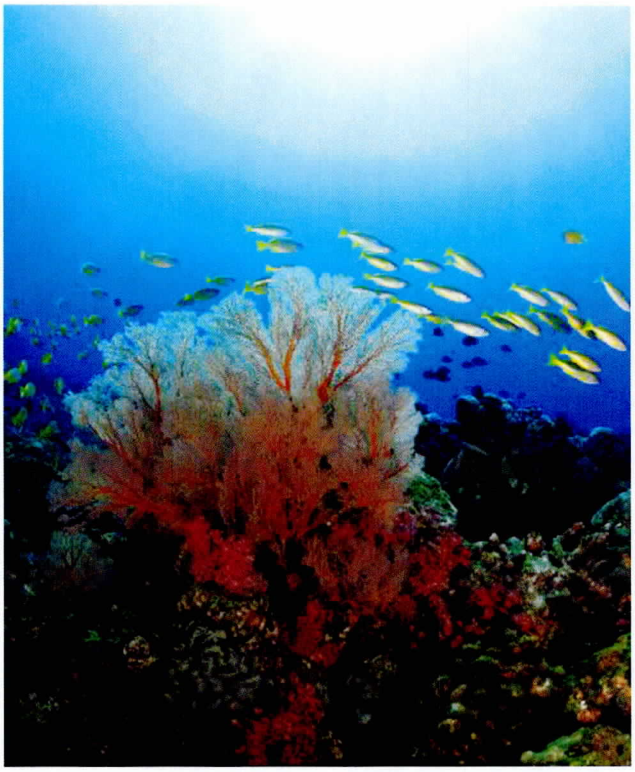
www.mikepoweredbydhi.com

Documentation

The MIKE 21 & MIKE 3 Flow Model FM models are provided with comprehensive user guides, online help, scientific documentation, application examples and step-by-step training examples.



References

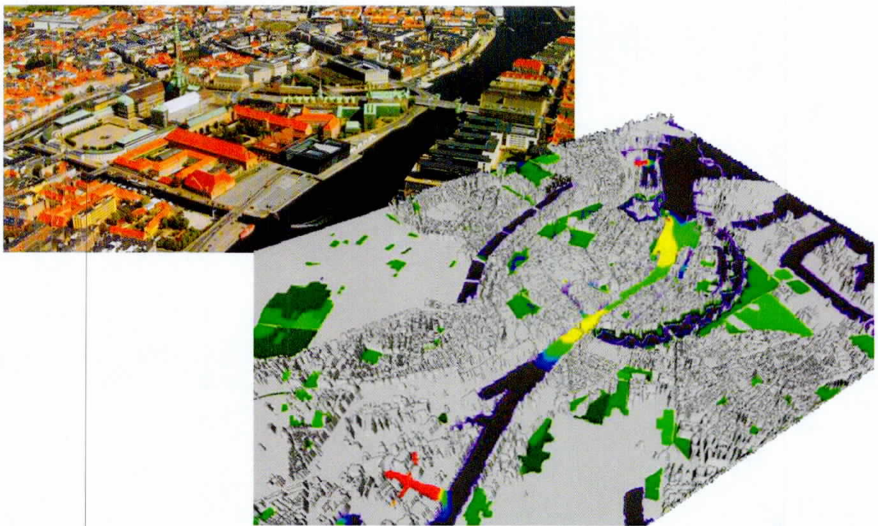
Petersen, N.H., Rasch, P. "Modelling of the Asian Tsunami off the Coast of Northern Sumatra", presented at the 3rd Asia-Pacific DHI Software Conference in Kuala Lumpur, Malaysia, 21-22 February, 2005

French, B. and Kerper, D. Salinity Control as a Mitigation Strategy for Habitat Improvement of Impacted Estuaries. 7th Annual EPA Wetlands Workshop, NJ, USA 2004.

DHI Note, "Flood Plain Modelling using unstructured Finite Volume Technique" January 2004 – download from

http://www.theacademybydhi.com/research-and-publications/scientific-publications

APPENDIX B– Transport Model, Short Description



MIKE 21 & MIKE 3 Flow Model FM

Transport Module

Short Description

The expert in **WATER ENVIRONMENTS**

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www.mikepoweredbydhi.com

MIKE213_TR_FM_Short_Description_.docx / AJS / 2015-11-20

MIKE 21 & MIKE 3 Flow Model FM - Transport Module

The Transport Module simulates the spreading and fate of dissolved or suspended substances in an aquatic environment under the influence of the fluid transport and associated dispersion processes. The substance may be of any kind, conservative or non-conservative, inorganic or organic. Non-conservative substances are distinguished by the manner in which they decay. Examples of linearly decaying substances are tracers that are absorbed to particulate matter.

The hydrodynamic basis for the Transport Module is calculated with the Hydrodynamic Module (HD). The hydrodynamic modules can be applied for both barotrophic (constant density) or baroclinic flows. In the latter case, the effect of variable density on the flow is included by solving the transport equations for salt and temperature. The viscosities or diffusivities in the hydrodynamic module are described either as simple constant or calculated using state-of-the-art turbulence models.

Application Areas

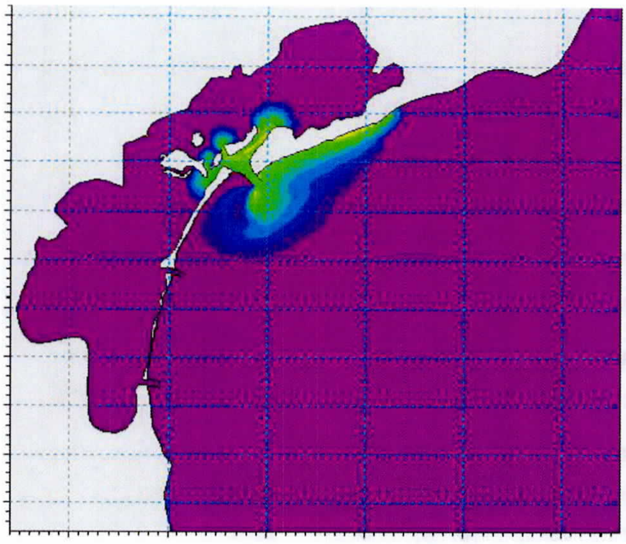
The Transport Module can be applied to a wide range of hydraulic and related phenomena. The application areas are generally problems where flow and transport phenomena are important with emphasis on coastal and marine applications, where the flexibility inherited in the unstructured meshes can be utilised.

Typical substances, which are modelled using the Transport Module are:

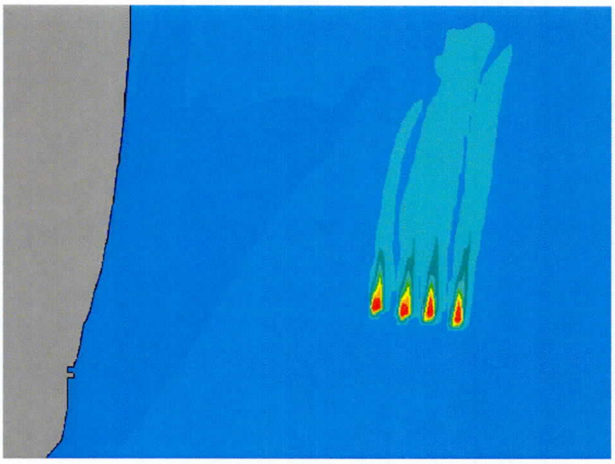
- Tracers
- Coliform bacteria
- Xenobiotic compounds

Typical applications include flushing studies, tracer simulations and simple water quality studies. In relation to point pollution sources the Transport Module can be used for conservative approximations of transport and dispersion of e-coli bacteria provided sufficient choice of decay coefficient.

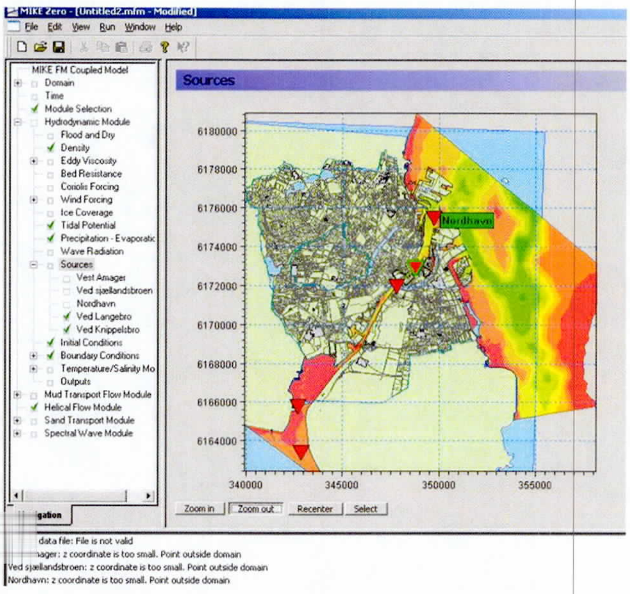
The Ecology and Water Quality Module (ECO Lab) is closely integrated with the Transport Module and the Hydrodynamic Module. ECO Lab simulates reaction processes in multi-compound systems or of substances with a more complex decay than linear, i.e. decay of substances that also depend on light intensity like e-coli. This enables complex ecosystem studies in coastal areas, estuaries and lakes.



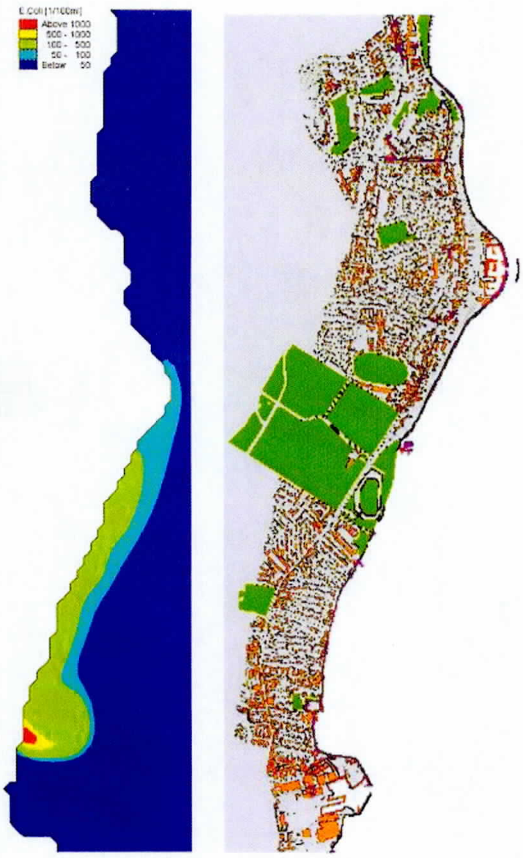
Typical applications with the MIKE 21 & MIKE 3 Flow Model FM Transport Module include tracer studies as shown above in the Venice lagoon



Example of plumes from outfall with colours indicating different concentrations



Example of user interface where sources from CSO's are specified to be used in model simulations to compare different abatement schemes, or online as input to forecasts of water quality



Example of bathing water quality forecasts from a municipality north of Copenhagen. The forecasts are made available on a dedicated bathing water quality webpage

Computational Features

The main features of MIKE 21 & MIKE 3 Flow Model FM – Transport Module are as follows:

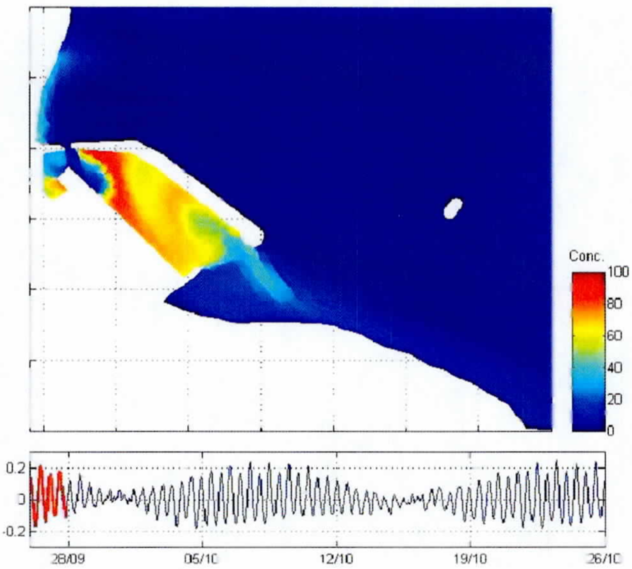
- Conservative substances
- Linear decay
- Sources and sinks (mass and momentum)

Model Equations

MIKE 21 & MIKE 3 Flow Model FM Transport Module is dynamically linked to the Hydrodynamic Module.

The modelling system is based on the numerical solution of the two/three-dimensional incompressible Reynolds averaged Navier-Stokes equations subject to the assumptions of Boussinesq and of hydrostatic pressure. Thus the model consists of continuity, momentum, temperature, salinity and density equations and it is closed by a turbulent closure scheme. The density does not depend on the pressure, but only on the temperature and the salinity.

For the 3D model, the free surface is taken into account using a sigma-coordinate transformation approach.



Flushing study example from a harbour on Tahiti.
Top: An initial concentration field is placed in the harbour and the dilution due to advection-dispersion processes are then simulated with the HD-TR modules.
Bottom: Time series of tidal elevations

Scalar quantity

The Transport Module can calculate the transport of a scalar quantity. The conservation equation for a scalar quantity is given by

$$\frac{\partial C}{\partial t}+\frac{\partial uC}{\partial x}+\frac{\partial vC}{\partial y}+\frac{\partial wC}{\partial z}=F_C+\frac{\partial}{\partial z}\left(D_v\frac{\partial C}{\partial z}\right)-k_pC+C_sS$$

The horizontal diffusion term is defined by

$$F_C=\left[\frac{\partial}{\partial x}\left(D_h\frac{\partial}{\partial x}\right)+\frac{\partial}{\partial y}\left(D_h\frac{\partial}{\partial y}\right)\right]C$$

For 2D calculations, the conservation equation is integrated over depth and defined by

$$\frac{\partial h\overline{C}}{\partial t}+\frac{\partial h\overline{u}\overline{C}}{\partial x}+\frac{\partial h\overline{v}\overline{C}}{\partial y}=hF_C-hk_p\overline{C}+hC_sS$$

Symbol list

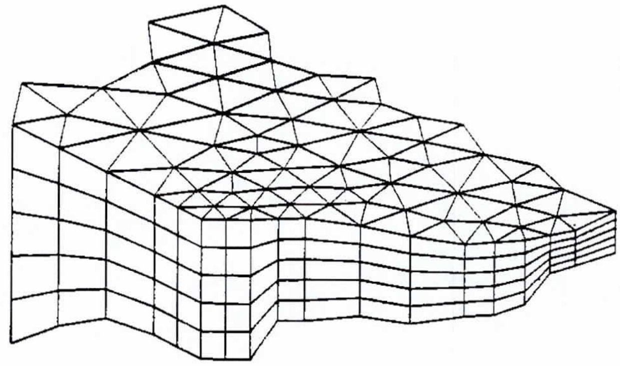
t	time
x, y, z	Cartesian coordinates
D_v	vertical turbulent (eddy) diffusion coefficient
S	magnitude of discharge due to point sources
F_c	horizontal diffusion term
D_h	horizontal diffusion coefficient
h	depth
$\overline{u}$, $\overline{v}$	depth-averaged velocity components
C	concentration of scalar quantity
k_p	linear decay rate of scalar quantity
C_S	concentration of scalar quantity in source

Solution Technique

The solution of the transport equations is closely linked to the solution of the hydrodynamic conditions.

The spatial discretization of the primitive equations is performed using a cell-centred finite volume method. The spatial domain is discretized by subdivision of the continuum into non-overlapping elements/cells. In the horizontal plane an unstructured mesh is used while in the vertical domain in the 3D model a structured mesh is used. In the 2D model the elements can be triangles or quadrilateral elements. In the 3D model the elements can be prisms or bricks whose horizontal faces are triangles and quadrilateral elements, respectively.

The time integration is performed using an explicit scheme.

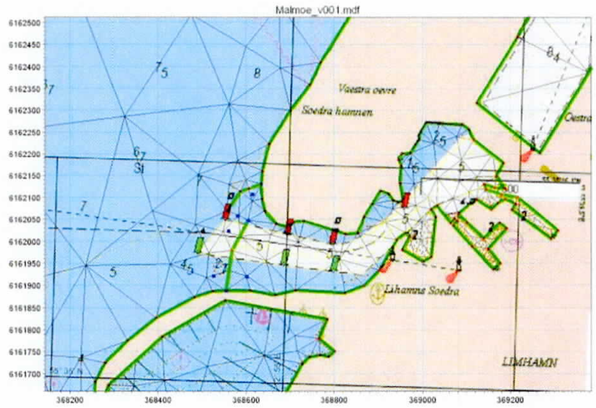


Principle of 3D mesh

Model Input Data

The necessary input data to the transport model is, besides the input for the hydrodynamic model alone, information about the components to simulate:

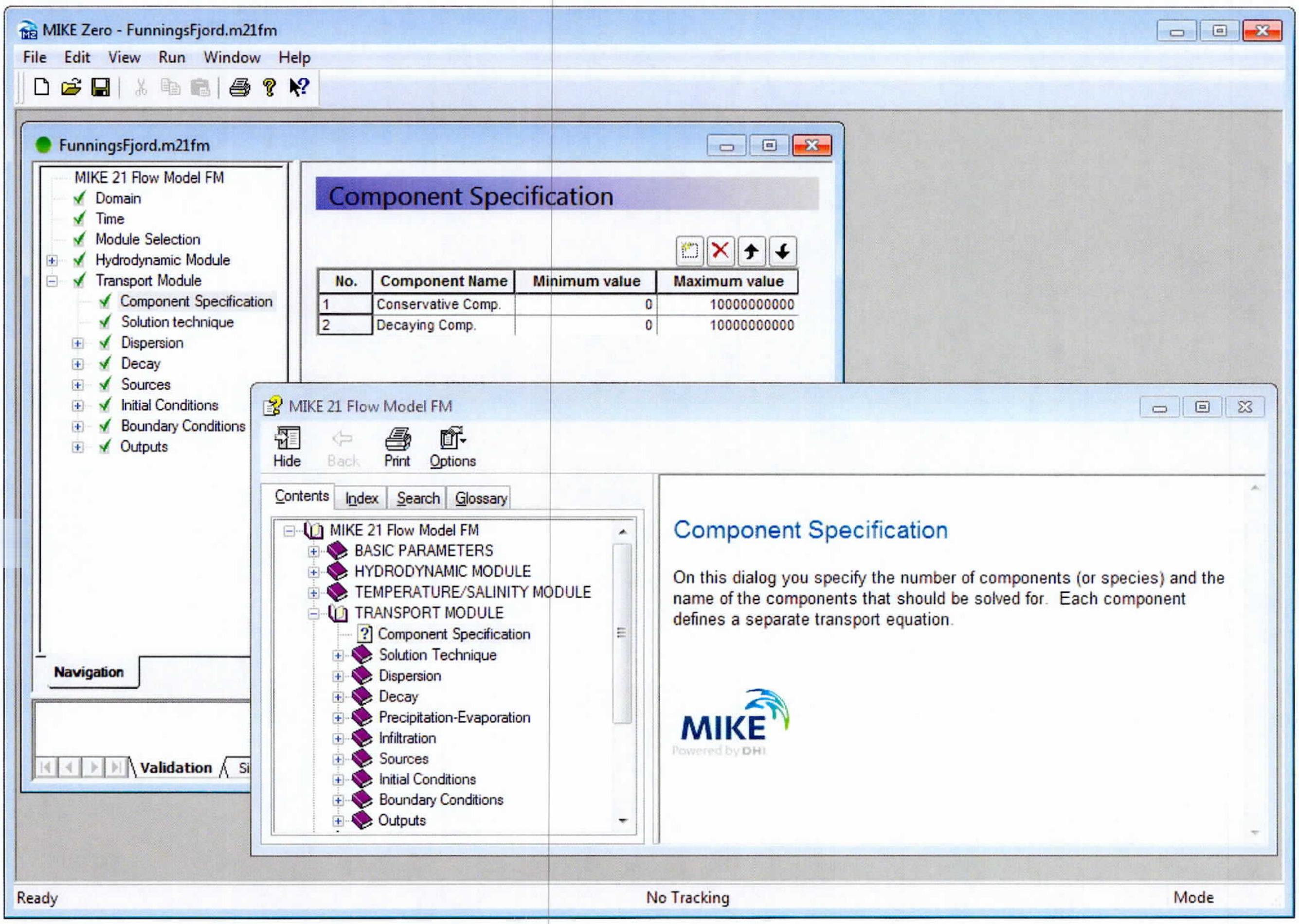
- Component type
- Dispersion coefficients
- Decay information
- Initial conditions
- Boundary conditions



Example of Flexible Mesh generated for a flushing study in Port of Malmoe, Sweden. The background image is from MIKE C-Map which enables extraction of land contours and water depths from digitized Admiralty Charts provided by Jeppesen Norway

Model Output Data

The output from the model includes the concentrations of the given components.
It is possible to specify the format of the output files in MIKE 21 & MIKE 3 as times series of points, lines, areas and volumes (three-dimensional calculations only).

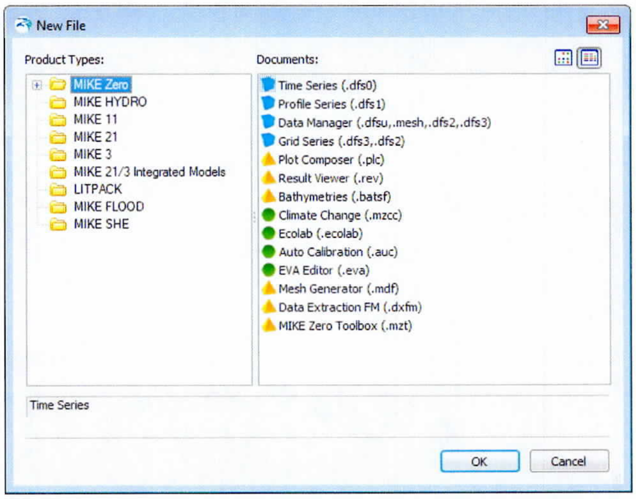


Graphical user interface of the MIKE 21 Flow Model FM, Transport Module, including an example of the Online Help System

Graphical User Interface

The MIKE 21 & MIKE 3 Flow Model FM, Transport Module is operated through a fully Windows integrated Graphical User Interface (GUI). Support is provided at each stage by an Online Help System.

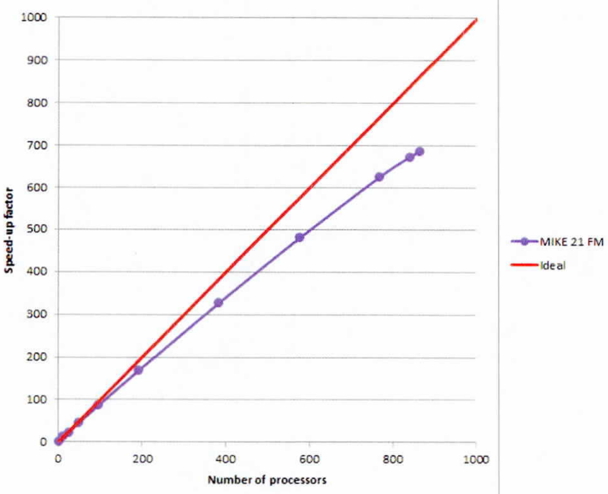
The common MIKE Zero shell provides entries for common data file editors, plotting facilities and a toolbox for/utilities as the Mesh Generator and Data Viewer.



Overview of the common MIKE Zero utilities

Parallelisation

The computational engines of the MIKE 21/3 FM series are available in versions that have been parallelised using both shared memory (OpenMP) as well as distributed memory architecture (MPI). The result is much faster simulations on systems with many cores.



MIKE 21 FM speed-up using a HPC Cluster with distributed memory architecture (purple)

Hardware and Operating System Requirements

The MIKE 21 & MIKE 3 Flow Model FM Transport Module supports Microsoft Windows 7 Professional Service Pack 1 (32 and 64 bit), Windows 8.1 Pro (64 bit), Windows 10 Pro (64 bit) and Windows Server 2012 R2 Standard (64 bit). Microsoft Internet Explorer 9.0 (or higher) is required for network license management as well as for accessing the Online Help.

The recommended minimum hardware requirements for executing the MIKE 21 & MIKE 3 Flow Model FM Transport Module are:

Processor:	3 GHz PC (or higher)
Memory (RAM):	4 GB (or higher)
Hard disk:	160 GB (or higher)
Monitor:	SVGA, resolution 1024x768
Graphic card:	64 MB RAM (256 MB RAM or higher is recommended)

Support

News about new features, applications, papers, updates, patches, etc. are available here:

www.mikepoweredbydhi.com/Download/DocumentsAndTools.aspx

For further information on MIKE 21 & MIKE 3 Flow Model FM software, please contact your local DHI office or the support centre:

MIKE Powered by DHI Client Care
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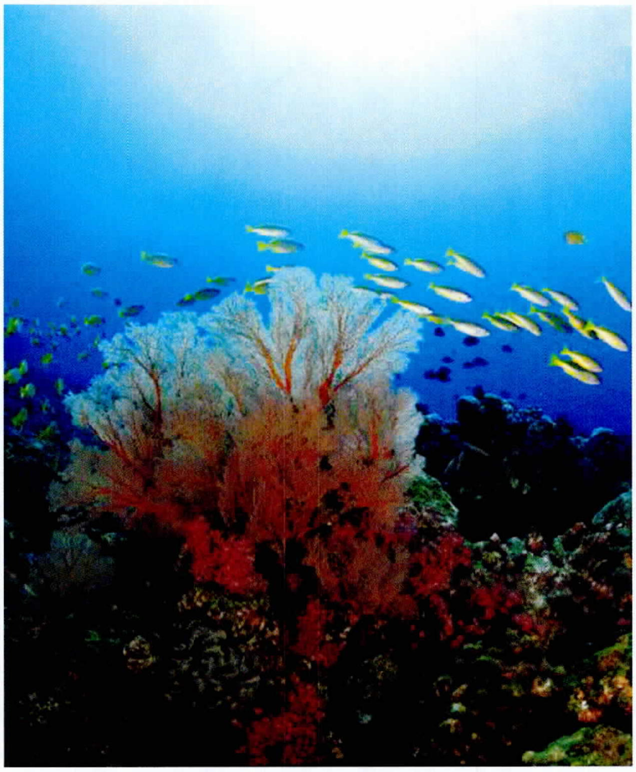
Tel: +45 4516 9333
Fax: +45 4516 9292

mike@dhigroup.com
www.mikepoweredbydhi.com

Documentation

The MIKE 21 & MIKE 3 Flow Model FM models are provided with comprehensive user guides, online help, scientific documentation, application examples and step-by-step training examples.



APPENDIX C– Water quality model results: scenario concentrations

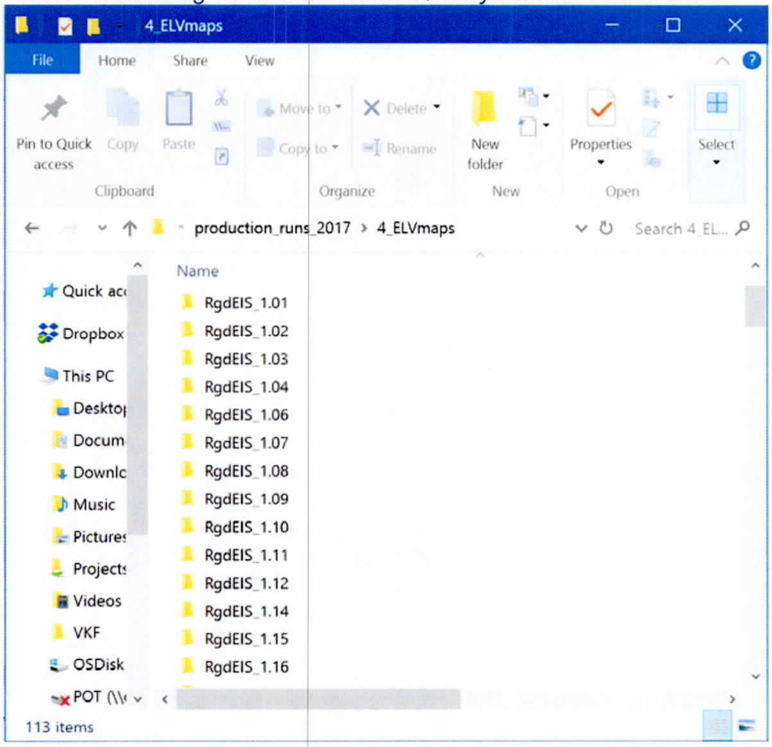
C Water Quality Model Results

Full water quality scenario results are provided as a digital appendix.

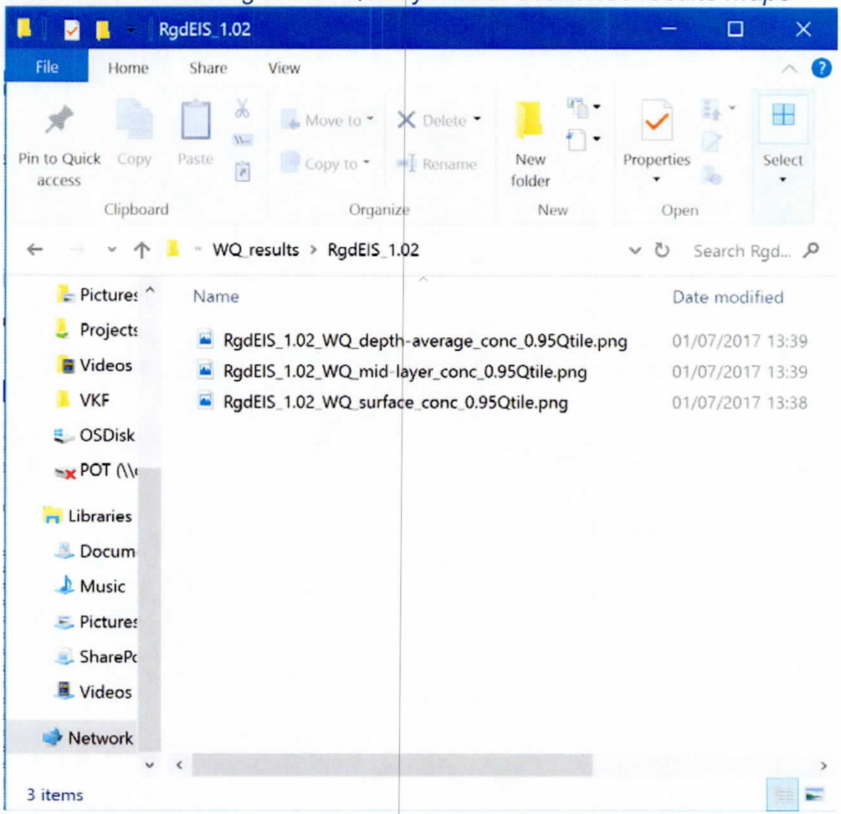
The appendix contains various subfolders named *RgdEIS_1.01, RgdEIS_1.02, RgdEIS_1.03,* etc. (see first image below). The number in the folder name refers to the water quality model scenario ID (see Table 6.2).

Within each sub-folder are a series of image files (.png) which show the results map for that scenario. The file name of the images refers to the water quality model scenario, the vertical reference layer, and the representative concentration. For example, the second image below shows the results for water quality model scenario 1.02, which are for a 95 percentile concentration. There are three images in the folder representing depth-average, surface, and mid-layer concentrations.

Folder containing results of Water Quality Model Scenarios



Sub-folder containing Water Quality Model Scenarios results maps



Greater Dublin Drainage Project

SID Application

ABP Case file 312131

Appendix Reference:

4.1

Appendix Description:

Sabrina Joyce-Kemper submission 7th June 2024

4.1

[redacted]

From: Aberson, Marja
Sent: 14 March 2019 14:47
To: O'Keeffe, Ciaran; 'dwhite@water.ie'
Cc: Kiernan, Sarah; McGlynn, Stephanie; 'ian.wilson@benthicsolutions.com'
Subject: RE: Malahide - shellfish monitoring

Also – please click on link for latest sample results (early Feb 19) for Malahide as analysed by the Marine Institute
https://webapps.marine.ie/HABs/AreaStatus/AreaStatusSummary?locationId=44&locationNameCode=Malahide%20%20(DN-ME)&locationType=Onshore&isFinfish=false#/biotoxin

Production Area	Sample Site	Sample Date	Species	Tissue	SampleCode	ASP mg/kg	AZP ug/g	DSP ug/g	PSP ug STXdiH equival
Carrigaholt	CE-CT-CT	04/02/2019	Crassostrea gigas	Whole	BTX1906051	n.d.(a)	0.02(a)	<LOD(a)	
Ardgroom	CK-AM-AM	04/02/2019	Mytilus edulis	Whole	BTX1906042	n.d.(a)	<LOD(a)	<LOD(a)	
Gouleenacoush	CK-GH-GH	04/02/2019	Mytilus edulis	Whole	BTX1906041	n.d.(a)	<LOD(a)	<LOD(a)	
Lough Foyle	DL-LF-MF	04/02/2019	Crassostrea gigas	Whole	BTX1906047	n.d.(a)	0.02(a)	<LOD(a)	
Lough Foyle	DL-LF-QP	04/02/2019	Mytilus edulis	Whole	BTX1906046	n.d.(a)	0.02(a)	<LOD(a)	
Lough Foyle	DL-LF-QP	04/02/2019	Ostrea edulis	Whole	BTX1906048	n.d.(a)	0.02(a)	<LOD(a)	
Kilmakilloge	KY-KE-KE	04/02/2019	Mytilus edulis	Whole	BTX1906039	n.d.(a)	<LOD(a)	<LOD(a)	
Carlingford	LH-CL-MY	04/02/2019	Mytilus edulis	Whole	BTX1906045	n.d.(a)	<LOD(a)	<LOD(a)	
Clew Bay North	MO-CN-IL	04/02/2019	Mytilus edulis	Whole	BTX1906044		<LOD(a)	<LOD(a)	
Bannow Bay	WX-BB-BB	04/02/2019	Crassostrea gigas	Whole	BTX1906050	n.d.(a)	<LOD(a)	<LOD(a)	
Bannow Bay	WX-BB-BB	04/02/2019	Mytilus edulis	Whole	BTX1906049	n.d.(a)	<LOD(a)	<LOD(a)	
Donegal Harbour	DL-DH-MS	05/02/2019	Mytilus edulis	Whole	BTX1906043	n.d.(a)	0.02(a)	<LOD(a)	
Malahide	DN-ME-ME	05/02/2019	Ensis siliqua	Whole	BTX1906054		<LOD(a)	<LOD(a)	
Gormanstown	MH-GN-GN	05/02/2019	Ensis siliqua	Whole	BTX1906055		<LOD(a)	<LOD(a)	
Achill South	MO-AS-CN	05/02/2019	Crassostrea gigas	Whole	BTX1906052	n.d.(a)	0.08(a)	<LOD(a)	
Waterford Harbour	WD-WH-WN	05/02/2019	Crassostrea gigas	Whole	BTX1906053	n.d.(a)	<LOD(a)	<LOD(a)	
Wexford Harbour	WX-WH-WH	05/02/2019	Mytilus edulis	Whole	BTX1906040		<LOD(a)	<LOD(a)	

LOD = Limit of Detection, LOQ = Limit of Quantification, ULQ = Upper Limit of Quantification, N.D. = Not Detected

Malahide	DN-ME-ME	05/02/2019	Ensis siliqua	Whole	BTX1906054		<LOD(a)	<LOD(a)		<LOD(a)	<LOD(a)	Op

Thanks
Marja.

Dr Marja Aberson | Jacobs | Senior Marine Ecologist | Environment, Maritime & Resilience | [redacted] | www.jacobs.com

From: Aberson, Marja
Sent: 14 March 2019 13:35
To: O'Keeffe, Ciaran <Ciaran.OKeeffe@jacobs.com>; 'dwhite@water.ie' <dwhite@water.ie>
Cc: Kiernan, Sarah <Sarah.Kiernan@jacobs.com>; McGlynn, Stephanie <Stephanie.McGlynn@jacobs.com>;

'ian.wilson@benthicsolutions.com' <ian.wilson@benthicsolutions.com>
Subject: RE: Marine ecology review of the Ecoli and Fisheries review from Jacobs and the 300k model

HI

FYI- here is the extract from :

Cefas, 2013. Impact of chronic microbial pollution on shellfish. *Project WT093.* Cefas/CREH report to DEFRA. 88 pp (report also attached).

Highlighted for both tables is the values for cockles (assumed worse case) and the 'all species', standard values for the SWD standard of 300 and the Class A of 230

<u>Note –</u> in table 5.3 of the memo i mistakenly lifted of the values for all three species for 75% target annual compliance for Class A and not 80%

Table 5 - Indicative water standards required to achieve shellfish flesh standard of 300 *E. coli* MPN/100g)

Species	No samples annual	Target annual compliance rate (%)	Compliance required in individual samples (%)	Geomean required in flesh (MPN/100g)	Estimated geomean *E. coli* in seawater (cfu/100ml)	Estimated 90%ile *E. coli* in seawater (cfu/100ml)
Mussels	4	95	99	28	2.2	8
	4	90	97	45	3.4	13
	4	80	95	57	4.3	16
	4	75	76	149	10	38
	12	90	95	57	4.3	16
	12	80	87	97	7	26
	12	75	76	149	10	38
Pacific oysters	4	95	99	14	2.1	16
	4	90	97	26	3.6	27
	4	80	95	36	4.8	36
	4	75	76	122	14	108
	12	90	95	36	4.8	36
	12	80	87	71	9	66
	12	75	78	112	13	100
Cockles	4	95	99	8	0.03	0.3
	4	90	97	16	0.05	0.5
	4	80	95	23	0.07	0.7
	4	75	76	102	0.28	2.8
	12	90	95	23	0.07	0.7
	12	80	87	53	0.16	1.5
	12	75	78	93	0.26	2.5
All species	4	95	99	2.8	0.39	5.6
	4	90	97	7.1	0.66	9.5
	4	80	95	11	0.88	13
	4	75	76	74	2.7	38
	12	95	99	2.8	0.39	5.6
	12	90	95	11	0.88	13
	12	80	87	32	1.6	23
	12	75	78	74	2.7	38

Table 6 - Indicative water standards required to achieve shellfish flesh standard of 230 *E. coli* MPN/100g

Species	No. samples /annum	Target annual compliance rate (%)	Compliance required in individual samples (%)	Geomean required in flesh (MPN/100g)	Estimated geomean *E. coli* in seawater (cfu/100ml)	Estimated 90%ile *E. coli* in seawater (cfu/100ml)
Mussels	4	95	99	21	1.7	6
	4	90	97	34	2.7	10
	4	80	95	44	3.4	12
	4	75	76	114	8	30
	12	90	95	44	3.4	12
	12	80	87	75	5.5	20
	12	75	76	114	8	30
Pacific oysters	4	95	99	11	1.7	12
	4	90	97	20	2.9	21
	4	80	95	28	3.8	28
	4	75	76	94	11	85
	12	90	95	28	3.8	28
	12	80	87	55	7	52
	12	75	78	86	11	79
Cockles	4	95	99	5.8	0.02	0.2
	4	90	97	12	0.04	0.4
	4	80	95	18	0.06	0.6
	4	75	76	79	0.22	2.2
	12	90	95	18	0.06	0.6
	12	80	87	41	0.12	1.2
	12	75	78	71	0.2	2.0
All species	4	95	99	2.2	0.33	4.8
	4	90	97	5.4	0.57	8
	4	80	95	8.7	0.75	11
	4	75	76	57	2.3	33
	12	95	99	2.2	0.33	4.8
	12	90	95	8.7	0.75	11
	12	80	87	25	1.4	20
	12	75	78	50	2.1	30

Dr Marja Aberson | Jacobs | Senior Marine Ecologist | Environment, Maritime & Resilience | +[redacted] | www.jacobs.com

From: O'Keeffe, Ciaran
Sent: 14 March 2019 12:08
To: 'dwhite@water.ie' <dwhite@water.ie>; Aberson, Marja <Marja.Aberson@jacobs.com>
Subject: FW: Marine ecology review of the Ecoli and Fisheries review from Jacobs and the 300k model

fyi

From: Cathriona Cahill <Cathriona.Cahill@rpsgroup.com>
Sent: 14 March 2019 11:33
To: O'Keeffe, Ciaran <Ciaran.OKeeffe@jacobs.com>
Cc: McGlynn, Stephanie <Stephanie.McGlynn@jacobs.com>; Ian Wilson <ian.wilson@benthicsolutions.com>
Subject: [EXTERNAL] Fwd: Marine ecology review of the Ecoli and Fisheries review from Jacobs and the 300k model

Hi Ciaran
Ian has set out some notes below on his review of the memo
Chat at 12

Get Outlook for Android

From: Ian Wilson <ian.wilson@benthicsolutions.com>
Sent: Thursday, March 14, 2019 11:03:57 AM
To: Cathriona Cahill
Cc: James McCrory; Simon Zisman
Subject: Marine ecology review of the Ecoli and Fisheries review from Jacobs and the 300k model

CAUTION: This email originated from outside of RPS.

Cathriona,

Thanks for the document. This makes for an interesting read and is very useful as a general literature review of the situation. However, this has highlighted a few potential points.

- The proposal for revised *E.coli* discharge is 300k/100ml, would appears to be very conservative and may create unnecessary impacts.
- The revised model was pulled out of the response document (already sent in the submissions). This uses the 250cfu/100ml as the bottom contour so is very insensitive to low level contours that may exist over the shellfish waters as a whole.
- The review was not specific for *Ensis*, but from an ecological point of view, the impact to this species from a chronic coliform is more likely to reflect that of the cockle than the mussel. This means that this species will be quite sensitive to continual import inputs.
- The details in the shellfish study indicates that there is a direct linear relationship between water quality and shellfish uptake of coliforms. Uptake is rapid within 1 hour of exposure and plateaus at 17 hours. Flesh counts reduce almost as quickly on flushing events so an equilibrium based on a tidal cycle and constant input could be expected.
- The key area of concern would be maintaining a Class A status for this species at these rates. A comparison from the 300k model and the uptake factor described for other species would suggest that this is unlikely to be maintained, although we have no current level of flesh or water quality for this area.
- Comparison with levels given in the submission for Velvet strand varies from 4 to 18 cfu this might be similar to what would be expected at the seabed in the Malahide SW. If we assumed an average of these rates at around 11cfu (based on a tidal flushing), then this would arguably only meet Class B for Mussels, with *Ensis* likely to be significantly more sensitive than this.

Overall, the question of meeting water quality requirement of <250cfu/100ml for the Shellfish waters is likely based on the model, but a chronic release based on the 300,000cfu/100ml is also likely to degrade the waters where Class A is unlikely to be achieved. Therefore, if a specific question is raised as to the expected Class qualification to shellfish as a result of this outfall within the Shellfish waters, it would be impossible to argue against a degradation

Greater Dublin Drainage Project

SID Application

ABP Case file 312131

Appendix Reference:

4.2

Appendix Description:

Sabrina Joyce-Kemper submission 7th June 2024

(4.2)

JACOBS

Memorandum

Kenneth Dibben House
Enterprise Road, Southampton Science Park
Chilworth, Southampton SO16 7NS
United Kingdom
T +44 (0)23 8011 1250
F +44 (0)23 8011 1251

Subject	**Literature review *E. coli***	**Project Name**	Dublin Drainage Project
Attention	<Name>		
From	Marja Aberson		
Date	13 March 2019		
Copies to	<Name>		

1. Aim

This short literature review of accumulation of the bacteria *Escherichia coli* in shellfish, encompasses the following:

Section 2: Summary of data and literature sources used.

Section 3: Potential limitations and important considerations identified.

Section 4: A high-level summary of the sensitivity of targeted commercial shellfish to potential pressures from the proposed discharge during operation (of the marine section).

Section 5: Background summary information of factors affecting concentrations of *E. coli* in the environment, in shellfish, and current understanding of the relationship between these parameters.

Section 6: Additional text to supplement 'The Applicant's response to consultees concerns of potential impact on shellfish waters and shellfish from the proposed discharge (of the marine section), as documented in Jacobs (2019).

2. Methods

Peer and non-peer reviewed literature has been sourced, and these have included the following:

- Cefas Project Reports to DEFRA (2006 --2013).
- Cefas Shellfish Water Quality Investigation Reports (2012)
- Scientific peer-reviewed literature (1984-2018).
- Marine Life Information Network (MarLin): Biology and Sensitivity Key Information Reviews. [Accessed On-Line March 2019]. The reviews are cited from the MarLIN sensitivity assessment process, which is currently being superseded by the MarESA approach to assessment for species and biotopes.

Much of the information summarised in this document, is cited from reports submitted by Cefas to DEFRA as part of the Projects WT1001 ('*Factors affecting the microbial quality of shellfish*') and WT0923 ('*Impact of chronic microbial pollution on shellfish*'). These technical reports themselves provided a comprehensive overview of scientific literature, and report upon results of experimental work that investigate the relationship between concentrations of *E. coli* in ambient waters and in the tissues of shellfish.

3. Limitations and considerations

- **The MarLin sensitivity review data is not available for all commercial shellfish species of interest, and with low level of associated evidence and/confidence in assessments made.**
- **Significant bias in studies of commercial shellfish species (e.g. *Mytilus edulis*) over others (*e.g. Ensis* sp.).**
- **Likely high inter-species variation in accumulation and depuration rates.**
- **Difficulty in assessment of mobile species (e.g. *Cancer pagurus* and *H. gammarus*) due to life history and lack of data.**
- **Assessments of rate of uptake and clearance are often undertaken under a microcosm laboratory condition where expected variations in environmental conditions will not be incorporated.**

4. Sensitivity Review

Table 4 1 summarises the sensitivity review of key commercial species harvested in the area, in response to all key potential pressures of the proposed discharge. Although *Pecten maximus* and *Mytilus edulis* are not listed as a targeted species in Northern Fingal (Table 9.17, EIAR) they are listed as a principal shellfish species in the area (Table 9.16, EIAR).

Potential pressures may encompass physical (smothering, increased sediment deposition and turbidity), chemical (changes in nutrient and oxygenation levels), and biological (increase in pathogens). No sensitivity review data was available for the following commercial species of interest: *Necora. puber, Homarus gammarus, Palaemon serratus* and *Buccinum undatum.*

Except *M. edulis*, all species are assessed to have a low level of intolerance and high recoverability to any potential physical disturbances, and with all species (except *P. maximus*) being of low sensitivity to such pressures overall. All species are assessed to have low level of sensitivity to chemical pressures overall, but with the bivalves *P. maximus, Ensis* sp. and *M. edulis* exhibiting an intermediate level of intolerance to one or both potential chemical pressures listed in Table 4 1. Responses to an increase in microbial pathogens/parasites had only been assessed in *Cancer pagurus* and *M. edulis*; with both species assessed as being of low sensitivity.

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Memorandum

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Table 4 1: Sensitivity of commercial shellfish species, as reviewed under the Marlin sensitivity assessment process.

[illegible]n name	Scientific name	Pressure	Pressure Type	Intolerance	Recoverability	Sensitivity	Evidence/ Confidence	Source
Brown crab	***Cancer pagurus***	Physical	Smothering	Low	Very high	Very low	High	Neal and Wilson (2008)
			Increase in suspended sediment	Low	High	Low	Very low	
			Increase in turbidity	Tolerant	Not relevant	Not sensitive	Very low	
		Chemical	Changes in nutrient level	Tolerant	Not relevant	Not sensitive	Very low	
			Changes in oxygenation	Tolerant	Very high	Not sensitive	High	
		Biological	Introduction of microbial pathogens/parasites	Intermediate	Moderate	Moderate	High	
Velvet swimming crab	***Necora puber***	**No data available**						Wilson (2008a)
European lobster	***Homarus gammarus***	**No data available**						Wilson (2008b)
Shrimp	***Palaemon serratus***	**No data available**						Neal (2008)
Whelk	***Buccinum undatum***	**No data available**						Ager (2008)
Great scallop	***Pecten maximus***	Physical	Smothering	Low	High	Moderate	Moderate	Marshall and Wilson (2008)
			Increase in suspended sediment	Low	High	Low	Low	
			Increase in turbidity	Tolerant	Not relevant	Not sensitive	Not relevant	
		Chemical	Changes in nutrient level	Intermediate	High	Low	Moderate	
			Changes in oxygenation	Low	High	Very low	Low	
		Biological	Introduction of microbial pathogens/parasites	**No data available**				
Razor clam	***Ensis* sp.**	Physical	Smothering	Tolerant	Not relevant	Not sensitive	High	Hill (2006)
			Increase in suspended sediment	Low	High	Low	High	
			Increase in turbidity	Low	High	Low	Moderate	
		Chemical	Changes in nutrient levels	Intermediate	High	Low	Low	
			Changes in oxygenation	Intermediate	High	Low	Moderate	

		Biological	Introduction of microbial pathogens/parasites	**No data available**				
Blue mussel	***Mytilus edulis***	Physical	Smothering	Intermediate	High	Low	Low	Tyler-Walters (2008)
			Increase in suspended sediment	Low	Intermediate	Not sensitive	High	
			Increase in turbidity	Tolerant	Not relevant	Not sensitive	Not relevant	
		Chemical	Changes in nutrient levels	Intermediate	High	Low	Low	
			Changes in oxygenation	Low	Very high	Very low	High	
		Biological	Introduction of microbial pathogens/parasites	Intermediate	High	Low	High	

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5. Accumulation of *E. coli* in commercial shellfish

5.1 *E. coli* concentrations in seawater

The degree of *E. coli* contamination of a receiving water body by a Waste Water Treatment Works (WwTW) will be primarily influenced by the level operational activity of the plant itself, but in addition to this the potential risk of accidental release from sewage overflows or plant failure. Heavy rainfall and increased fluvial inputs may also increase the loading and subsequent *E. coli* contamination of a receiving water body (Craig *et al.,* 2008; Cefas, 2012a; Cefas, 2012b).

The concentration of the bacteria *E. coli* within crude sewage itself will not exhibit a clear normal distribution pattern (curve) with often skewed abundances as bacteria often occurs in clumps. Following dilution with the receiving waters, the distribution curve of bacteria will be expected to flatten across its range of concentrations, thereby also increasing its variation in levels (Cefas, 2013). The fate and transport of faecal bacterial once released into ambient waters will be influenced by a number of complex and interacting processes where concentrations may be further affected by temperature, salinity, tidal conditions, current velocities and geomorphological features of the water body itself. Discharges into shallow tidal inlets with constricted entrances may create complex tidal currents and flow patterns restricting the potential mixing and dilution of any contaminants in the water column (e.g. Portsmouth Harbour, UK (Cefas, 2012a)). Discharges into an open coastal system subject to strong tidal currents may promote rapid diffusion and dilution of faecal bacteria levels in the plume. Hydrodynamic modelling of the narrow, Dart Estuary (Devon, UK) were simulated across five days in January for a sewage overflow of untreated sewage discharge of 200 m^3 (Garcia *et al.,* 2018). It was computed that overall, the largest area of *E. coli* contamination (>10 cfu/100ml) occurred during periods of neap tides and low river discharges, but also with a maximum value obtained during neap tide and high river discharges; these both representing the worse-case scenarios.

The exponential decay (die-off) rates of *E. coli* in the environment will be a function of natural factors including temperate, salinity and irradiation (Garcia *et al.,* 2018). A review by Craig *et al.,* (2004) concludes that in general, within the water column, there is a positive relationship with rates of decay and temperature and sunlight. However, an increase in turbidity of the water may restrict any solar penetration through the water column. An *in-situ* study by Craig *et al,* (2004), further showed that *E. coli* can persist in coastal sediments even after any rapid decline of levels in the overlying water. Within contaminated sediments, particle size has also been shown to be important factor with an increase in *E. coli* decay rates in those sediments comprised of larger particles and containing low organic carbon. It may be that increased nutrient availability in those finer sediment may provide an important food source for bacteria.

5.2 *E. coli* concentrations in shellfish (review by Cefas, 2012c)

Accumulation of *E. coli* bacteria in bivalves will occur during filter-feeding (process of water pumping and filtration). This process can be limited by the physical properties of the filter pump and concentration of food in the water. Filter feeding has been shown to be autonomous and not regulated at the organism level with processes kept open and operating at a constant rate during optimal conditions. The efficiency of accumulation can naturally vary with external environmental conditions such as concentration and composition of particulates, temperature, current speed, and in part viscosity of the water.

Pumping rates are shown to increase with increasing temperature and also with a decrease in viscosity; of which is in itself temperature dependant. Effects of changes in salinity have not been shown to be as important as temperature but with a general pattern of delayed valve opening with a decrease in salinity. Euryhaline bivalves can tolerate and thus feed in lower saline conditions (e.g. *M. edulis*) than others (e.g. *Ostrea edulis* and *Ensis* sp.). Species-specific responses to different environmental conditions thus may overall, naturally result in different rates of accumulation.

There has been shown to be wide inter-specific differences in relative levels of accumulation and so contamination in different bivalves. For example, levels of *E. coli* in *M. edulis* and *Cerastoderma edule* have been shown to be approximately 1<2, to 3 times higher than *Magallana gigas* (previously called *Crassostrea gigas*), respectively. Variations in accumulation may be attributable to physiological differences but also due to methods of growth (e.g. in bags on bed verses grown directly on bed itself). Even among shellfish of the same species in any one bed, the distribution of *E. coli* in tissues can be variable both spatially and over time, with levels between monitoring points varying by 2-3 orders of magnitude within just a few hours (Walker *et al.,* 2017; Cefas, 2011).

5.3 Uptake of *E. coli* in shellfish in response to concentrations in seawater

It can be difficult to directly quantify the relationship between *E. coli* concentrations in the water to the uptake and accumulation in the flesh of shellfish. However, recently funded DEFRA projects undertaken by Cefas in the UK sought to: explore the relationship between microbial quality of shellfish flesh and seawater, investigate the dynamics of uptake and clearance of *E. coli* in shellfish subject to chronic contamination, identify water concentrations of *E. coli* which would be compliant with the Shellfish Water Directive (SWD) "guideline" standard (G) of 300 cfu/100g (in 75% of samples), and make recommendations regarding an *E. coli* standard (water column standard verses shellfish flesh) for shellfish protected areas (Cefas, 2011;Cefas, 2012b; Cefas, 2013).

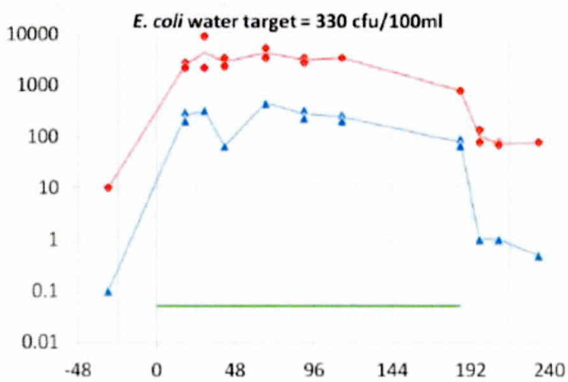
5.3.1 Relationship between concentrations in seawater and shellfish

The relationship between *E. coli* counts in sampled seawater and shellfish flesh of three species (*O. edulis, M. gigas* and *Mytilus* spp. (*M. edulis* and *Mytilus galloprovencialis* data not separated)), sampled between 1991-1994 within six different production areas in the UK was analysed (Cefas, 2011). The level of contamination between the three bivalves, as expected was variable with *M. edulis* being more contaminated overall and for all species a greater geometric mean concentration calculated in the tissues than in the seawater. For all data pooled (all three species, n=602) a positive linear relationship between increasing *E. coli* levels in the seawater and in the shellfish was apparent, however, with a wide spread of values around the computed regression line. This wide range in measured values around the predicted values is an expected artefact of data obtained under natural environmental conditions.

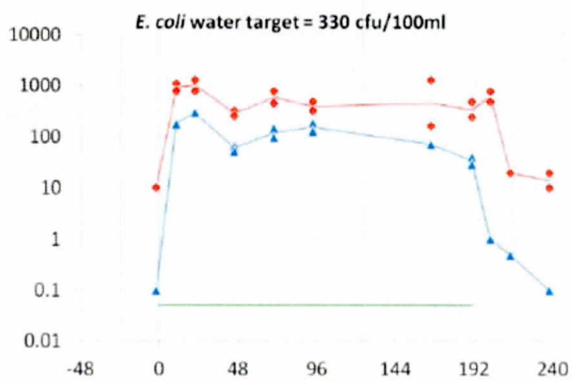
Microcosm tank experiments monitored the uptake of *E. coli* in the tissues of the bivalves *M. edulis, M. gigas* and *C. edule* exposed to chronic exposure (continuous dosing for 5 days) to a range of water quality levels (1 cfu/100ml – 330 cfu/100ml) (Cefas, 2013). Across all concentrations, a rapid uptake of *E. coli* was shown for all species to a maximum 'equilibrium' (plateau) state (within 17 hours) and on cessation of dosing, a rapid clearance was also exhibited.Previous studies have shown that there is a threshold for *E. coli* concentrations in the water, above which bivalves are unable to accumulate more bacteria, however this maximum 'equilibrium' state will vary between both individuals and species (Cefas, 2011).

Figure 5.1 shows the time-series data for each species in the microcosm tanks under the maximum target *E. coli* seawater conditions (330 cfu/100ml). Changes in concentrations in the shellfish appear to mirror changes in the ambient seawater for all species during the 10-day experiment. Where only a low percentage (35% overall) of the variation in concentrations of shellfish tissue was explained by concentrations in the water from analysis of historic monitoring data (Cefas, 2011), under these microcosm conditions, this was found to be much higher at 55 – 60%. The overall factorial increase between seawater and shellfish *E. coli* concentrations (as calculated across all tank concentrations) ranged from 11.7 for *M. gigas,* 15.2 for *M. edulis,* and 330 for *C. edule* with a wider range of accumulation rates found overall for *C. edule* at each seawater tank concentrations. Although flesh concentrations increased linearly with concentrations of the tank seawater, there was no direct association with an increase in seawater concentration of the microcosms and resulting accumulation factor.

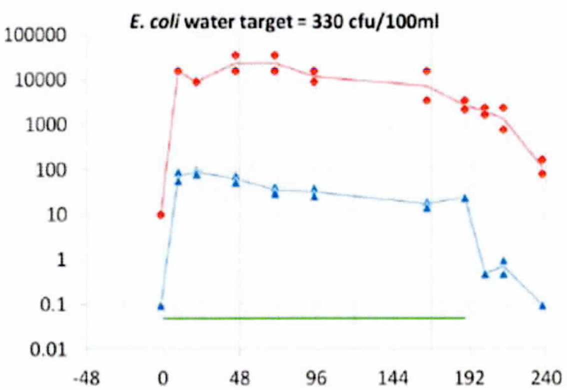
The rate of accumulation in tissues in the study was overall proportionate to the changes in water quality, the rate of clearance following the end of dosing was not as much (Figure 5.1). Bacteria can be rapidly cleared from shellfish when exposed to clean waters, with an initial phase of greatest clearance lasting <10hrs then followed by a less evident phase of 10-30 hrs. Within 24 hours of exposure to un-contaminated waters, clearance rates of approximately 100 times the initial concentrations have been observed in mussels and oysters (Cefas, 2011).



a) *Mytilus edulis*



b) *Magallana gigas*



c) *Cerastoderma edule*

Figure 5.1: Time series of levels of *E. coli* in tank water and tissues of a) *M. edulis,* b) *M. gigas* and c) *C. edule* for the target tank water concentration of 330 cfu/100ml. X-axis is hours relative to start of sewage dosing with Green line = period of sewage dosing. Red line = flesh concentrations and Blue line = tank water concentrations (Cefas, 2013).

Investigations of *E. coli* accumulation in *M. edulis*, *C. edule* and *M. gigas* was also undertaken in Mumbles Bay, UK across 10- day exposure period in September 2011, by attaching specimen bags to the intertidal zone at the site (Cefas, 2013). The relative ordering in inter-species *E. coli* accumulation remained valid with other studies and the microcosm experiment (e.g. greatest uptake in *C. edule*). However, no clear statistically significant difference between mean *E. coli* concentrations between the three species sampled from these environmental investigations was reported; only in comparison with *E. coli* seawater concentrations. Variation recorded in both water and flesh concentration is expected and will reflect variations in the environmental waters.

Direct measurements of water quality in the study area did not significantly correlate with *E. coli* shellfish concentrations. Therefore, a hydrodynamic two-dimensional water quality model (DIVAST) predicted *E. coli* concentrations for Swansea Bay was also done to provide near-real-time prediction of *E. coli* concentrations for where the shellfish bags had been positioned. The results of the model could not find a statistically significant correlation between water quality and the laid shellfish in this study. Diurnal and tidal patterns in concentrations have been found to be important, indicating a ubiquitous and high 'natural' variability in *E. coli* concentrations with differences exceeding 2 $\log_{10}$ orders diurnally even under dry conditions (review by Cefas, 2013). Such short term variability in bacterial concentrations may now be considered the 'normal' condition

5.3.2 Predicting compliance using *E. coli* seawater concentrations

Using the historic data collected in 1991-1994, models were computed for the three shellfish species *O. gigas, M. gigas* and *Mytilus* spp., to predict compliance with the SWD G value of 300 cfu/100g against a range of *E. coli* water quality concentrations (Cefas, 2011). The greatest proportion of samples compliant was shown to be for the Pacific oyster *M. gigas*. Assessing all three species together, indicated that a geometric mean threshold of 9.6 cfu/100ml and a 90th percentile of 55 cfu/100ml in seawater would be equivalent to the current SWD G standard.

The indicative thresholds for *E. coli* water concentrations for each species to meet the SWD G based on this study is listed in Table 5 1, and for 90% compliance with thresholds for Class B (<4,600 cfu/100g) is listed in Table 5 2. However, in terms of compliance with Class A threshold (<230 cfu/100m) none of the samples in this study met the criteria.

Later studies by Cefas (2013) also calculated indicative water quality standard values, to meet both the SWG G and Class A thresholds for concentration of *E. coli* in shellfish. Estimations were semi-quantitative (pass/fail), based either on samples taken quarterly, or monthly per annuum looking at overall distribution of readings to derive parameters. It is assumed that samples are taken equally spaced through the year and are independent; excluding any risk-based or biased sampled.
Table 5 1 and Table 5 3 lists the indicative standards estimated for meeting the SWD G and Class A thresholds based on monthly sampling per annum. The indicative *E. coli* seawater concentrations for individual species are more conservative when compared to values calculated based on monitoring data (Cefas, 2011).

As the thresholds determined in the Cefas (2011) study were based on historic data (1991-1994), it has been recommended that these are validated with more up to date samples from production areas to draw more accurate comparisons and be comparable with the microcosm experiments of project WT0923 (Cefas, 2013).

Table 5 1: Indicative concentrations of *E. coli* in seawater (geometric mean and 90th percentile) to achieve 75%* compliance with SWD G (300 fcu/100g) in shellfish. *Cefas (2013) data predicted for 75% target annual compliance rate.

Species	Study Type	Geometric mean Seawater cfu/100ml	90th percentile seawater cfu/100ml	Sample size	Reference
Mytilus spp.	Natural sampling	8.9	102	313 individuals (pooled sites)	Cefas (2011)
Mytilus edulis	Microcosm	10	38	predicted from 12 samples taken per annum	Cefas (2013)
Magallana gigas	Natural sampling	41	492	111 individuals (pooled sites)	Cefas (2011)
Magallana gigas	Microcosm	13	100	predicted from 12 samples taken per annum	Cefas (2013)
Ostrea. edulis	Natural sampling	8.3	64	178 individuals (pooled sites)	Cefas (2011)
Cerastoderma. edule	Microcosm	0.26	2.5	predicted from 12 samples taken per annum	Cefas (2013)

Table 5 2: Indicative concentrations of *E. coli* in seawater (geometric mean) to achieve target annual 90% compliance with SWD standard for harvesting Classification B (<4,600 cfu/100g) in shellfish (Cefas, 2011).

Species	Study	Geometric mean seawater cfu/100ml	Number of samples
Mytilus spp.	Natural sampling	33	313 individuals (pooled sites)
O. edulis	Natural sampling	177	178 individuals (pooled sites)
M. gigas	Natural sampling	4,200	111 individuals (pooled sites)

Table 5 3: Indicative concentrations of *E. coli* in seawater (geometric mean and 90th percentile) to achieve annual 80% compliance with SWD standard for harvesting Classification A (<230 cfu/100g) in shellfish (Cefas, 2013).

Species	Study	Geometric mean seawater cfu/100ml	90th percentile seawater cfu/100ml	Number of samples/annum
M. edulis	Microcosm	8	30	12
C. edule	Microcosm	0.2	2.0	12
M. gigas	Microcosm	11	79	12

6. The Greater Dublin Drainage Project (GDD)

The below section lists responses from the 'Applicant' to consultee submissions following the lodging of the Planning Application; responses are regarding the impact of Proposed Project on shellfish and shellfish waters during operation. The responses are sourced and numbered, as cited in the *Greater Dublin Drainage Report: Response to Submissions* (Jacobs, 2019).

Succeeding each statement response(s) is further information that aims to support/ or expand upon these given statements.

6.1.1 Concerns regarding impact of Proposed Project on designated shellfish waters

457. In summary the plumes arising…….from the discharge of treated wastewater from the proposed outfall pipeline route (marine section) fall outside the designated shellfish waters. Furthermore, the modelled data for the discharge during the Operational Phase indicates that the impact plume has a limited spatial impact and will disperse significantly into the prevailing oceanography at the site. This fact coupled with the discharge parameters will ensure there will be no impact to shellfish waters.

Response remains valid.
Comparisons with monitoring studies of the dispersal and fate of *E. coli* in water bodies in the UK where they are more restrictive in tidal flow and exposure, would support conclusions that the outcome of the model for the GDD project has a plume with a restricted impact on any surrounding areas, such as the designated shellfish waters at Malahide.

6.1.2 Concerns regarding impact of Proposed Project on shellfish

364. Schedule 2 of S.I. No. 268/2006 does not set values for the coliform concentrations in the water column. Schedule 4 of S.I. No. 268/2006 sets a guide value for coliform concentrations equal to or less than 300 faecal coliforms per 100 millilitres in the shellfish flesh and intervalvular liquid but does not set values for coliform concentrations in the water column.

Response remains valid.
There is at present no agreed upon *E. coli* seawater concentration guideline value in which to monitor against. Recent studies have shown that for compliance with the current SWD G, there can be a wide range in predicted *E. coli* water concentrations calculated, that primarily depend on the targeted species in question and methods of assessment (e.g. microcosms vs. environmental studies). As such these studies have not support the application of a single guideline value for water quality standard, where more than one species is harvested.

Such studies done to date have focussed on only a few commercial species, primarily the blue mussel *Mytilus edulis,* the Pacific oyster *Magallana gigas (*previously known as *Crassostrea gigas)* and the common cockle *Cerastoderma edule*. There is no data available for those commercial bivalve species known to be harvested within the study area (razor clam *Ensis* sp), whelks (*Buccinum undatum*) and large mobile crustaceans (*Homarus gammarus* and *Cancer pagurus*).

366. There is no direct relationship between the concentration of coliforms in overlying water and the concentration of coliforms in shellfish flesh as both the uptake/accumulation and clearance/removal of coliforms by filter-feeding shellfish is a dynamic process affected by many variables (e.g. temperature, food availability, salinity, shellfish age, season, reproductive state, health of the shellfish and the impacts of toxins and other contaminants.

Statement may require further validation if questioned further on.
Although there is still a high level of variance in the data that remains unexplained when paired values of concentrations of *E. coli* in seawater verses shellfish are analysed; there is still a clear linear relationship between these two measured parameters. However, differences in the strength of this relationship has been shown to vary between species and between artificial microcosm conditions to *in situ* studies in the field, where natural fluxes in environmental conditions may mask any patterned responses or reduce any predicted effects.

It will be important to acknowledge that following exposure that there will be likely rapid increase (within 1 hour) in uptake and assimilation of *E. coli* in tissues of bivalves, with 'equilibrium' reached within 17 hours (in these tested cases), and clearance following end of exposure. Microcosm studies done to date have looked at chronic exposure, with aim of continuous contamination over a period of 5 days. In this data set, declines and subsequent increases in tissue concentration occurred during this dosing period when there had been a short-term fault in equipment, reducing the flow of diluted sewage into the test tanks. The patterned decline with decline in water concentration bears evidence that under natural conditions when these fluxes occur it will instantly result in a reduction in tissues of shellfish, and as likely to occur regularly and over longer periods this will naturally allow clearance to occur (e.g. during tidal periods). However, it also highlights the rapid physiological response by bivalves to uptake, which may occur following heavy rainfall for example which may for the short term increase uptake in tissue of resident shellfish.

Variations in uptake and maximum concentrations at 'equilibrium' state between species has been shown, with an agreed ranking of greater concentration accumulated in cockles compared to mussels and oysters. The literature suggests that there is a maximum accumulation level a species can reach, independent of any further increase concentrations in the ambient waters. The duration of exposure will be of importance, for allowing full clearance from the tissues. It is unlikely that bivalve shellfish of the study area will be subject to prolonged exposure periods comparable with these experimental studies (e.g. 5-10 days) and

367. The potential impacts on the Malahide shellfishery were examined using a revised modelling simulation examining the discharge of coliforms at a concentration of 300,000 cfu/100ml for both the proposed Average Daily Flow and Flow to Full Treatment scenarios.

370. For Flow to Full Treatment scenario, the maximum predicted coliform concentration in the water near the seabed was 327 cfu/100ml. For 80% of the time the predicted concentrations were less than 147 cfu/100ml with the average coliform concentration over the course of the simulation predicted to be 78 cfu/100ml. The coliform concentrations fluctuate between a maximum value on flooding tides and zero concentrations on ebbing tides. This provides equal time for uptake/accumulation and subsequent clearance/removal of any coliforms by shellfish. No impact is predicted on the shellfish water quality as a result of the proposed discharge.

Response may require to be updated
The modelled simulation at 300,000 cfu/100ml for normal operation of the proposed WwTP may be considered to be conservative (C. O'Keeffe *pers. comm.* 12 March 2019). 2018 discharge data from Ringsend WwTP have reported variable levels, with very few data points exceeding 200,000 cfu/100ml, and with an overall average discharge of 79,000 cfu/100ml. The maximum modelled coliform in the water near the seabed of 327 cfu/100ml, will therefore, likely be considerably less than this, as will the concentrations for 80% of a given period, and the overall average.

There will be variation in rate of uptake and rate of clearance between species, as shown in previous studies. This will also be expected to vary across seasons. During winter periods (low temperature and solar irradiation), the natural decay of *E. coli* in the water column may be slower than in the summer months, possibly also further impacted by increased rainfall and fluvial inputs during this period. The lowered values currently sourced for the Ringsend WwTP were taken outside of the bathing season (e.g. the winter months with no UV treatment) and excluding an overflow or plant failure event, may indicate a worst-case chronic exposure scenario for the receiving water body and one that is not as conservative as the modelled scenarios.

Local shellfisheries harvest throughout the year but with specific collection periods for some species. Harvesting of the razor clam *Ensis* sp. (predominantly *Ensis siliqua)* occurs over the winter months in the area. The Malahide production area (site name: DN-ME) has a shellfish harvesting classification of A, and as per the status of the last sample analysed (taken 5 February 2019), remains as 'Open'. Monthly monitoring data for biotoxins over the last 12 months (January 2018 – February 2019) reported on only one occasion (14 June 2018) a failure (status changed to 'Closed pending') but an additional sample taken that month, had a reported status then of 'Open' (Marine Institute, 2019).

Unfortunately, studies to date of *E. coli* accumulation in *Ensis* spp. have not been undertaken, with focus on other commercially important bivalves. Substances within sediments are known to have longer residence time than water-borne contaminants. As bottom dwelling infaunal species, there is the higher risk that they will be exposed to any contaminants within the sediment compared to bivalves that grow above the seabed. *Ensis* spp. tend to inhabit coarser sediments, but with spatial distribution in different sediments between this con-specifics. Such sediments will likely contain a lower organic content and thus support a relatively lower resident population of bacteria than finer sediments.

It will be imprudent to estimate a potential accumulation factor in the tissues of razor clams at Malahide as current work has shown a wide range of uptake rates and maximum concentrations between bivalve species, and with spatio-temporal differences also expected. The distance of the Malahide production area from the point-source (outfall pipe), and consideration of the predicted plume in the far field zones, and the current data from an existing WwTP in Dublin Bay, reduces the level of assessed risk of contamination to shellfish. It will be important to acknowledge potential increased risks to harvesting post heavy rainfall events and the expected natural tidal and seasonality in water column *E. coli* concentrations when harvesting.

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8. Glossary

Definitions sourced and adapted from: Cefas (2012c),

Accumulation: Uptake and storage of FIOs within the cells of the living shellfish species.

Accumulation factor: Measure of the intensity of the accumulation of FIOs in bivalve shellfish. This measure is given by the ration between the concentration of FIOs in shellfish relative to the concentration of FIOs in the overlying water.

Bivalve filter pump: Group or bands of lateral cilia on filaments arranged in parallel within the mantle cavity of the bivalve.

Chronic exposure: Contact of shellfish with *E. coli* in the overlying waters that occurs over a long time (e.g. > 5 days).

Clearance: Process by which shellfish eliminate FIOs (e.g. from filter-feeding in bivalve species).

Microcosm: Artificial simplified ecosystem up under often laboratory conditions to predict responses to a variation in environmental conditions.

of quality based on the recent model used and the uptake data that is currently available for this species. We need to be sure of IW and Jacobs position on this if this is raised in the OH. Note that this is a socio-economic and not an ecological issue.

Regards

Ian Wilson
Benthic Solutions Limited
[redacted]
ian@benthicsolutions.co.uk
www.benthicsolutions.co.uk

From: Cathriona Cahill <Cathriona.Cahill@rpsgroup.com>
Sent: 13 March 2019 18:38
To: Ian Wilson <ian.wilson@benthicsolutions.com>
Cc: James McCrory <James.McCrory@rpsgroup.com>; Simon Zisman <Simon.Zisman@rpsgroup.com>
Subject: Fwd: Marine

Hi Ian
See attached.
I will give you a call to discuss in the morning

Get Outlook for Android

From: McGlynn, Stephanie <Stephanie.McGlynn@jacobs.com>
Sent: Wednesday, March 13, 2019 6:30:34 PM
To: Cathriona Cahill
Cc: Kiernan, Sarah
Subject: RE: Marine

CAUTION: This email originated from outside of RPS.

Hi Cathriona,

Please see attached preliminary memo re. shellfish from our expert.

Could you please revert as soon as possible with any comments and we will aim to arrange a call with the shellfish experts and relevant specialists tomorrow.

Kind regards,

Stephanie

From: Cathriona Cahill <Cathriona.Cahill@rpsgroup.com>
Sent: 13 March 2019 15:29
To: McGlynn, Stephanie <Stephanie.McGlynn@jacobs.com>; Kiernan, Sarah <Sarah.Kiernan@jacobs.com>

Cc: O'Keeffe, Ciaran <Ciaran.OKeeffe@jacobs.com>
Subject: [EXTERNAL] Marine

Hi Girls
Apologies for the delay.

Just to note that Ian has proposed to include Figure 1 which addresses the failure event at the outfall pipeline. (please note this is new information)
However, I am unsure now if this should be included based on Ciarán's email last night regarding the change in the failure event.
Also see comment re: shellfish.

Let me know if you need to discuss.

Cathriona Cahill
Associate Environment
RPS | Consulting UK & Ireland
West Pier Business Campus
Dun Laoghaire, Co. Dublin A96 N6T7, Ireland
[redacted]
rpsgroup.com

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Greater Dublin Drainage Project

SID Application

ABP Case file 312131

Appendix Reference:

4.143

Appendix Description:

Sabrina Joyce-Kemper submission 7th June 2024

[redacted]

From: Dara White <dwhite@water.ie>
Sent: 25 April 2019 12:46
To: Ronan Kane
Subject: FW: Confidential: GDD - Ecoli levels in Discharge
Attachments: 20190324_GDD_20k_cfu_v3.docx

From: O'Keeffe, Ciaran [mailto:Ciaran.OKeeffe@jacobs.com]
Sent: 25 March 2019 18:27
To: Dara White <dwhite@water.ie>
Subject: Confidential: GDD - Ecoli levels in Discharge

Dara,

Amended document on the 20,000 cfu/100ml discharge run which includes analysis of the ecoli concentrations in the water column along the southern boundary of the designated shellfish area.

Regards

Ciarán

Jacobs Engineering Ireland Limited
Merrion House, Merrion Road, Dublin 4, Ireland
Registered in Ireland under number 111945

4·36.

Summary of UV disinfection runs

Two scenarios were simulated to assess the impacts of discharging UV treated effluent with a coliform concentration of 20,000 cfu/100ml.

Scenario #1: Synthesised flow @ 20,000 cfu/100ml, no wind

The model commenced the simulation on 18/04/2015 at 00:00hrs with the proposed GDD Project discharging at the synthesised flow profile presented in Figure 1 (below). The Average Daily Flow (ADF) is included in Figure 1 for reference. The concentrations of coliforms in the effluent was 20,000 cfu/100ml. No wind field was specified.

Scenario #2: Synthesised flow @ 20,000 cfu/100ml, recorded wind field

The model commenced the simulation on 18/04/2015 at 00:00hrs with the proposed GDD Project discharging at the synthesised flow profile presented in Figure 1 (below). The concentrations of coliforms in the effluent was 20,000 cfu/100ml. Recorded wind speed and direction data from Dublin Airport was defined and presented in Figure 6 below..

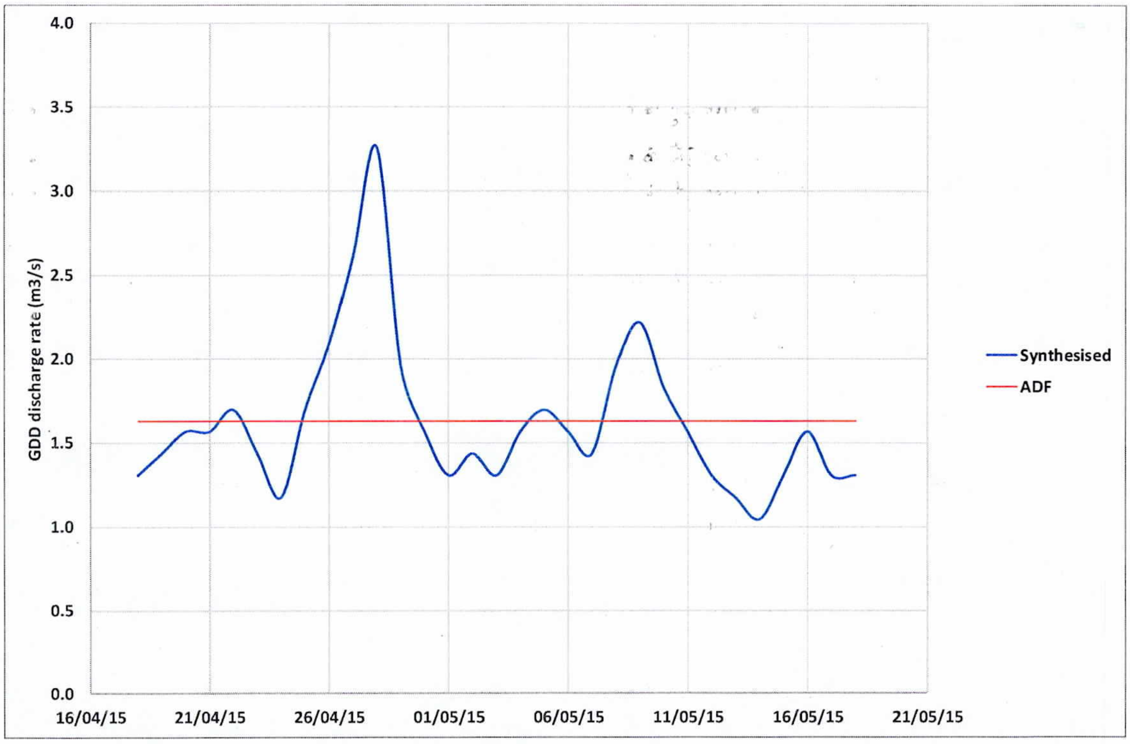

Figure 1: Synthesised GDD discharge rate

The results were analysed at the designated Malahide Shellfishery sampling point. The concentration of coliforms over the course of the simulation for both scenarios (No Wind, and Wind) are presented in Figure 2 below.

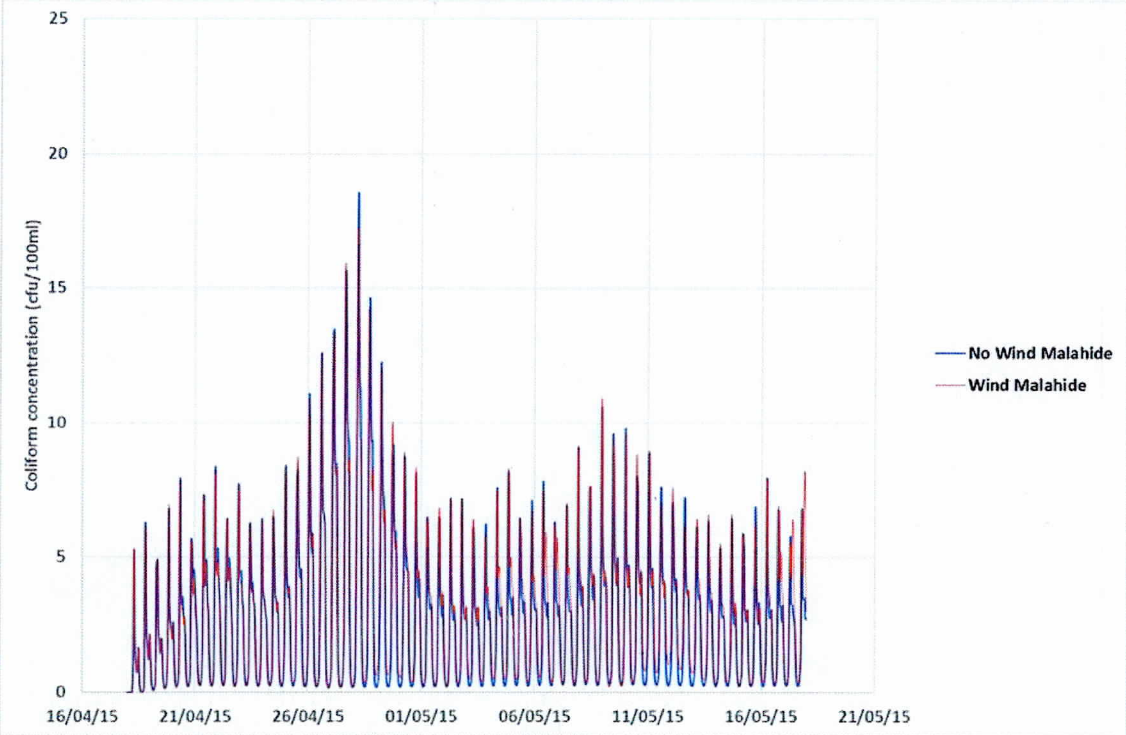

Figure 2: Predicted coliform concentrations at Malahide shellfish sampling point for No Wind and Wind scenarios.

There is no appreciable difference in predicted coliform concentrations between the No Wind, and Wind scenarios. The predicted concentrations were analysed statistically to determine compliance with the proposed "All Species" geometric mean concentration on coliforms in the water column of 1.4. The results from the statistical analysis for the two scenarios are presented in the table below, along with the estimated statistics for a discharge at constant ADF of 1.63 m3/s with no wind defined.

	No Wind	Wind	ADF No Wind	
Geometric Mean	1.49	1.76	1.16	*
90%ile	6.46	6.60	6.32	*

The geometric means calculated for both scenarios (No Wind [1.49], and Wind [1.76]) are greater than the "All Species" value of 1.4. It is suggested the reason for this is the character of the synthesised flow rate shown in Figure 1 with peak flows at Flow to Full Treatment levels resulting in increased mass of coliforms discharging through the outfall.

Five locations along the southern edge of the designated shellfish waters were also examined, both statistically and as a timeseries plots. The position of the five locations are presented in Figure 3, below.

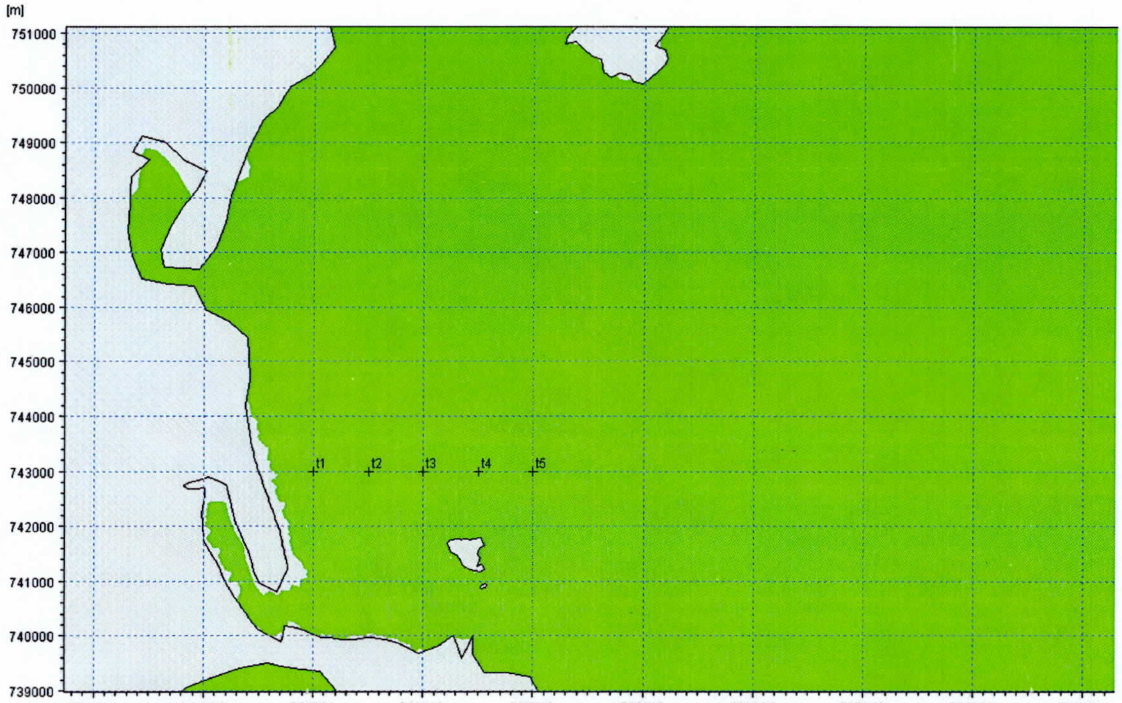

Figure 3: Position of the 5 locations across southern shellfish boundary.

The evolution over time of the predicted coliform concentrations is presented in Figure 4 and Figure 5 for the No Wind, and Wind scenarios respectively.

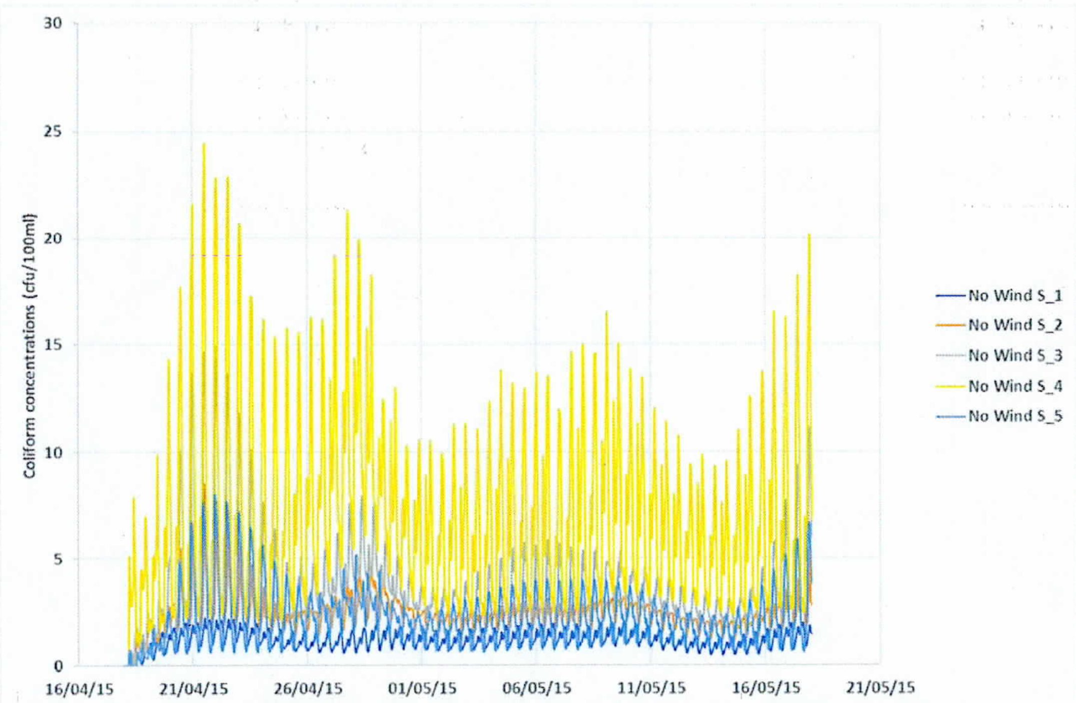

Figure 4: Coliform concentrations at 5 locations along southern Shellfish designation (No Wind)

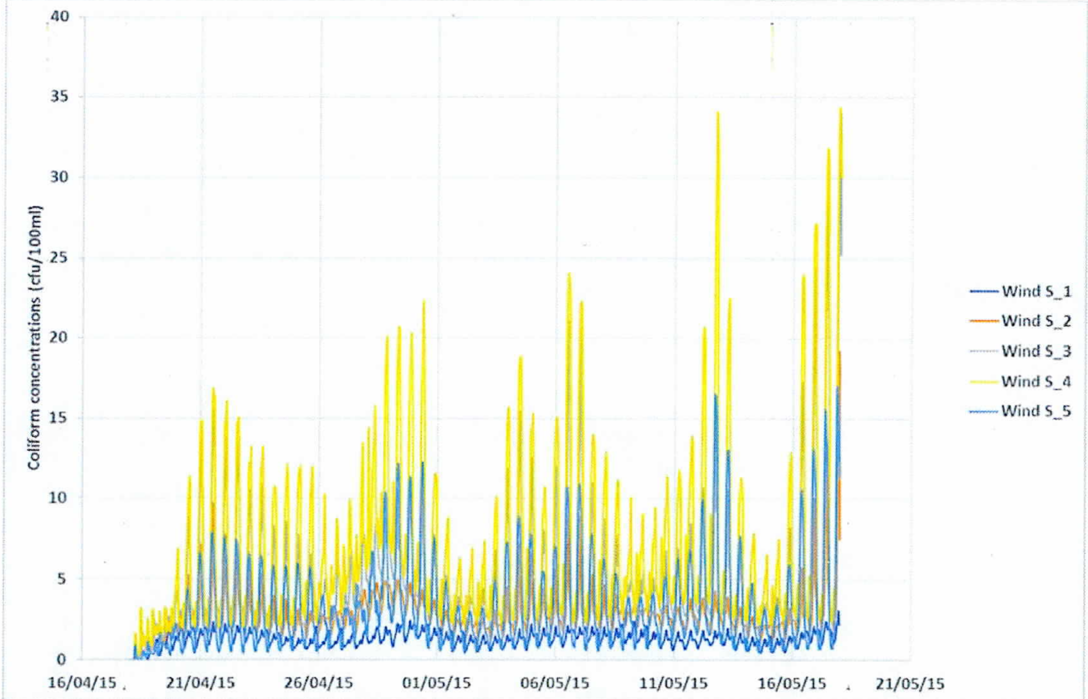

Figure 5: Coliform concentrations at 5 locations along southern Shellfish designation (No Wind)

Both the above Figures show that highest coliform concentrations predicted at Location S_4 just to the northwest of the outfall. During the Wind scenario, locations S3 and S_5 are also predicted to experience higher than normal concentrations.

The statistical assessment of both scenarios at the 5 locations along the southern boundary of the designated shellfish waters are presented in the tables below.

	Synthesised Flows @20,000 cfu/100ml (No Wind)					
	SMP	S_1	S_2	S_3	S_4	S_5
Geometric Mean	1.49	1.22	2.41	3.49	6.03	2.01
90%ile	6.46	1.79	3.14	5.48	12.97	3.89

	Synthesised Flows @20,000 cfu/100ml (with Wind)					
	SMP	S_1	S_2	S_3	S_4	S_5
Geometric Mean	1.76	1.34	2.76	4.35	5.78	2.65
90%ile	6.60	1.99	4.31	8.88	14.86	7.57

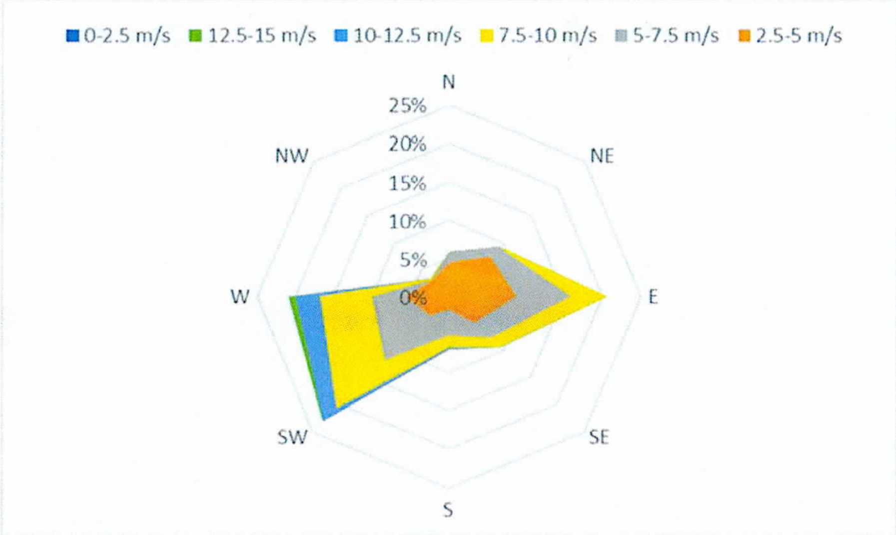

Figure 6: Dublin Airport windrose (18/04/2015 – 18/05/2015)

Impact on Bathing Waters

The results were analysed at the designated bathing water sampling points on Portmarnock Velvet Strand and Claremont Beach and presented in Figure 7 and Figure 8 respectively.

Predicted concentrations of coliforms at Portmarnock Velvet Strand were very low and show little variation between the NoWind and Wind scenarios.

Predicted concentrations of coliforms at Claremont were low and but showed significant variation between the NoWind and Wind scenarios, with the Wind scenario predicting increased coliform concentrations following periods of easterly winds. This would be expected given the beach's location with respect to the proposed outfall location.

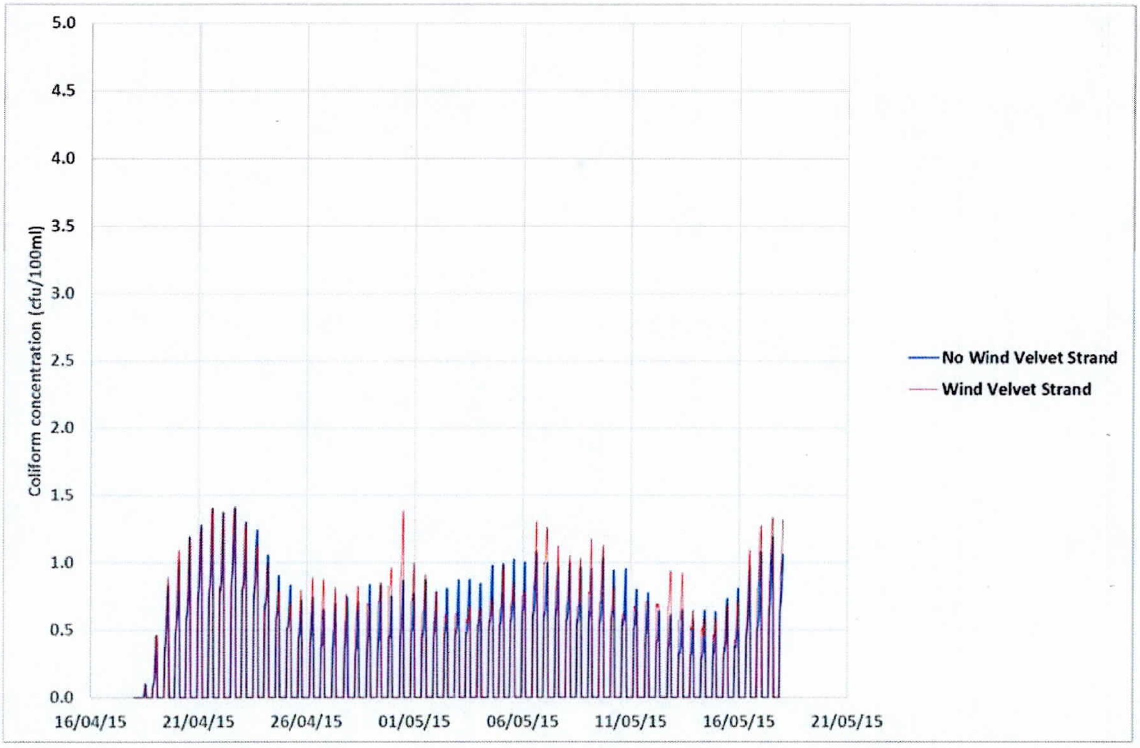

Figure 7: Predicted coliform concentrations at Portmarnock Velvet Strand for both scenarios.

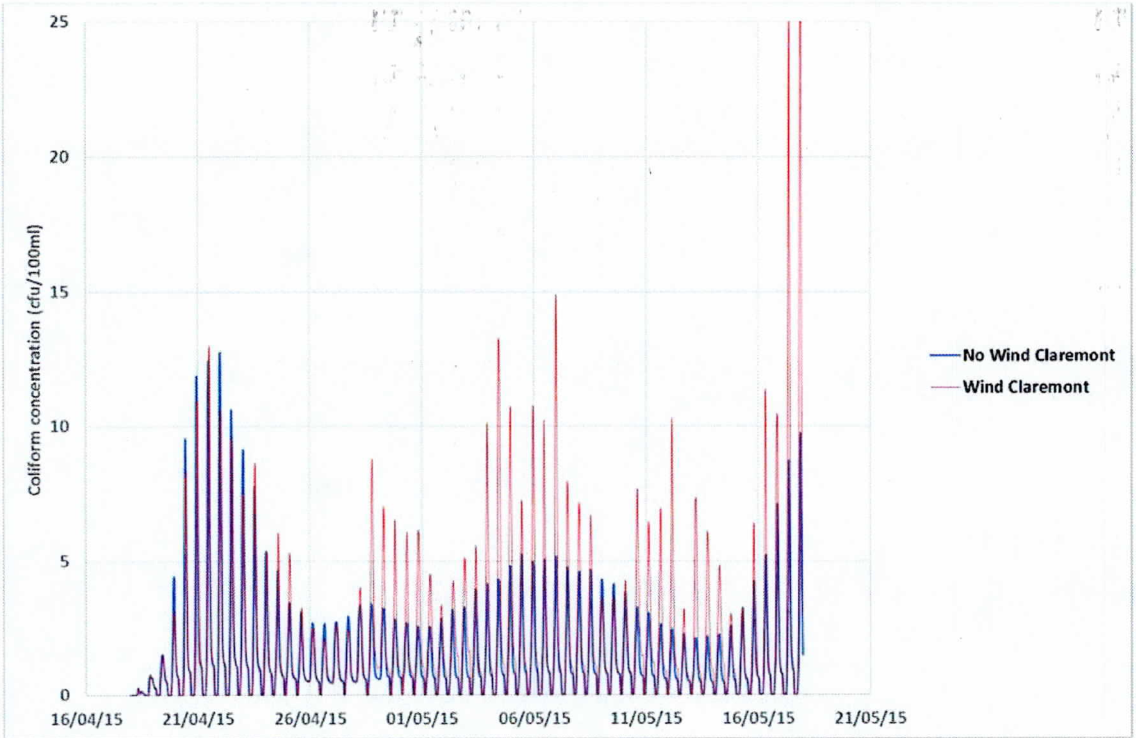

Figure 8: Predicted coliform concentrations at Claremont Beach for both scenarios.

4-4

Revised / Updated
Malahide Pollution Reduction Programme

Malahide, County Dublin

General Characteristics Map

Designated Shellfish Area	**No. 32 - Malahide, County Dublin**
Area of Designated Shellfish Area	**36.31 km2**
Area of Catchment	**376.66 km2**

Shellfish Area
Sampling Point

0 0.4 0.8 1.2 1.6 km

Copyright Ordnance Survey Ireland and Government of Ireland OSI licence No.

Name	**Malahide Shellfish Area**
Map number	32
Year of designation	2009
Area	36.3 km^2
River Basin District	Eastern RBD
County	Dublin
Location of sampling point	53 deg 27.394 min North (Lat) 6 deg 4.457 min West (Long)
Catchment area	376.66 km^2
Catchment area within 20 km zone	317.96 km^2

1.0 INTRODUCTION

1.1 Programme Objective

Compliance with the standards and objectives established by the Quality of Shellfish Waters Regulations 2006 (S.I. No. 268 of 2006) (as amended) for the designated shellfish growing waters at Malahide and with Article 5 of Directive 2006/113/EC of the European parliament and of the Council on the quality required for shellfish waters.

1.2 Pollution Reduction Programme

This pollution reduction programme for the shellfish growing waters at Malahide has been established by the Minister for the Environment, Community and Local Government in order to protect and improve water quality in the designated shellfish growing areas in Malahide and in particular, to ensure compliance with the standards and objectives for these waters established by the 2006 Quality of Shellfish Waters Regulations (S.I. No. 268 of 2006) and with Article 5 of Directive 2006/113/EC of the European parliament and of the Council on the quality required for shellfish waters.

1.3 Supporting Characterisation Report and Toolkit of Measures

The Pollution Reduction Programme stems from the work undertaken in the characterisation report for Malahide. The characterisation is designed to achieve the following:

- establish the catchment that influences the water quality of the designated area;
- identify the different types of pressures or impacts prevalent in the catchment;
- establish an initial assessment of the water quality within the catchment and within the designated shellfish area using all water quality data available;
- from the above three elements identify the pressures that are active in the catchment and subsequently impacting the water quality in the designated shellfish area;
- having identified the pressures impacting on the water quality the characterisation report prioritises them in relation to their impact.

The characterisation report thus provides a prioritised list of pressures/impacts/effects on water quality. The pollution reduction programme or action plan takes this prioritised list and addresses each issue with actions to help ensure that compliance with the relevant water quality standards is achieved or ensured.

The measures/actions included in this PRP to address the identified pressures on shellfish water quality in this catchment are based on a National Toolkit of Measures. The National Toolkit has been derived from earlier work carried out on the River Basin Management Plans under the Water Framework Directive (WFD), reflecting the common objective to improve water quality in the two Directives. In addition, designated shellfish waters are part of the WFD Register of Protected Areas, providing a further link between the Pollution Reduction Programmes and River Basin Management Planning.

Within each individual PRP specific measures from the National Toolkit are applied, where required, to address the key and secondary pressures identified in each of the designated shellfish waters.

1.4 Strategic Environmental Assessment and Habitats Directive Assessment

The Strategic Environmental Assessment (SEA) and Habitats Directive Assessment (HDA) processes were carried out in tandem with the PRP compilation process. These assessments both informed the development of alternatives considered for the PRP and included detailed high-level assessments highlighting the potential positive and negative impacts (including cumulative impacts) associated with application of the measures contained in the National Toolkit. In addition, a more focussed assessment was also carried out which considered the individual and cumulative impacts associated with implementation of the measures brought forward into this individual PRP.

As a result of the SEA and HDA assessments mitigation measures were identified in order to reduce potential negative impacts associated with implementation of the PRP. The relevant mitigation measures are included in Annex 2 of the PRP. The mitigation measures arising from the SEA are noted in black, while the mitigation measures arising from the HDA noted in blue.

1.5 Monitoring of Water Quality

The Marine Institute is carrying out a monitoring programme to monitor the condition of waters in the shellfish growing area and to verify compliance, or otherwise with the water quality standards outlined in Schedules 2 and 4 of the Quality of Shellfish Waters Regulations (S.I. No. 268 of 2006) and summarised in Table 1 of the Characterisation Report (Chapter 1 of the Characterisation Report refers). The Marine Institute will submit a report on water quality in respect of the designated area to the Minister each year, and will immediately bring to the attention of the Department of the Environment, Community and Local Government any non-compliance with a water quality standard to enable investigation to be undertaken.

1.6 Review/monitoring of Pollution Reduction Programme

This pollution reduction programme will be kept under review by the Minister and will be updated and amended as needed from time to time, having regard to water quality conditions within the shellfish growing area including changes in water quality in response to the implementation of measures and other factors arising in the catchment that may affect water quality in the designated area.

The pollution reduction programme will be reviewed at intervals not exceeding three years and, where necessary, at lesser intervals if the monitoring data indicates a deterioration in water quality status or a risk that the objectives or standards laid down in the Regulations will not be achieved.

When the Pollution Reduction Programme is being reviewed the most current baseline data will be consulted.

Prior to the incorporation of the PRP into the second cycle of the River Basin Management Plans a review of the Strategic Environmental Objectives for Water will be carried out as against those drawn up for assessment of the first cycle River Basin Management Plans to ensure that the Shellfish PRP help to meet the wider Water Framework Directive water quality objectives.

1.7 Monitoring of Environmental Impacts

Article 10 of the SEA Directive requires that monitoring be carried out in order to identify at an early stage any unforeseen adverse effects due to implementation of the PRP, with the view to taking remedial action where adverse effects are identified through monitoring. An Environmental Monitoring Programme has been developed which focuses on aspects of the environment that are likely to be impacted by the PRPs. The Environmental Monitoring Programme is included in Table 5 of the National Toolkit of Measures. The Department of the Environment, Community and Local Government will be the authority responsible for collecting and collating data under the Environmental Monitoring Programme. The data will be collected at the same time the pollution reduction programme is reviewed.

1.8 Monitoring Implementation of Pollution Reduction Programme

This PRP is effectively a sub-basin plan of the River Basin Management Plan for the catchment and will be implemented during the first implementation cycle under the Water Framework Directive (i.e up to 2015).

Implementation of the pollution reduction programme will be monitored by Water Quality Section of the Department of the Environment, Community and Local Government.

The contact person is:

Mr. Aidan Brennan
Assistant Principal
Water Quality Section
Department of the Environment, Community and Local Government,
Newtown Road
Wexford.

Phone No: 053 9117466 (+00 353 53 9117466)
Fax No: 053 9117603 (+00 353 53 9117603)
Email: aidan.brennan@environ.ie

2.0 STATUS/IMPACTS

Overall status	The results of monitoring (2009) undertaken for the purposes of the Shellfish Waters Directive (2006/113/EC) and Schedules 2 and 4 of the Quality of Shellfish Waters Regulations (S.I. No. 268 of 2006) indicated faecal contamination within / in the vicinity of this shellfish area. The most up to date results of monitoring (2012) indicate that this area is in compliance with the Guide Value of 300 faecal coliforms / 100ml. However due to the previous indication it is prudent to continue with the actions outlined in this Pollution Reduction Programme. The results of Shellfish Water Monitoring indicate that there are no water quality issues within / in the vicinity of this shellfish area. Monitoring of shellfish flesh for food hygiene purposes

	(2012) indicates faecal contamination in this shellfish area. The bivalve mollusc production areas in Malahide are classified as 'Class B' for the purposes of EC Regulation 854/2004. However, the available shellfish monitoring at this site is in compliance with the shellfish guideline value for faecal coliforms as indicated above. Chapter 3 of the Characterisation Report refers.
Other issues	None
3.0 PRESSURES/RISKS	
3.1 Key Pressures	Analysis of the Characterisation Report for this designated shellfish water suggests that the key pressures are urban wastewater systems and on-site waste water treatment systems. Chapter 5 (summary at 5.3) of the Characterisation Report refers.
Urban wastewater systems	Malahide Portrane/Donabate Swords See Annex 1
On-site waste water treatment systems	There are 6,500 on-site waste water treatment systems in this catchment and their density is higher than the national average. The characterisation report indicates that a substantially smaller number are located within the coastal region of the catchment, which may have a direct impact on the shellfish area. The characterisation report also indicates that the hydrological condition of the majority of the catchment poses a risk to surface waters, the risk to surface waters from pathogens is high throughout the catchment as is the likelihood of inadequate percolation. In response to measures identified in the Pollution Reduction Programme to address OSWWTS pressures in the vicinity of the designated shellfish area Fingal County Council • Have carried out an assessment of the risk to the microbiological quality of shellfish from effluent discharges in April 2011. • Have prepared and submitted to the EPA a report on "Portrane/Donabate Agglomeration – Assessment of the Impact on Shellfish Waters, Wastewater Discharge Licence No: D0114-01 • have conducted a rural house count in 2011 compiling information on septic tanks and OSWWTS at domestic premises. This will be completed in 2012 and a risk assessment on

	the impact of OSWWTS can then be carried out. • are currently updating records on the drainage system in Fingal . • have introduced an Fats, Oil & Grease Licencing Programme • are carrying out reviews of all Trade Effluent Discharges • are carrying out investigative monitoring of rivers in the catchment • carried out an information campaign comprising of a leaflet drop • identified a measures /enforcement programme to be implemented under the Water Pollution Act and Section 70 of the Water Services Act The European Court of Justice has ruled against Ireland in relation to on-site wastewater treatment systems (ref. Case C-188/08). The Court found that by failing to adopt the necessary legislation to comply with Articles 4 and 8 of Council Directive 75/442/EEC as regards domestic waste waters disposed of in the countryside through septic tanks and other individual waste water treatment systems, Ireland has failed to fulfil its obligations under that directive. To address the ruling, the Water Services (Amendment) Act 2012 was signed by the President on 02/02/2012. This Act introduces a new system of registration and inspection for septic tanks and other on-site waste water treatment systems. The Act also sets out the responsibilities of households served by those systems (including requirements to carry out remedial actions where necessary).
3.2 Potential Secondary Pressures	Agriculture
Agriculture	Estimates of fertiliser usage are higher than the national averages. Areas of wet soil types in the catchment mean that there is a potential risk of agricultural runoff. In response to measures identified in the Pollution Reduction Programme to address Agricultural pressures in the vicinity of the designated shellfish area Fingal County Council has engaged with consultants to carry out farm inspections in the Turvey River catchment in Donabate. 10 farm inspections have been carried out to date. In addition to this • an information campaign was carried out in this Shellfish Catchment area which comprised of a leaflet drop and an informal session with the landowner. • identified a measures /enforcement programme

	to be implemented under the Water Pollution Act and Section70 of the Water Services Act
4.0 PROTECTED AREAS	
Designated Shellfish Areas	Malahide designated Shellfish Waters

5.0 ACTION PROGRAMME – MEASURES	
5.1 Key Pressures	
Urban Wastewater Systems	**<u>Overview:</u>** A system for the licensing or certification by the EPA of waste water discharges from areas served by local authority sewer networks was established in accordance with the requirements of the Waste Water Discharge (Authorisation) Regulations, 2007 (S.I. No. 684 of 2007). In accordance with these Regulations the EPA is not allowed to grant an authorisation for a waste water discharge, which, in the opinion of the EPA, would: • cause a deterioration in the chemical status or ecological status (or ecological potential as the case may be) in the receiving body of surface water, • exclude or compromise the achievement of the objectives established for protected species and natural habitats in the case of European sites where the maintenance or improvement of the status of water is an important factor in their protection or which is inconsistent with the achievement of environmental quality standards established under national Regulations in relation to designated bathing waters, designated shellfish waters, areas designated for the protection of freshwater fish and areas designated for the abstraction of water intended for human consumption. The requirements of the European Communities (Quality of Shellfish Waters) Regulations, 2006 (as amended) have been fully integrated into the EPA licensing process In addition this process takes into account the effect of viruses on the quality of shellfish waters. The licence will require detailed actions including infrastructural works, if required, by the licensee within specified time-frames if the discharge does not comply with the above Regulations. Each licence granted will be subject to enforcement by the EPA. Full details of each application and licence decision can be viewed online at www.epa.ie. The following is the position with the key waste water treatment plants for Malahide: <u>Malahide</u> - A Waste Water Discharge Licence was granted in respect of Malahide in March 2011 to Fingal County Council pursuant to the requirements of the Waste Water Discharge (Authorisation) Regulations, 2007(as amended). The Local Authority must comply with the conditions as set out in the Licence and in particular sections 5.6, 5.7 and 5.8 with regard to impact of Discharge, possible need for disinfection treatment and notification of incident to specified authorities. <u>Portrane/Donabate</u> - secondary treatment WWTP in place The EPA issued a waste water discharge licence on the 30th of October 2009. Conditions 5.6 and 5.7 of the licence state that a

	microbiological quality assessment of the shellfish in the designated shellfish area shall be carried out by April 2011(this time period is required to allow a comprehensive study to be completed) This assessment was carried out by FCC in April 2011 and the report was prepared and submitted to EPA. <u>Swords</u> - secondary treatment plus nutrient removal WWTP in place. A licence application was made by Fingal County Council in December 2007 pursuant to the requirements of the Waste Water Discharge (Authorisation) Regulations, 2007. This application is currently under assessment. In the cases above, compliance with any EPA Wastewater Discharge Authorisation will require detailed actions, including infrastructural works, if required, by the licensee within specified time-frames if the discharge does not comply with the above Regulations. Each licence granted will be subject to enforcement by the EPA. The financial investments to ensure compliance with any EPA licence conditions requiring additional urban waste water collection or treatment can be made under the Water Services Investment Programme.
On-site waste water treatment systems	Fingal County Council were to identify systems directly adjacent to estuarine and coastal waters and water courses as well as systems serving large populations and to undertake investigation of the likely extent of microbial contamination of Designated Shellfish Waters from adjoining dwellings and Section 4 licensed activities. Section 70 of the Water Services Act 2007 places a duty of care on owners of septic tanks and provides local authorities with enforcement powers including prosecution to address any problems identified. The Report on Possible Risks from On-Site-Wastewater Treatment Systems on Designated Shellfish Water Areas, received from Fingal County Council for the Malahide Designated Shellfish Water Area has been reviewed and it is considered that it would be prudent to implement additional measures as follows to ensure compliance with the Pollution Reduction Programme requirements: • Fingal County Council should take the necessary follow up enforcement action with the occupiers of dwellings where there is risk of untreated effluent entering the designated waters • All new planning applications for dwellings to be served by on-site waste water treatment systems in the Local Authority Area should be required to demonstrate compliance with the EPA Code of Good Practice for Waste Water Treatment & Disposal Systems Serving Single Houses. This will minimise any potential risk of discharge of pathogens to the shellfish water from any new dwelling in the area. • The need for on-site inspections based on the national implementation plan to be drawn up by the EPA should be factored into the overall risked based approach for inspections under the Water Services (Amendment) Act 2012.

	• An advisory leaflet on management of OSWWTS's should be issued to each dwelling inspected in the catchment by Fingal County Council. This will comply with an education mitigation measure included in the SEA which is outlined in the PRP • follow up with the measures/enforcement programme as detailed to ensure compliance with the Pollution Reduction Programme requirements:
5.2 Potential Secondary Pressures	
Agriculture	The Report on Possible Risks from Agriculture on Designated Shellfish Water Areas, received from Fingal County Council and for the Malahide Designated Shellfish Water Area has been reviewed and it is considered that it would be prudent to implement additional measures as follows • ensure effective and targeted implementation of the Good Agricultural Practice Regulations • follow up with the measures/enforcement programme as detailed to ensure compliance with the Pollution Reduction Programme requirements:
Future Development	Under Article 4 of the European Communities (Quality of Shellfish Waters) Regulations 2006 (S.I. No. 286 of 2006) (as amended), every public authority that has functions the performance of which may affect shellfish waters shall perform those functions in a manner that will promote compliance with the objectives of this pollution reduction programme and with the objectives of the Shellfish Waters Directive. The functions of particular importance – in light of the objectives of Directive 2006/113/EC and of this PRP – include waste water treatment (licensing and operations), implementation of the GAP Regulations, waste management (licensing and operations), effluent discharge licences, planning and development and building control. Continued monitoring will be carried out during the lifetime of the PRP. Should this monitoring identify pressures that are impacting on shellfish water quality in the designated area, the PRP will be appropriately amended.

Compliance with the Parameters set out in the Directive[1] The Directive prescribes the minimum ((Mandatory (I)) quality criteria which must be met by shellfish waters and guideline values (G) which Member States must endeavour to observe. Not all of the Parameters have both Guide and Mandatory values.			
		Compliance with Mandatory Values (Y/N)	**Compliance with Guide Values (Y/N)**
Parameter 1	PH (I)	Y	
Parameter 2	Temperature (G)		Y
Parameter 3	Coloration (after filtration) (I)	Y	
Parameter 4	Suspended Solids (I)	Y	
Parameter 5	Salinity (I & G)	Y	Y
Parameter 6	Dissolved Oxygen (I & G)	Y	Y
Parameter 7	Petroleum Hydrocarbons (I)	Y	
Parameter 8	Organohalogens (I & G)	Y	Y
Parameter 9	Trace Metals (I & G)	Y	Y
Parameter 10	Faecal Coliforms (G)		Y

[1] Compliance for Parameters 1 to 7 - taken from 2011 monitoring results
Compliance for Parameters 8 & 9 - taken from 2010 monitoring results
Faecal Coliform compliance – 2012 monitoring results

Annex 1

Water Services Authority	Agglomeration Name	Registration Number	Population Equivalent	Status
Fingal County Council	Malahide	D0021-01	> 10,000	Licensed
Fingal County Council	Portrane/Donabate	D0114-01	2,000- 10,000	Licensed
Fingal County Council	Swords	D0024-01	> 10,000	Under Assessment

Annex 2 - Mitigation Recommendations from the SEA process

The Strategic Environmental Assessment carried out for the Shellfish PRPs has highlighted potential positive and negative environmental impacts (including cumulative impacts) associated with implementation of the range of measures outlined in the National Toolkit of Measures, all of which are aimed at controlling pressures which impact on shellfish water quality.

In most cases, the PRPs identify the need for further investigation to supplement existing information on the types and extent of the pressures which are currently affecting shellfish water quality. Following this, the next step in the protection of shellfish waters will be the introduction of measures from the National Toolkit to address the identified pressures. It should be noted that this PRP is a dynamic document and will be updated regularly in order to outline if, and where, measures are required following the completion of the investigations.

The table below outlines the mitigation measures required to reduce potential impacts from measures in the National Toolkit associated with the key and potential secondary pressures currently identified for this catchment. When considering implementation of specific measures from the National Toolkit, it is required that the relevant mitigation measures below be considered to reduce any potential negative impacts (mitigation measures arising from the Habitats Directive Article 6 Assessment are noted in blue).

Should further key and secondary pressures be identified in this catchment in future, then the full list of mitigation measures, which is included in Table 4 of the National Toolkit, should be consulted to determine if any of those apply. In addition, the authority/organisation/individual responsible for implementing each of the mitigation measures below is listed in Table 4 of the National Toolkit.

	NATIONAL TOOLKIT MEASURE	ASSOCIATED MITIGATION MEASURE
WFD4	**POINT SOURCE & DIFFUSE SOURCE DISCHARGES** Actions: Water Pollution Acts and regulations: • License discharges to surface waters and sewers from small scale industrial and commercial sources. Review licenses at intervals of not less than 3 years. Keep registers of discharge licenses and make them available to the public. • Serve notices or directions on persons requiring measures to be taken in order to prevent or control pollution of waters, where necessary. • Notify Local Authorities of accidental discharges and spillages of polluting materials which enter, or are likely to enter, waters. Other actions: Urban Wastewater Treatment Plants: • Measures for improved management: keep register of plant capacity and update annually; install facilities to monitor influent loads and effluent discharges in accordance with Environmental Protection Agency guidelines and best practice; put auditable procedures in place to monitor compliance of licensed discharges; implement training procedures for staff involved with licensing of discharges; monitor receiving water quality upstream and downstream of the point of discharge. • Optimise treatment plant performance by the implementation of a performance management system. • Revise existing Water Pollution Act industrial licence conditions and reduce allowable pollution loading. • Review existing Industrial Pollution Prevention Control licence conditions and reduce allowable pollution load. • Investigate contributions to the collection system from unlicensed discharges. • Investigate contributions to the collection system of specific substances known to impact ecological status resulting from licensed and unlicensed discharges and issue or revise licenses to reduce or remove such specific substances in the discharge.	Detailed assessment of higher risk works will be required to include environmental considerations (based on EIA guidance). It is recommended that lower risk work should be compelled to consider environmental issues as part of the registration process.

	• Upgrade plant to increase capacity where necessary. • Upgrade plant to provide nutrient removal treatment where necessary. Actions: Wastewater Discharge Authorisation Regulations: • License large Local Authority WWTPs and certify smaller WWTPs as specified in the Regulations (taking account of WFD objectives). Review licenses at intervals not less than 3 years. Enforce compliance with WWTP licensing conditions. Maintain a register of WWTP licences and certificates and make available on request. Inform other relevant public authorities when an application or review is received. Actions: Water Services Act: • Prepare and implement Water Services Strategic Plans. • Duty of care on owners of premises to ensure that treatment systems for wastewater are kept in good condition. Actions: Planning and Development Act (unsewered systems) • Permit on-site waste water treatment systems subject to site suitability assessment. Other actions: Unsewered Systems: • Amend Building Regulations to give effect to new codes of practice for single houses and large systems.	
WW1	**WASTE WATER TREATMENT PLANTS** Measures intended to reduce loading to the treatment plant: • Limit or cease the direct importation of polluting matter (e.g. liquid wastes, landfill leachate, sludges). • Investigate the extent of use and impact of under-sink food waste disintegrators and take appropriate actions. • Investigate fats/oils/grease influent concentrations and take actions to reduce FOG entering the collection system.	This measure should be accompanied by an education and awareness campaign for householders and commercial premises aimed at reducing pollution at source. This campaign should include information on the use and disposal of household chemicals, oils, detergents, paints, solvents, etc as well as information on phosphorus-related pollution. Consideration should also be given to targeting specific audiences on issues such as discharges to water and the importance of wetland sites to water quality. This measure will require project level Habitats Directive Assessment if alternative facilities for treatment of waste are constructed, e.g. incinerator.

WW2	**WASTE WATER TREATMENT PLANTS** Impose development controls where there is, or is likely to be in the future, insufficient capacity at treatment plants.	This measure will need to link to the development planning process, e.g. by including a requirement to address wastewater capacity as part of the scope in any accompanying SEA for development plans. This measure will need to consider whole catchment loading.
WW6 to WW9	**WASTE WATER TREATMENT PLANTS** WW6: Where necessary to achieve water quality objectives install secondary treatment at smaller plants where this level of treatment would not otherwise be required under the urban wastewater treatment regulations. WW7: Apply a higher standard of treatment (stricter emission controls) where necessary. WW8: Upgrade the plant to remove specific substances known to impact on water quality status WW9: Install ultra-violet or similar type treatment.	WWG to WW9: Negative impacts on climate associated with GHG emissions related to additional energy requirements for these measures should be offset by use of renewable energy sources or similar. WW6 to WW9: If these alternatives involve the building of a new plant or an extension to an existing plant a Habitats Directive Assessment will be required. Prior to any proposals for a new plant, further investigation will be required to show that a new plant will have the desired improvements in water quality for which it is being built. WW6 to WW8: If additional landtake is required for these measures, environmental studies will be undertaken to assess the impact on the environment. WW9: A Habitats Directive Assessment will be required prior to introduction of UV or similar treatment when the discharge is within or adjacent to a protected area.
WW10	**WASTE WATER TREATMENT PLANTS** Relocate the point of discharge.	A Habitats Directive Assessment will be required to demonstrate that the relocation will not negatively impact on protected areas.
UP3	**ON-SITE WASTE WATER TREATMENT SYSTEMS** For new developments: • At planning assessment stage, apply the GIS risk mapping / decision support system and codes of practice • Notice to planning authority required immediately prior to the installation of on-site effluent treatment systems including percolation areas and polishing filters.	The pre-planning process should assess whether Habitats Directive Assessment would be required for new development within or adjacent to a protected area.

UP5 to UP7	**ON-SITE WASTE WATER TREATMENT SYSTEMS** UP5: Enforce requirements for percolation. UP6: Enforce requirements for de-sludging. UP7: Consider connection to municipal systems.	UP5 & UP6: An education programme should be carried out in tandem with new requirements for tank maintenance, including guidance on disposal of sludges. UP6: Intelligent transport programmes should be put in place to minimise the amount of emissions associated with movement of sludges from on-site treatment systems. UP7: Upgraded treatment works should be required to introduce BAT, including the use of renewable energy sources, in order to reduce GHG emissions and others resulting from increased demand for treatment. UP6 & UP7: New wastewater treatment infrastructure, including sludge disposal infrastructure, will be subject to environmental assessment at the project level to reduce indirect impacts to biodiversity, landscape, cultural heritage and climate. UP7: A Habitats Directive Assessment will be required for new structures.

**Note: It should be noted that in this case the term Habitats Directive Assessment refers to the assessment process as specified in Article 6 of the Habitats Directive. This starts with screening to determine whether a likely significant impact from the plan/programme is expected to occur to a Natura 2000/Ramsar site as a result of activities in/adjacent to/in the catchment of a Natura 2000/Ramsar site. If, in accordance with Habitats Directive Assessment guidance (guidance produced by the EU and DoEHLG in Ireland), it can be shown that there is no potential for impact at the screening stage, no further assessment may be required. However when the plan/programme being screened lies within or adjacent to a Natura 2000/Ramsar site then such a determination must be made in consultation with NPWS. If the plan/programme is within the catchment (surface and groundwater) of a Natura 2000/Ramsar site, such consultation with NPWS is only necessary for those water dependent Natura 2000 sites which are listed in the WFD Register of Protected Areas.*

Shellfish Pollution Reduction Programme

As required by Article 5 of the Shellfish Water Directive 2006/113/EC and Section 6 of the Quality of Shellfish Waters Regulations, 2006 (S.I. No. 268 of 2006)

Characterisation Report Number 32

MALAHIDE SHELLFISH AREA
COUNTY DUBLIN

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ABBREVIATIONS

AA	Appropriate Assessment
BOD	Biochemical Oxygen Demand
CFB	Central Fisheries Board
CSO	Combined Sewer Overflow
DED	District Electoral Division
DEHLG	Department of Environment Heritage and Local Government
DO	Dissolved Oxygen
EPA	Environmental Protection Agency
EU	European Union
Ha	Hectare
IPPC	Integrated Pollution Prevention Control
Kg	Kilogram
LU	Livestock Units
NACE	European industrial activity classification
MI	Marine Institute
NPWS	National Parks and Wildlife Service
OSWTS	On-Site Waste Water Treatment System
P.E.	Population Equivalent
PRP	Pollution Reduction Programme
RBD	River Basin District
RBMP	River Basin Management Plan
SAC	Special Area of Conservation
SEA	Strategic Environmental Assessment
SFPA	Sea Fisheries Protection Authority
SPA	Special Protection Area
SWMC	Shellfish Waters Management Committee
TCE	Tetrachloroethylene
WFD	Water Framework Directive
WSIP	Water Services Investment Programme
WTP	Water Treatment Plant
WWTP	Waste Water Treatment Plant

1.0 INTRODUCTION

Article 5 of the Shellfish Directive (2006/113/EC) and section 6 of the Quality of Shellfish Waters Regulations (S.I. No. 268 of 2006) require the development of Pollution Reduction Programmes (PRPs) for designated shellfish areas in order to support shellfish life and growth and to contribute to the high quality of directly edible shellfish products. Shellfish PRPs relate to bivalve and gastropod molluscs, including oysters, mussels, cockles, scallops and clams. They do not cover shellfish crustaceans such as crabs, crayfish and lobsters.

1.1 Aims and responsibility

The objectives of Shellfish PRPs are to:

- Protect or improve water quality in designated shellfish areas;
- Achieve compliance with water quality parameter values outlined in Annex I of the Shellfish Waters Directive (2006/113/EC) and Schedules 2 and 4 of the Quality of Shellfish Waters Regulations (S.I. No. 268 of 2006);
- Determine the factors responsible for any non-compliances with the water quality parameter values; and
- Ensure that implementation of the Shellfish PRPs does not lead, directly, or indirectly, to increased pollution of coastal and brackish waters.

Under the Regulations, the Department of Communications, Marine and Natural Resources is responsible for the development of Shellfish PRPs. However, this responsibility was transferred to the Department of the Environment, Heritage and Local Government (DEHLG) on 5th November 2008. An Inter-Departmental /Inter Agency Shellfish Waters Management Committee (SWMC) supports the Department in the development of the Shellfish PRPs.

The Regulations also place an obligation on every public authority to perform its functions in a manner that promotes compliance with the Directive and the Regulations, and to take such actions as are necessary to secure compliance with the Directive and the Regulations and with the Shellfish PRPs.

1.2 Shellfish water quality parameters

Compliance with the directive is measured against achievement of shellfish water quality parameter values outlined in Annex I of the Shellfish Waters Directive (2006/113/EC) and Schedules 2 and 4 of the Quality of Shellfish Waters Regulations (S.I. No. 268 of 2006). Table 1 summarizes these values. Mandatory (I) values must be fully achieved while it must be endeavoured to achieve guideline values (G).

TABLE 1 - Parameters listed in Annex I of the Shellfish Water Directive

Physical	Guideline Values (G)	Mandatory Values (I)
pH (pH units)		7 – 9 pH units
Temperature (°C)	A discharge affecting shellfish waters must not cause the	No mandatory value set in the Directive

	temperature of the waters to exceed by more than 2°C the temperature of waters not so affected	
Colouration (after filtration) (mg Pt/l)		A discharge affecting shellfish waters must not cause the colour of the waters after filtration to deviate by more than 10 mg Pt/l from the colour of unaffected waters
Suspended Solids (mg/l)		A discharge affecting shellfish waters must not cause the suspended solid content of the waters to exceed the content in unaffected waters by more than 30%
Salinity (%)	12 to 38%	≤ 40% A discharge affecting shellfish waters must not cause their salinity to exceed the salinity of unaffected waters by more than 10%
Chemical	**Guideline Values (G)**	**Mandatory Values (I)**
Dissolved oxygen (Saturation %)	≥ 80%	≥ 70% Should an individual measurement indicate a value lower than 70%, measurements shall be repeated An individual measurement may only indicate a value of less than 60% if there are no harmful consequences for the development of shellfish colonies
Petroleum hydrocarbons		Hydrocarbons must not be present in the shellfish water in such quantities as to: - produce a visible film on the surface of the water and/or a deposit on the shellfish - have harmful effects on the shellfish
Organohalogenated substances	The concentration of each substance in shellfish flesh must be so limited that it contributes in accordance with Article 1 (of the Directive), to the high quality of shellfish products	The concentration of each substance in the shellfish water or in shellfish flesh must not reach or exceed a level which has harmful effects on the shellfish larvae
Metals (Ag, As, Cd, Cr, Cu, Hg, Ni, Pb and Zn) (mg/L)	The concentration of each substance in shellfish flesh must be so limited that it contributes in accordance with Article 1 (of the Directive), to the high quality of shellfish products	The concentration of each substance in the shellfish water or in the shellfish flesh must not exceed a level which gives rise to harmful effects on the shellfish and their larvae The synergic effects of these metals must be taken into consideration
Others	**Guide Values (G)**	**Mandatory Values (I)**
Faecal coliforms (per 100 mL)	≤ 300 per 100 mL in the shellfish flesh and intervalvular liquid	No mandatory value set in the Directive

Substances affecting the taste of shellfish		Concentration lower than liable to impair the taste of the shellfish
Saxitoxin (produced by dinoflagellates)	No limit given	No limit given

1.3 Designated shellfish areas

Fourteen shellfish areas were originally designated in 1994 under the Quality of Shellfish Waters Regulations (S.I. No. 200 of 1994, revoked by S.I. No. 268 of 2006). A further 49 areas were subsequently designated in 2009 under the European Communities (Quality of Shellfish Waters) (Amendment) Regulations, 2009 (S.I. No. 55 of 2009). All 63 designated sites are illustrated in Figure 1 below.

Note: Map numbers I to XIV refer to waters originally designated under the European Communities (Quality of Shellfish Waters) Regulations 2004 (S.I. No. 200 of 1994), while map numbers 1 to 45 refer to waters designated under the European Communities (Quality of Shellfish Waters) (Amendment) Regulations 2009 (S.I. 55 of 2009). The referenced maps can be found in the relevant regulatory documents.

FIGURE 1 - 63 designated shellfish areas

1.4 Development of Shellfish Pollution Reduction Programmes

The Directive and Regulations require that any non-compliances with the shellfish water quality parameter values are identified. The Directive and Regulations further require that the factors responsible for such non-compliances are identified.

Information on impacts and pressures has therefore been collated in an individual characterisation report for each shellfish site from available inventories. The likelihood of the pressures to impact on shellfish water quality parameter values in the shellfish areas has been estimated.

Individual site Pollution Reduction Programmes (PRPs) and a supporting toolkit of measures outline the measures which can be used to control pressures where necessary to protect and improve water quality in a specific shellfish area.

The 2009 Shellfish PRPs (including the supporting characterisation reports and toolkit of measures) represent an initial phase of Shellfish PRP development, drawing on available information sources. Their development has been a desk-based exercise and they provide a good indication of the main pressures likely to be impacting on shellfish water quality and the measures that can be used to control those pressures. Ongoing assessment and monitoring of shellfish waters will be used to confirm the effectiveness of these programmes and to refine the programmes where necessary. As the shellfish monitoring database grows, and as programmes are implemented, incremental changes will be made to ensure compliance with the standards and objectives established.

PRPs produced during 2009 supersede Action Programmes which were developed in 2006 for the 14 original shellfish areas.

1.5 Assessment of Shellfish Pollution Reduction Programmes

A Strategic Environmental Assessment (SEA) of the Shellfish PRPs and supporting toolkit of measures has been carried out in accordance with the requirements of the EU Strategic Environmental Assessment Directive (2001/42/EC). SEA is a process for evaluating, at the earliest appropriate stage, all of the possible environmental effects of plans or programmes before they are adopted while giving the public and other interested parties an opportunity to comment and to be kept informed of decisions and how they were made. The assessment of the PRPs resulted in mitigation of some of the measures contained in the PRPs and toolkit of measures that were identified as likely to lead to adverse effects on other aspects of the environment. The reports associated with the SEA process can be downloaded from www.environ.ie.

An 'Appropriate Assessment' of the Shellfish PRPs has been carried out in parallel with the SEA assessment in accordance with the requirements of the EU Habitats Directive (92/43/EEC). Appropriate Assessment is a process for evaluating the implications of plans or programmes for sites which have been designated for the protection and conservation of habitats and species of European importance. The reports associated with the Appropriate Assessment can be downloaded from www.environ.ie.

1.6 Links with the River Basin Management Plans

The EU Water Framework Directive (2000/60/EC) provides a framework for the protection and restoration of the aquatic environment and terrestrial ecosystems and wetlands directly depending on the aquatic environment. In accordance with the requirements of the directive, River Basin Management Plans (RBMPs) were published in draft form in December 2008 with the final RBMPs published in December 2009. They are the primary plans in place in relation to the water environment for the foreseeable future.

Article 13(5) of the WFD states that 'river basin management plans may be supplemented by the production of more detailed programmes and management plans for sub-basin, sector, issue, or water type, to deal with particular aspects of water management'. Shellfish PRPs are an example of such programmes. In addition, Article 13(4) and Annex VII of the WFD requires that RBMPs include 'a register of any more detailed programmes and management plans for the River Basin District dealing with particular sub-basins, sectors, issues or water types, together with a summary of their contents'. The Shellfish PRPs are included in the registers of each of the River Basin Districts.

Articles 4 (1)(c) and 4 (2) of the WFD specify that, in relation to protected areas, where more than one of set of objectives relate to a given body of water, the most stringent shall apply. Designated shellfish areas are included in the WFD register of protected areas provided for in Articles 6 and 7 of the directive.

The WFD strengthens and consolidates a number of existing environmental directives while repealing others on a phased basis. The Shellfish Directive is due to be repealed by the WFD in 2013. Shellfish PRPs are therefore closely aligned with the RBMPs.

1.7 Layout of the Shellfish Pollution Reduction Programmes

Characterisation Report

- **Section 1**
  Section 1 is an introductory section which puts the Characterisation Reports in context and outlines their contents.

- **Section 2**
  Section 2 describes the general characteristics of the designated shellfish areas as well as their contributing catchments.

- **Section 3**
  Section 3 describes water quality in the designated shellfish areas.

- **Section 4**
  Section 4 consists of a series of maps illustrating the general characteristics of the shellfish areas and catchments, as well as the marine and land-based pressures in the catchments.

- **Section 5**
  Section 5 provides a series of tables summarising the marine and land-based pressures in the catchments. The likelihood of the pressures to impact on shellfish water quality parameters is discussed. A summary is also provided highlighting the key pressures and potential secondary pressures which are most likely to be impacting on shellfish water quality parameters. The discussions in this section draw on available information including information generated during the WFD implementation process and geographical features of significance. The differing nature of the pressures are also taken into account as pressures vary substantially in terms of how severely they are likely to impact on shellfish water quality parameters.

Pollution Reduction Programmes

- The Pollution Reduction Programmes summarise the specific measures for controlling the key and potential secondary pressures, identified in this characterisation report, which are most likely to be impacting on shellfish water quality in Malahide shellfish area. This can be downloaded from www.environ.ie.

Toolkit of Measures

- The supporting toolkit of measures outlines all of the measures available for controlling all of the pressures which can impact on shellfish water quality. Due to the close alignments between the Shellfish PRPs and the RBMPs, the toolkit is drawn from the programme of measures contained within the RBMPs. This strengthens the integration of shellfish management and wider water quality management policy in Ireland. The toolkit can be downloaded from www.environ.ie.

2.0 GENERAL CHARACTERISTICS

Name	Malahide Shellfish Area
Map number	32
Year of designation	2009
Area	36.3 km^2
River Basin District	Eastern IRBD
County	Dublin
Location of sampling point	53 deg 27.394 min North (Lat) 6 deg 4.457 min West (Long)
Catchment area	376.66 km^2
Catchment area within 20 km zone	317.96 km^2

Malahide is situated in County Dublin in the Eastern River Basin District (Map 1). The designated shellfish area is 36.3 km^2 in area and extends from Lambay Island to Portmarnock. Balbriggan/Skerries shellfish area is situated in adjacent tidal waters.

The contributing catchment is 376.66 km^2 in area (Map 3) and drains number of rivers including the Broadmeadow and the Ward.

Swords is the largest urban centre in the catchment with a population of 27,175. There are also a number of the other large towns including Malahide, with a population of 13,824, Portmarnock, with a population of 8,376, Rush, with a population of 6,769, and Asbourne, with a population of 6,362. The Greater Dublin area is home to 90% of the Eastern River Basin District's population while most of the urban population outside this area is centred round rivers or ports. Farming accounts for 75% of the land use within the catchment.

2.1 Protected areas

The designated shellfish area lies within Malahide candidate SAC (Map 11). Other SACs which intersect the shellfish area's catchment are Baldoyle Bay, Howth Head, Lambay Island, Rogerstown Estuary and Ireland's Eye. Recreational waters include Rush, Portrane, Sutton, Donabate, Portmarnock and Malahide. Nutrient sensitive areas include the Broadmeadow Estuary. Ramsar sites include Baldoyle Bay and the Broadmeadow and Rogerstown estuaries. SPAs include Baldoyle, the Broadmeadow Estuary, Howth Head, Ireland's Eye, Lambay Island and Rogerstown.

2.2 Shellfish growing activity

The cultivation or razor clams is predominant in the area.

3.0 WATER QUALITY IN THE SHELLFISH AREA

Dedicated shellfish monitoring data has been collated and compared with shellfish water quality parameter mandatory and guideline values outlined in Annex I of the Shellfish Waters Directive (2006/113/EC) and Schedule 2 and 4 of the Quality of Shellfish Waters Regulations (S.I. No. 268 of 2006) (Table 1).

Additional monitoring data from other monitoring programmes has also been collated in order to highlight any water quality issues in the vicinity of the shellfish areas. This can aid in the identification of the pressures most likely to impact on the shellfish areas and thereby in the identification of any measures to be applied. Datasets were collated from the Environmental Protection Agency (EPA), the Marine Institute (MI) and the Sea Fisheries Protection Authority (SFPA). Where applicable these additional monitoring data were compared with the shellfish water quality parameter mandatory and guideline values outlined in Annex I of the Shellfish Waters Directive (2006/113/EC) and Schedules 2 and 4 of the Quality of Shellfish Waters Regulations (S.I. No. 268 of 2006) (Table 1).

Marine Institute Shellfish Monitoring Programme

The MI carries out shellfish monitoring at designated shellfish areas. This dedicated shellfish monitoring programme involves analysing for general components, metals and organics in both water and biota samples. The results have been compared with the shellfish mandatory and guideline values outlined in Table 1.

For this designated area there are no water MI water samples available but there was one biota sample available for 2008. The shellfish guidelines values outlined in Table 1 were not breached in this sample.

Faecal coliform biota results were also available from the MI at all shellfish areas from November 2008, February 2009, May 2009 and August 2009. The shellfish guideline value for faecal coliforms in biota outlined in Table 1 was breached in the May 2009 sample.

EPA Marine Monitoring Programme

The EPA Marine Monitoring Programme analyses for general components in water samples at a large number of marine sites around Ireland.

There is 1 EPA site located in the designated area with monitoring data available from the period 2006 to 2008 for pH and dissolved oxygen. The values outlined in Table 1 for these parameters were not breached in the samples at the designated site.

WFD Monitoring Programme

WFD status classifications from the WFD monitoring programme were used as indicators of compliance with shellfish water quality parameter values. WFD status classifications are based on a variety of parameters including biological, physico-chemical, chemical and hydromorphological elements. The monitoring information on which the marine status classifications are based was collected by the EPA, the

Marine Institute, the National Parks and Wildlife Service (NPWS) and the Central Fisheries Board between 2005 and 2008.

The WFD status of the coastal water body, within which the shellfish area is situated, is 'moderate' and therefore unsatisfactory, reflecting unsatisfactory dissolved organic nitrogen levels. The Broadmeadow transitional water, which flows into the designated area, is also 'moderate', reflecting the results of some of the general components and phytoplankton samples (Map 12).

Shellfish Flesh Monitoring Programme

Shellfish flesh classifications (carried out under the European Communities (Live Bivalve Molluscs) (Health Conditions for Production and Placing on the Market) Regulations, 1996 (S.I. No. 147 of 1996)) were also used as indicators of faecal contamination in shellfish. Sampling is carried out by the Sea Fisheries Protection Authority (SFPA) on at least a monthly basis

The licensed area within Malahide is classified as Class B meaning that shellfish may be placed on the market for human consumption only after treatment in a purification centre or after relaying so as to meet the health standards for live bivalve molluscs laid down in the EC Regulation on food safety (Regulation (EC) No 853/2004). This indicates faecal contamination in this shellfish area.

Overall Water Quality

The dedicated shellfish samples available for this shellfish area indicated a non-compliace with the shellfish guideline value for faecal coliforms outlined in Annex I of the Shellfish Waters Directive (2006/113/EC) and Schedule 4 of the Quality of Shellfish Waters Regulations (S.I. No. 268 of 2006) (Table 1). Ongoing shellfish monitoring will strengthen the assessment of compliance status at this shellfish area.

The results of the WFD monitoring programme indicate that there are water quality issues within the area and in some of the waters discharging in the vicinity of this shellfish area.

The shellfish flesh classification indicates faecal contamination in the shellfish area.

4.0 CHARACTERISATION MAPS

The following series of maps illustrate the general characteristics of the designated shellfish area and its contributing catchment, as well as the marine and land-based pressures that could potentially impact on the shellfish area. The pressures are further divided into point source pressures, diffuse source pressures and morphological pressures.

Some of the point source pressures are symbolised according to whether they are 'at risk' or 'not at risk'. These risk designations were developed during the WFD implementation process. Some of the designations date back to the Article V characterisation process in 2004 and 2005 but many of the risk designations were updated in 2008 to feed into the draft RBMPs. The risk designations are based on a variety of information, for example, waste water treatment plants can be designated as 'at risk' because they are serving a larger population then they were designed to cater for or because their discharges are impacting on water quality. Section 5 of this characterisation report provides the detail behind the risk designations for each of the pressures and discusses their likelihood to be impacting on shellfish water quality parameters.

Whilst the risk designations under the WFD provide a useful screening tool for pressures, their relevance in terms of any water quality issues measured in Shellfish Waters has been assessed in further detail to identify key pressures at a particular site. For example the WFD risk may be based on particular impacts to freshwater ecology which are not pertinent to the shellfish water status.

TABLE 2 - List of maps

Map No.	Map Title	Details
General Characteristics Maps		
MAP 1	Designated shellfish area	Designated shellfish area with summary statistics.
MAP 2	Licensed shellfish areas	Department of Agriculture, Fisheries and Food register of licensed shellfish areas within the designated shellfish area.
MAP 3	Contributing catchment	Nested river water bodies and inter-coastal freshwater bodies discharging in the vicinity of the designated shellfish area.
MAP 4	Topography	Topography of the contributing catchment.
MAP 5	Soil wetness	Soil wetness which indicates drainage characteristics
MAP 6	Vulnerability of groundwaters to pathogens from subsoil discharges	Potential risk of pathogens from sub-soils discharges reaching groundwaters. Based on vulnerability, presence of alluvium, mineral content of soils, wetness, aquifer type, subsoil depth and subsoil permeability.

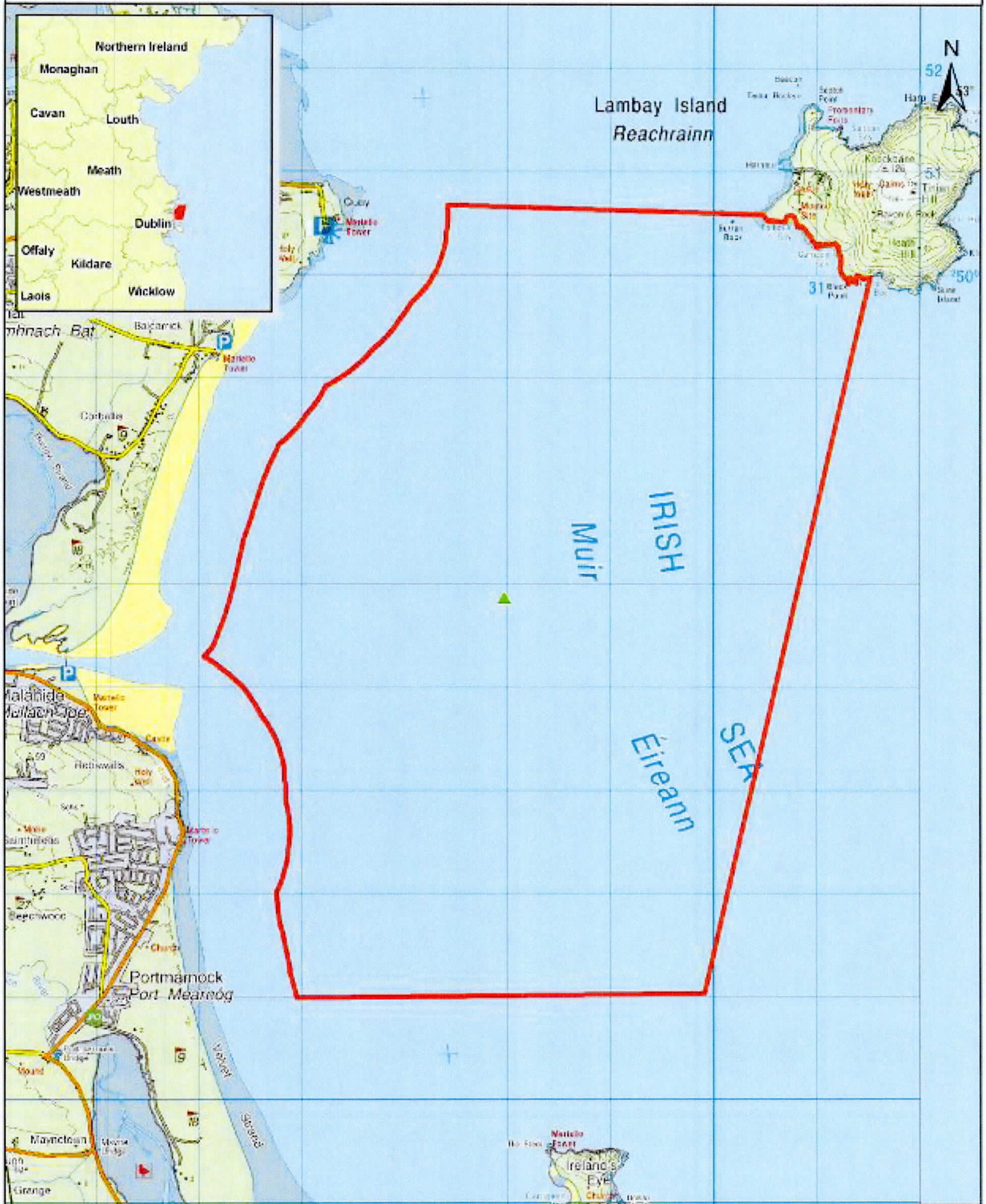
Map No.	Map Title	Details
MAP 7	Vulnerability of groundwaters to phosphorus from subsoil discharges	Potential risk of phosphorus from sub-soils discharges reaching groundwaters. Based on vulnerability, presence of alluvium, mineral content of soils, wetness, aquifer type, subsoil depth and subsoil permeability.
MAP 8	Vulnerability of surface waters to pathogens from subsoil discharges	Potential risk of pathogens from sub-soils discharges reaching surface waters. Based on vulnerability, presence of alluvium, mineral content of soils, wetness, aquifer type, subsoil depth and subsoil permeability.
MAP 9	Vulnerability of surface waters to phosphorus from subsoil discharges	Potential risk of phosphorus from sub-soils discharges reaching surface waters. Based on vulnerability, presence of alluvium, mineral content of soils, wetness, aquifer type, subsoil depth and subsoil permeability.
MAP 10	Likelihood of inadequate percolation in subsoils	Likelihood of inadequate percolation in subsoils. Based on aquifer type, vulnerability and subsoil permeability.
MAP 11	Designated protected areas	SACs, SPAs, freshwater pearl mussel areas, recreational waters, drinking waters, nutrient sensitive areas, water dependant habitats and RAMSAR sites within the contributing catchment.
MAP 12	WFD surface water status	River, lake, transitional and coastal water body status resulting from the WFD monitoring programme.
MAP 13	EPA diffuse risk assessment	Water body based risk to waters from diffuse sources. Based on the percentages of diffuse land cover per water body including peatlands, coniferous forestry, agriculture and urban areas.
Marine Pressures Maps		
Point Source Pressures		
MAP 14	Marine finfish farms	Marine finfish farms in the vicinity of the designated shellfish area. Taken from the Marine Atlas.
Morphology Pressures		
MAP 15	Fishing gear activity	Fishing gear activity in the vicinity of the designated shellfish area. Taken from the Marine Atlas.
MAP 16	Structures	Marine morphology structures such as bridges and causeways

Map No.	Map Title	Details
MAP 17	Physical modifications	Physical modifications such as shoreline reinforcement, embankments, reclaimed land, capital and maintenance dredging, aggregate removal, dumping at sea and heavily modified waters within the designated shellfish area.
Land-based Pressures Maps		
Point Source Pressures		
MAP 18	Municipal waste water systems	Urban waste water treatment plants and combined sewer overflows within the contributing catchment. These are symbolized based on their risk designations.
MAP 19	Agricultural and aquacultural point source pressures	Pig units, and freshwater fish farms within the contributing catchment.
MAP 20	Industrial point source pressures	Industrial IPPCs, Section 4s, water treatment plants, abstractions, mines, quarries, landfills and contaminated sites within the contributing catchment. These are symbolized based on their risk designations.
Diffuse Source Pressures		
MAP 21	On-site waste water systems	On-site waste water treatment plants within the contributing catchment.
MAP 22	Dairy and drystock livestock units	Dairy and drystock livestock units per hectare of farmed land within each DED in the contributing catchment.
MAP 23	Nitrogen fertiliser usage	Nitrogen fertiliser usage per hectare of farmed land within each DED in the contributing catchment.
MAP 24	Phosphorus fertiliser usage	Phosphorus fertiliser usage per hectare of farmed land within each DED in the contributing catchment.
MAP 25	Forestry types with acidification risk areas	Forest cover in the contributing catchment with areas identified as being at risk from acidification.
MAP 26	Forestry types with eutrophication risk areas	Forest cover in the contributing catchment with areas identified as being at risk from eutrophication.
MAP 27	Forestry types with sedimentation risk areas	Forest cover in the contributing catchment with areas identified as being at risk from sedimentation.
Morphology Pressures		

Map No.	Map Title	Details
MAP 28	Structures	Barriers to migration, both natural and man-made in the contributing catchment.
MAP 29	Physical modifications	Channelisation, heavily modified and artificial water bodies in the contributing catchment.

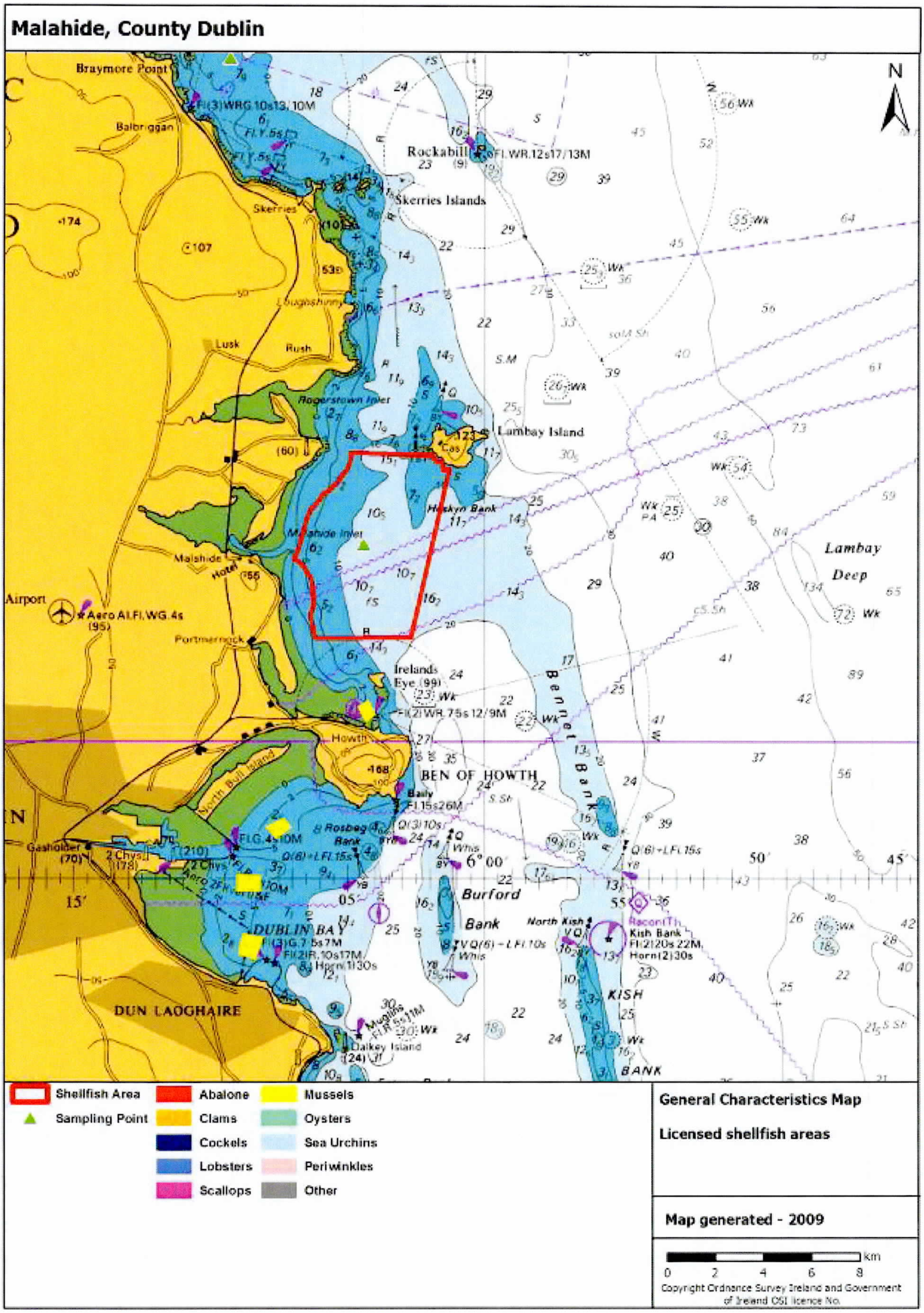
MAP 1 - Designated shellfish area

Malahide, County Dublin

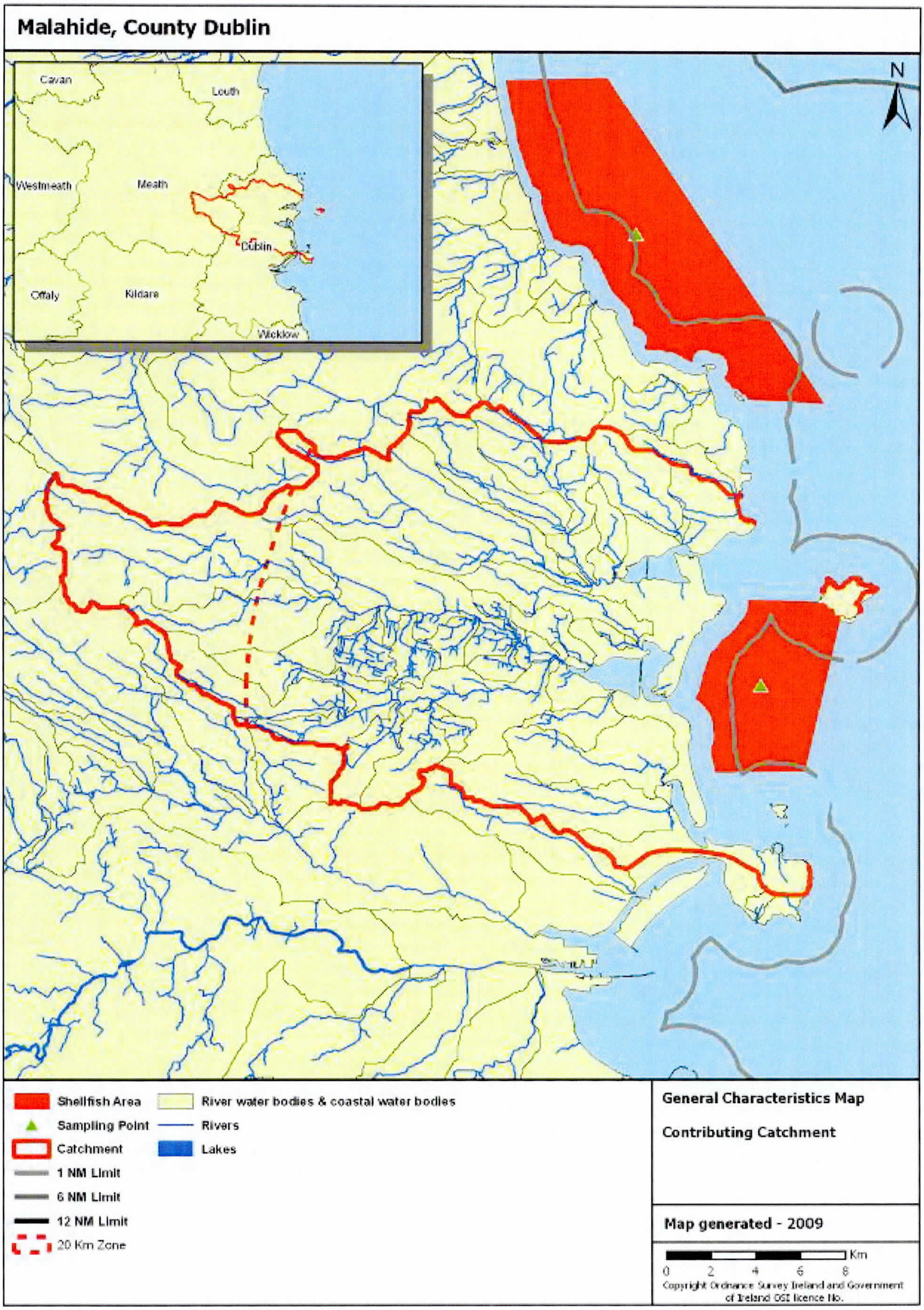


General Characteristics Map		**Shellfish Area**
Designated Shellfish Area	**No. 32 - Malahide, County Dublin**	**Sampling Point**
Area of Designated Shellfish Area	**36.31 km2**	km 0 0.4 0.8 1.2 1.6
Area of Catchment	**376.66 km2**	Copyright Ordnance Survey Ireland and Government of Ireland OSI licence No.

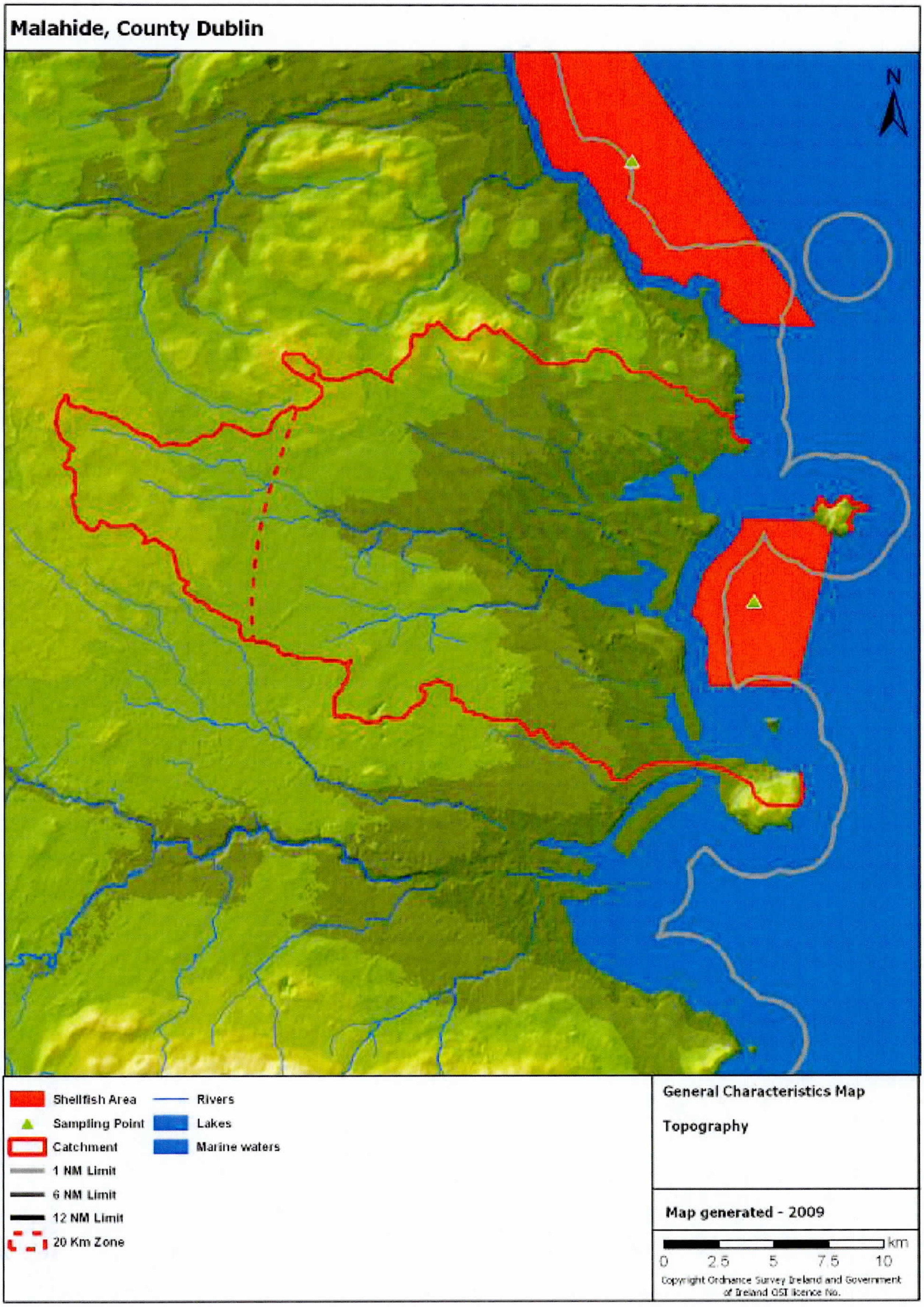
MAP 2 - Licensed shellfish areas



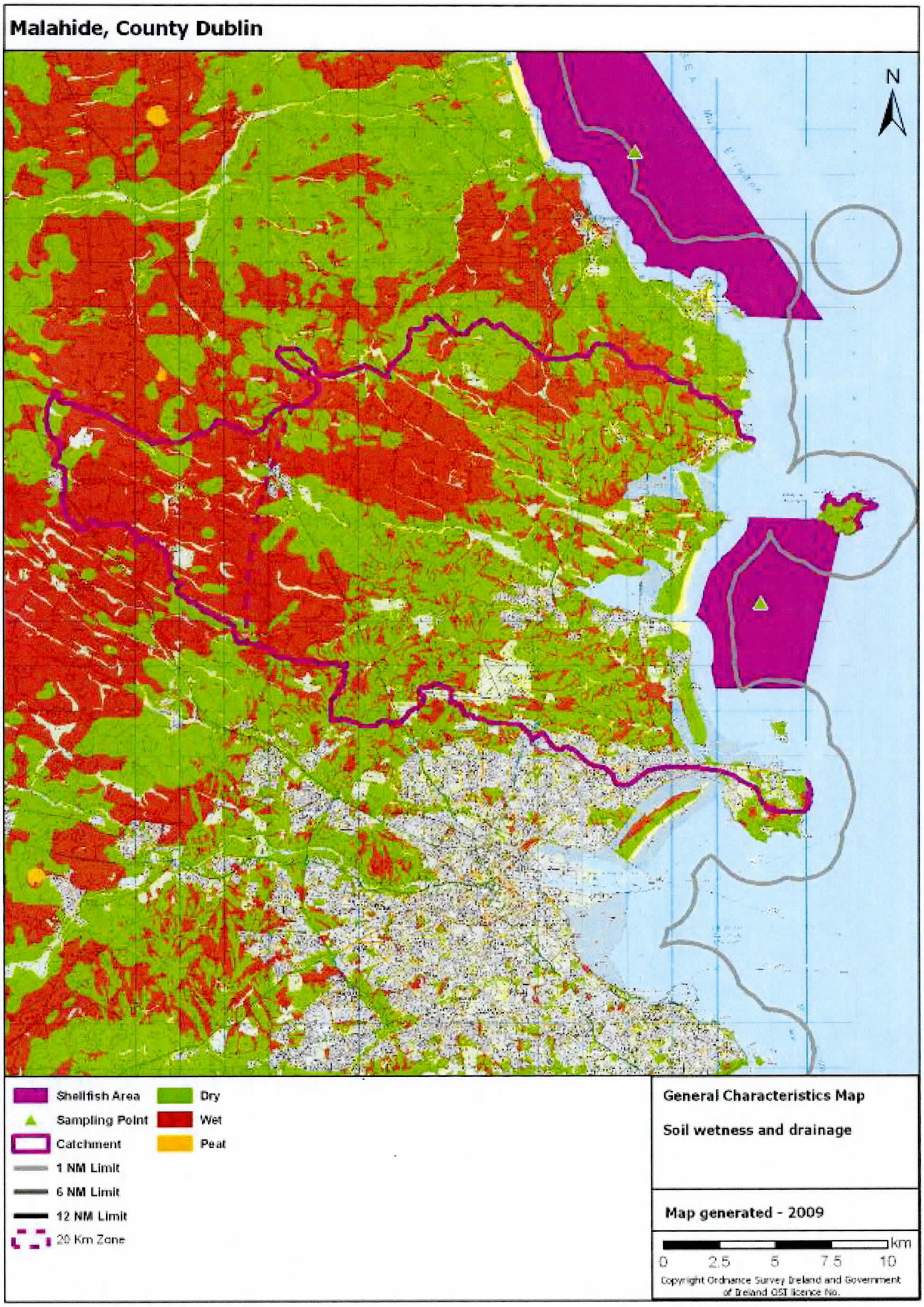
MAP 3 - Contributing catchment



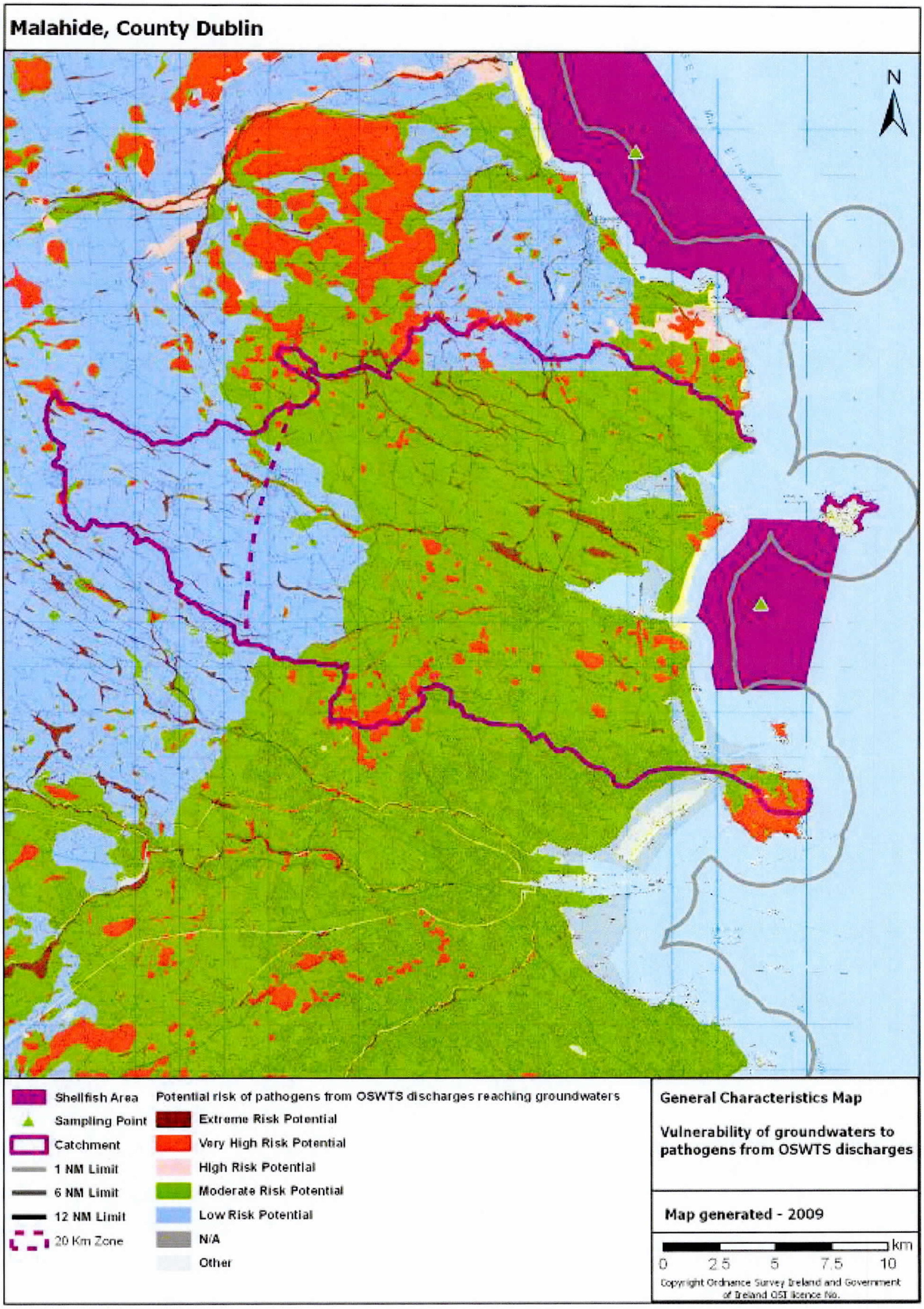
MAP 4 – Topography



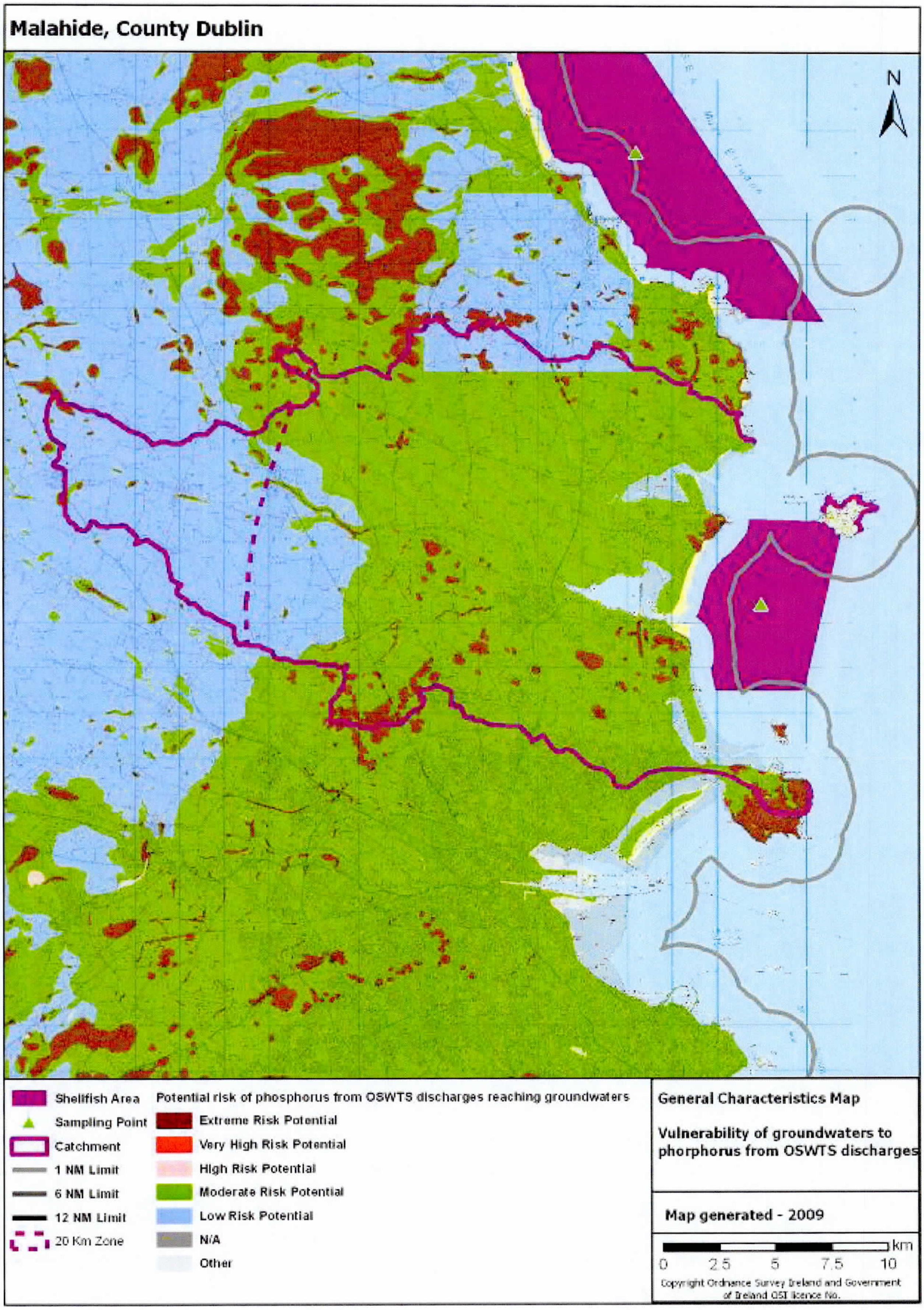
MAP 5 - Soil wetness



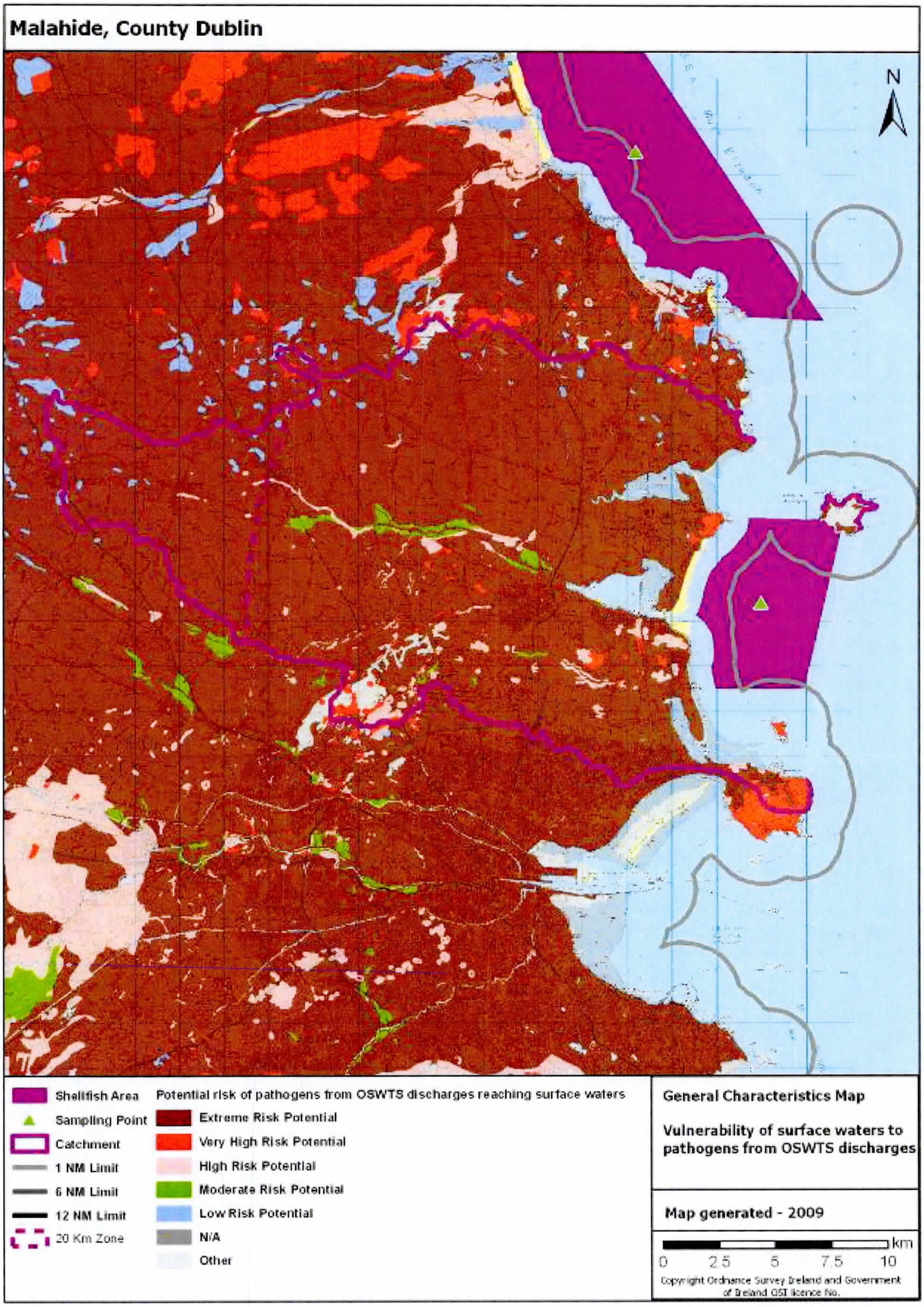
MAP 6 - Vulnerability of groundwater to pathogens from subsoil discharges



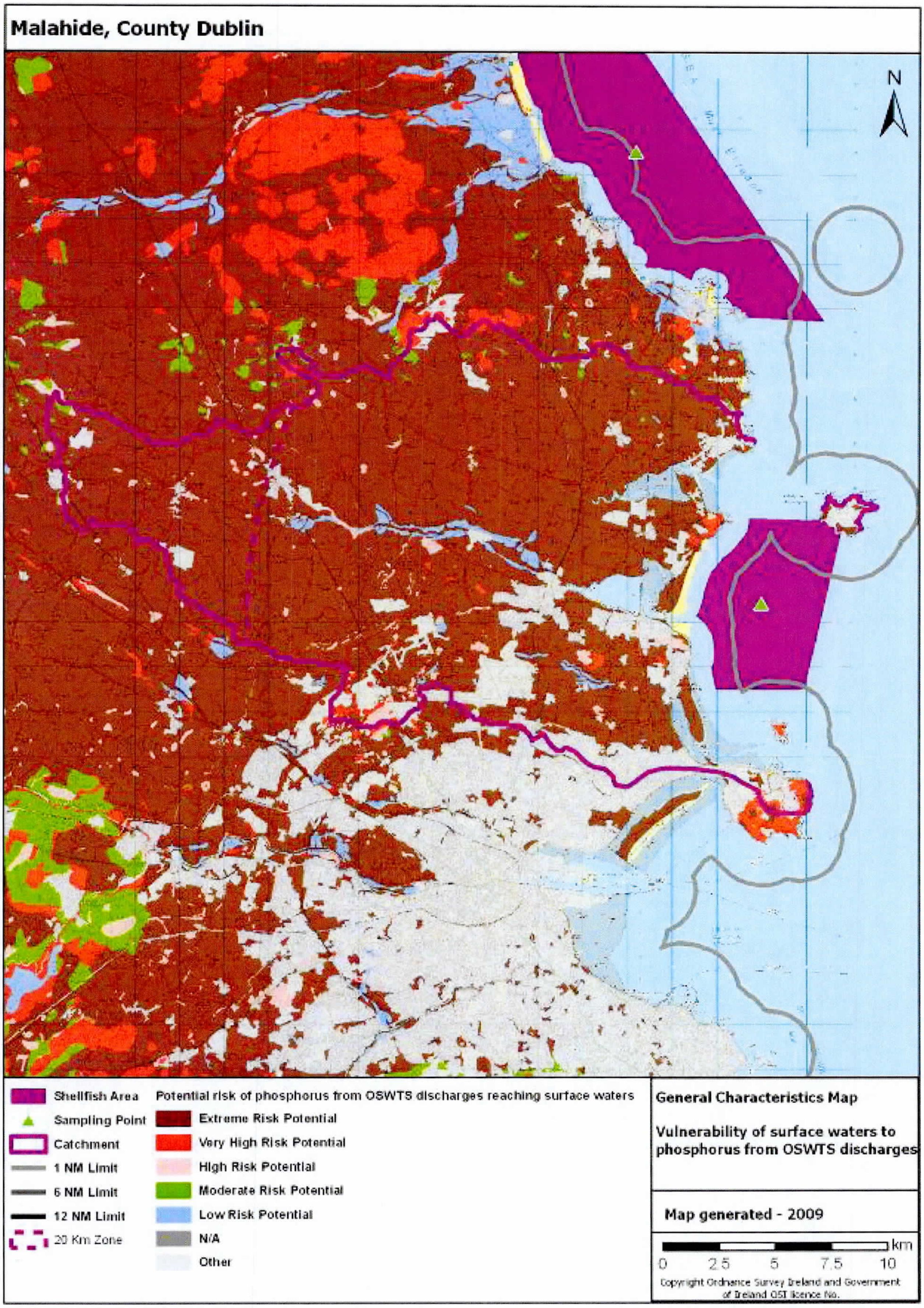
MAP 7 - Vulnerability of groundwater to phosphorus from subsoil discharges



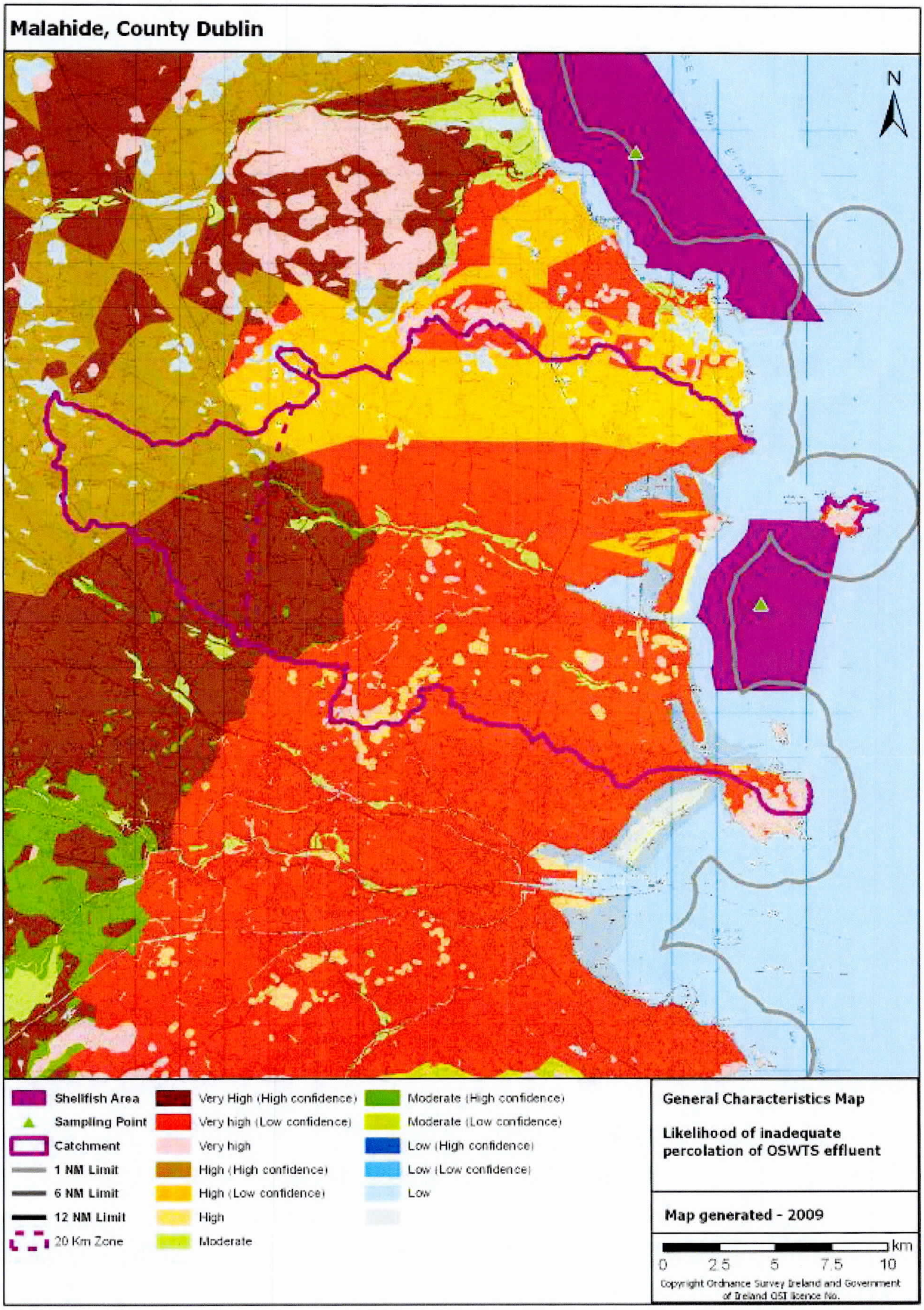
MAP 8 - Vulnerability of surface waters to pathogens from subsoil discharges



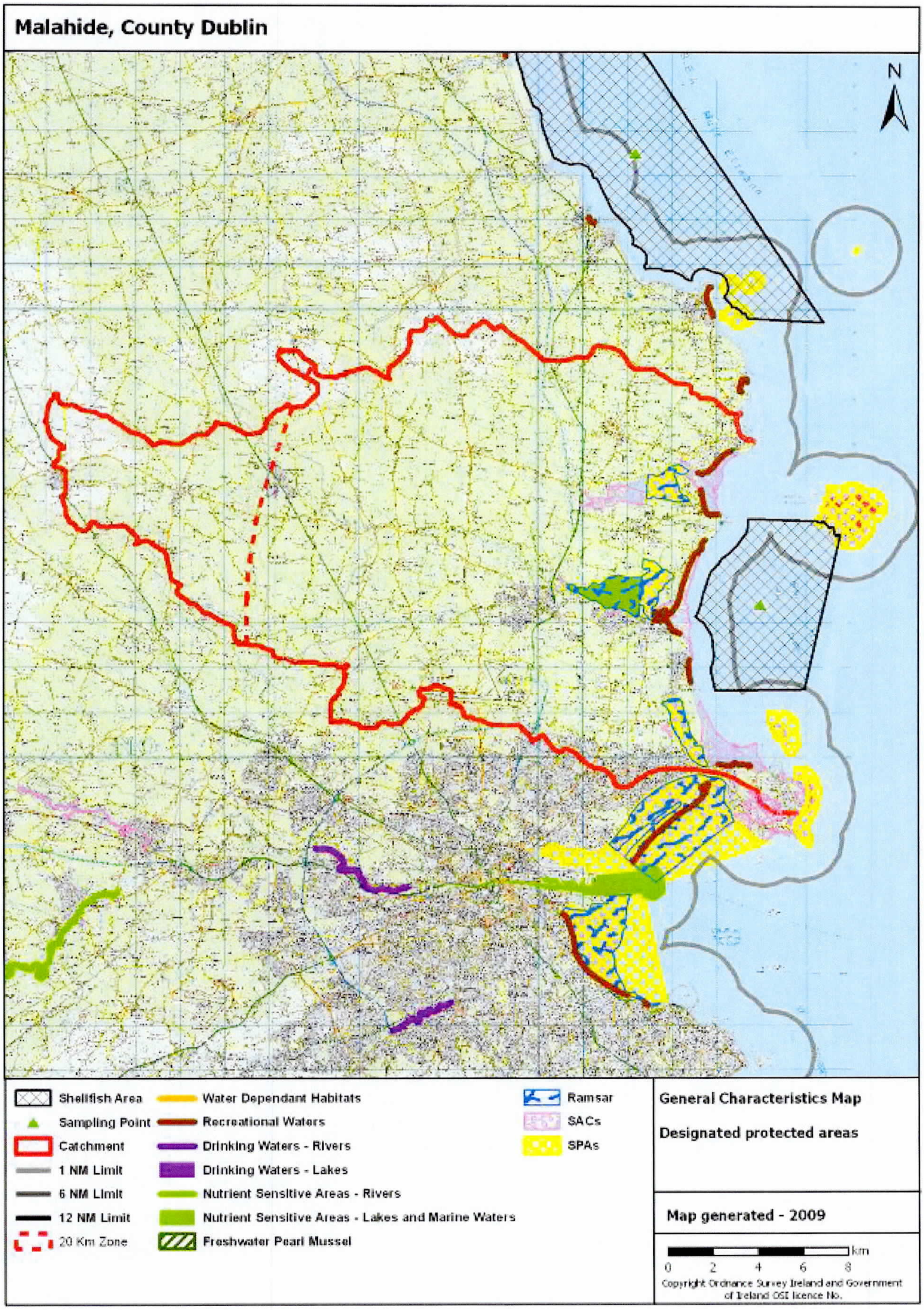
MAP 9 - Vulnerability of surface waters to phosphorus from subsoil discharges



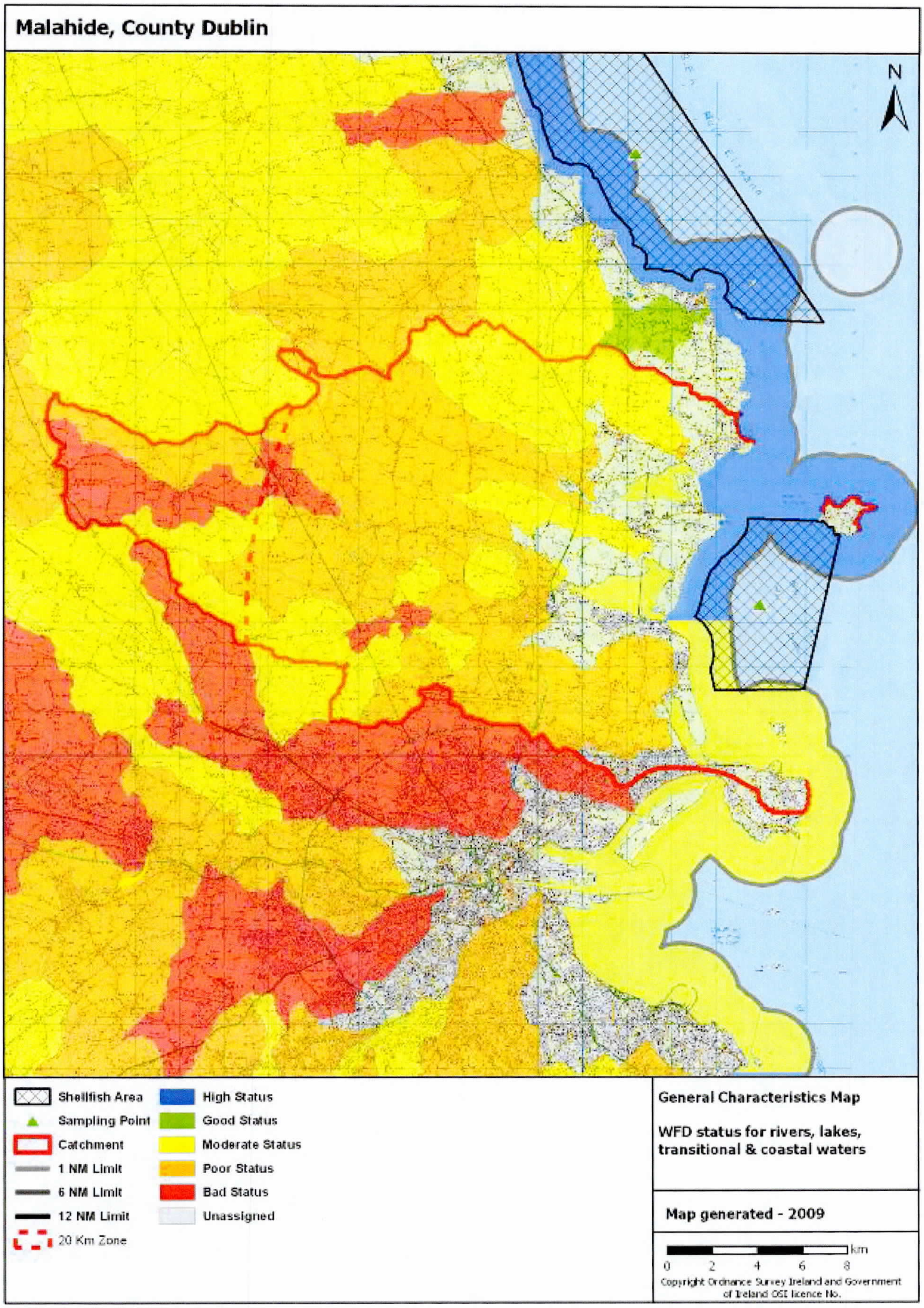
MAP 10 - Likelihood of inadequate percolation in sub-soils



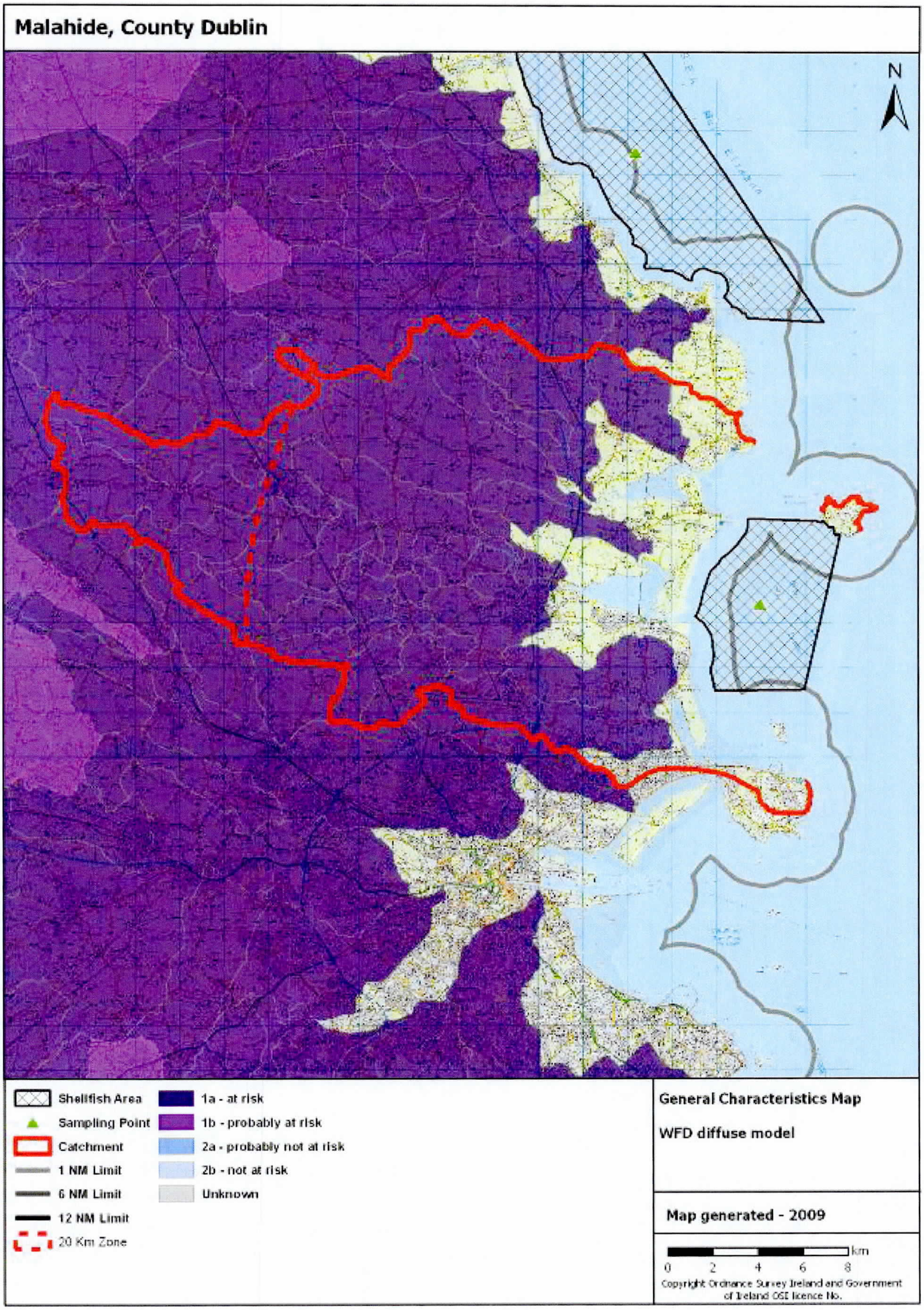
MAP 11 - Designated protected areas



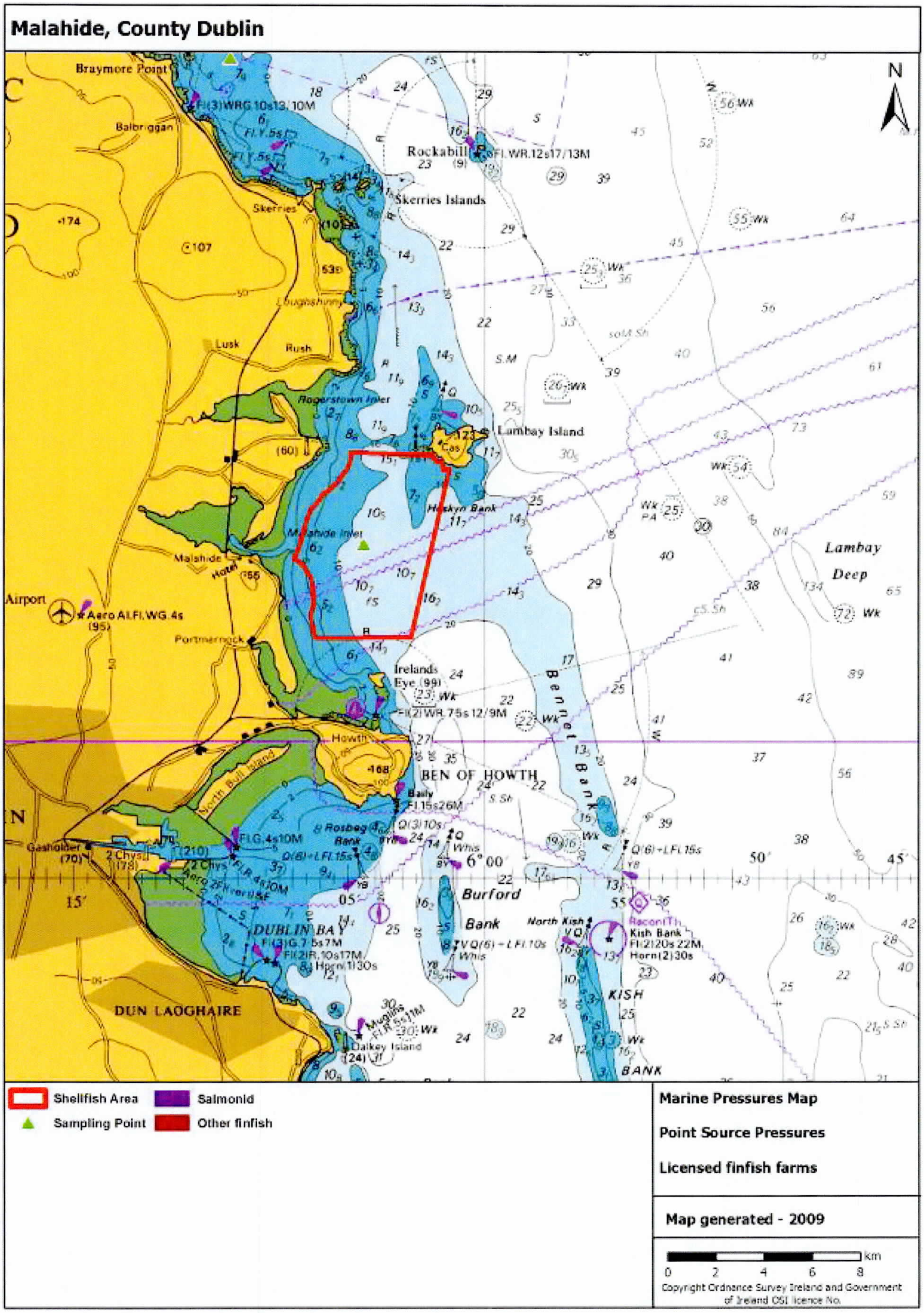
MAP 12 - WFD surface water status



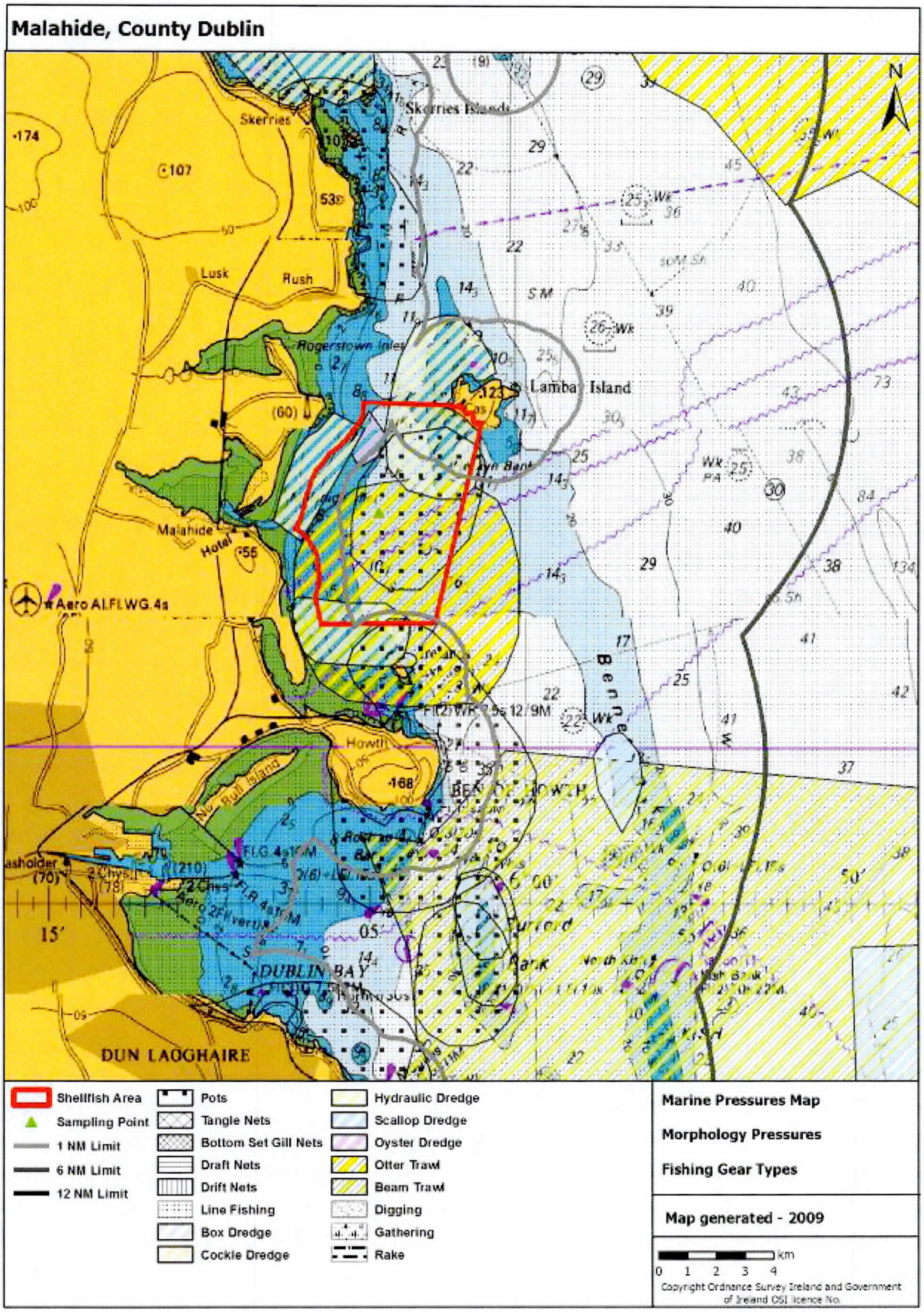
MAP 13 - Diffuse risk assessment



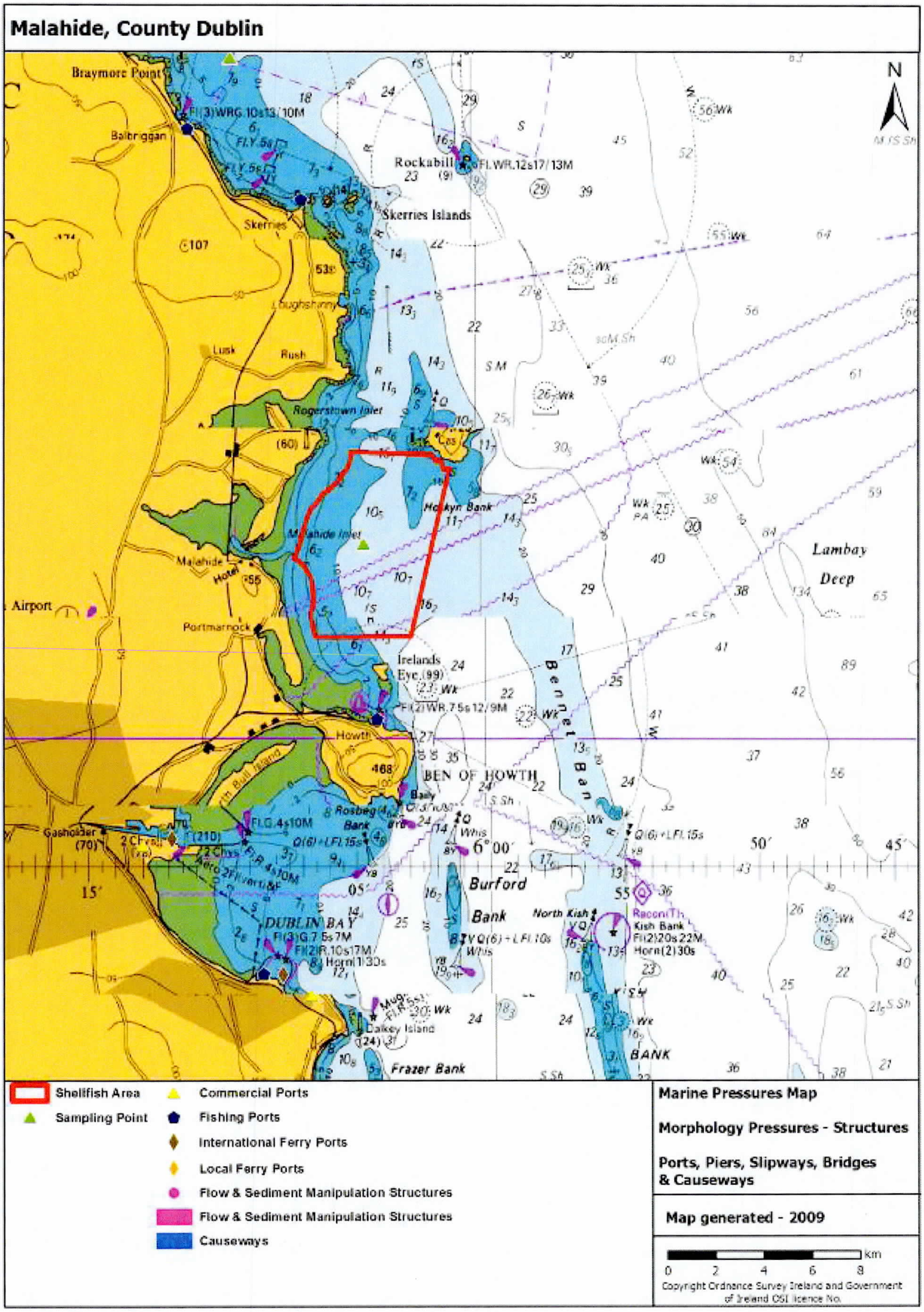
MAP 14 - Licensed finfish areas (None in the vicinity of this shellfish area)



MAP 15 - Fishing gear activity

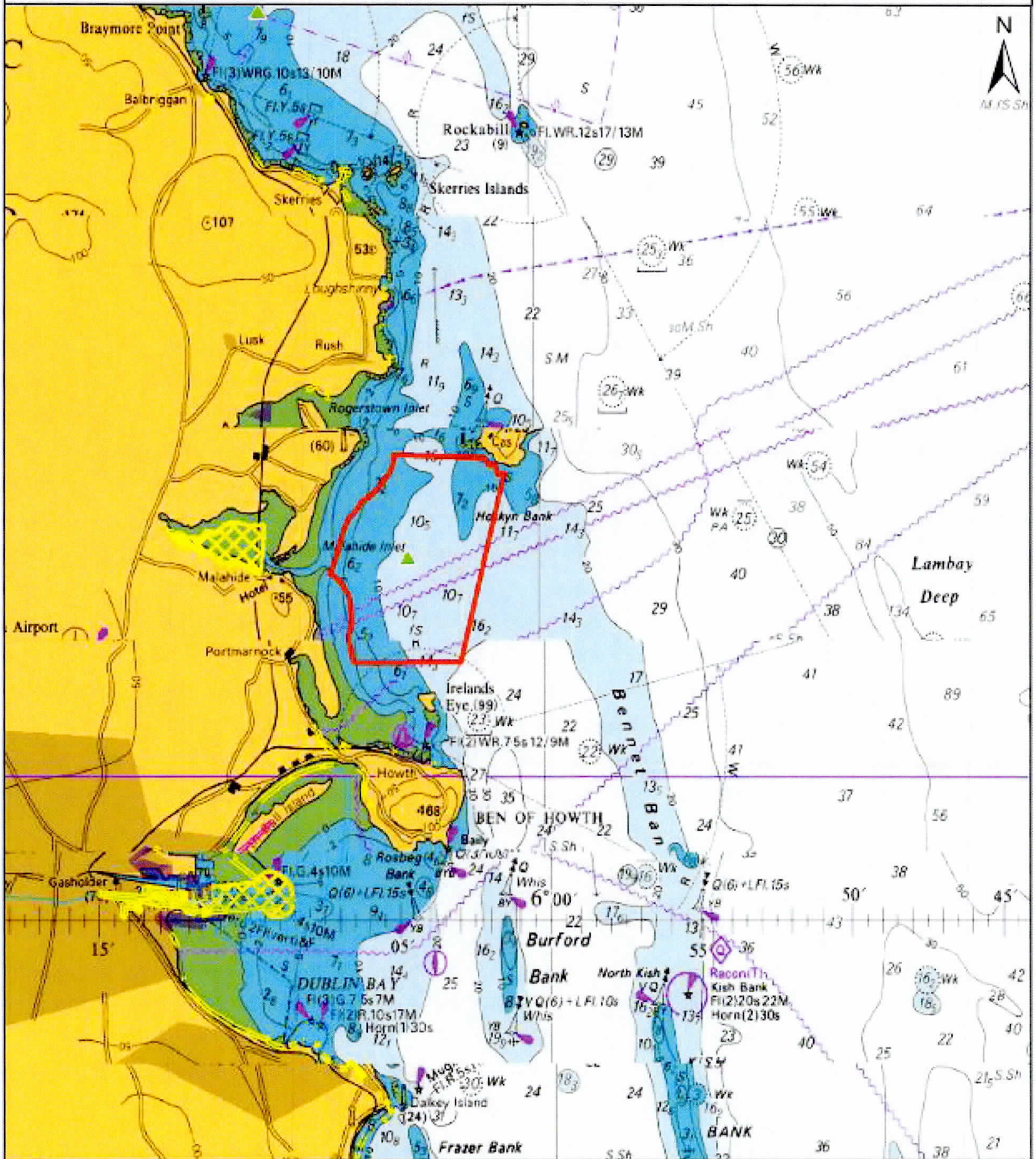


MAP 16 - Marine structures



MAP 17 - Marine physical modifications

Malahide, County Dublin



Shellfish Area	Shoreline Reinforcements
Sampling Point	Embankments
	Reclaimed Land
	Capital Dredging
	Maintenance Dredging
	Aggregate Removal
	Dumping at Sea
	Marine HMWBs

Marine Pressures Map

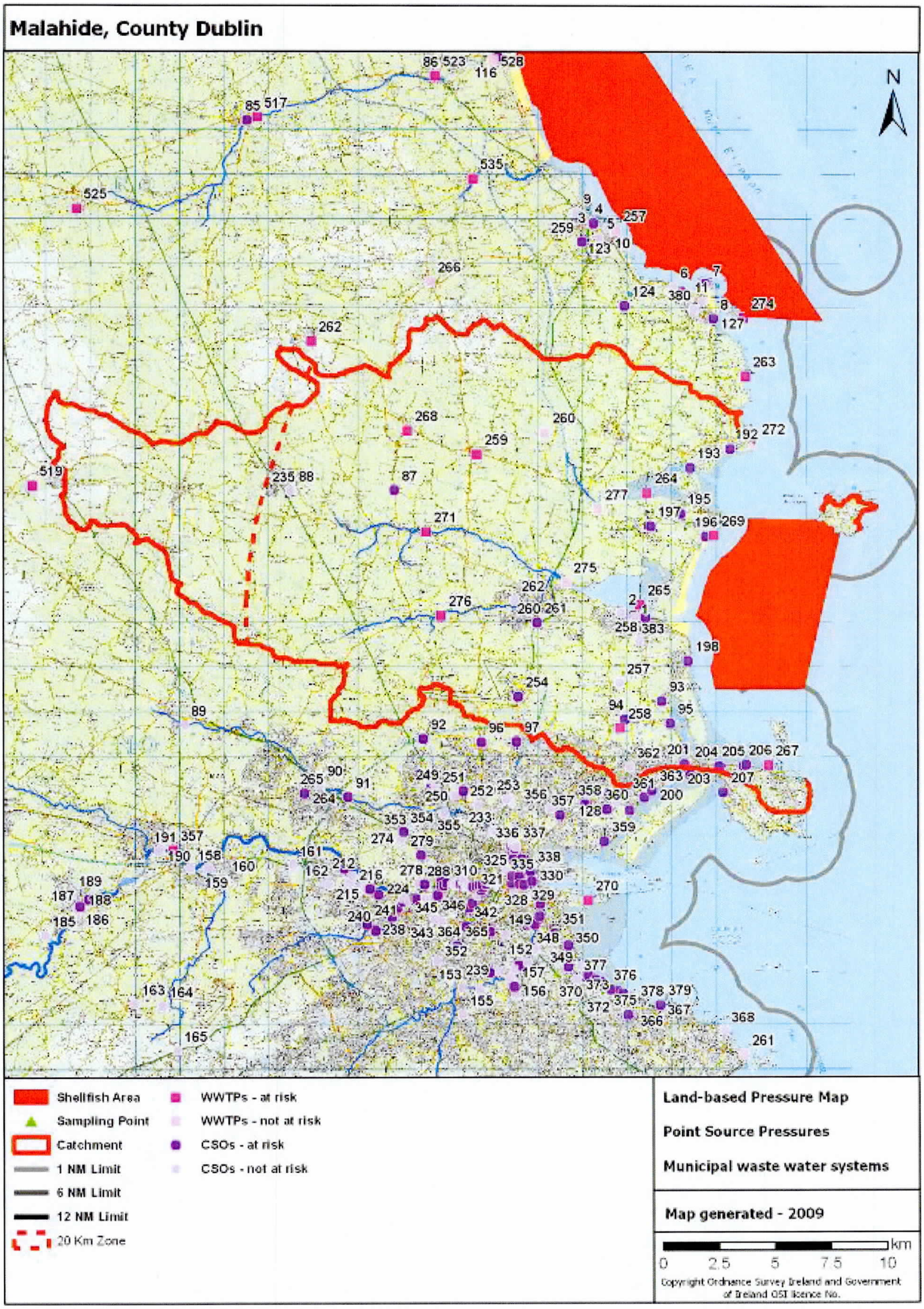
Morphology Pressures - Phys mods

Shoreline enforcement, reclaimed land, dredging & heavily modified

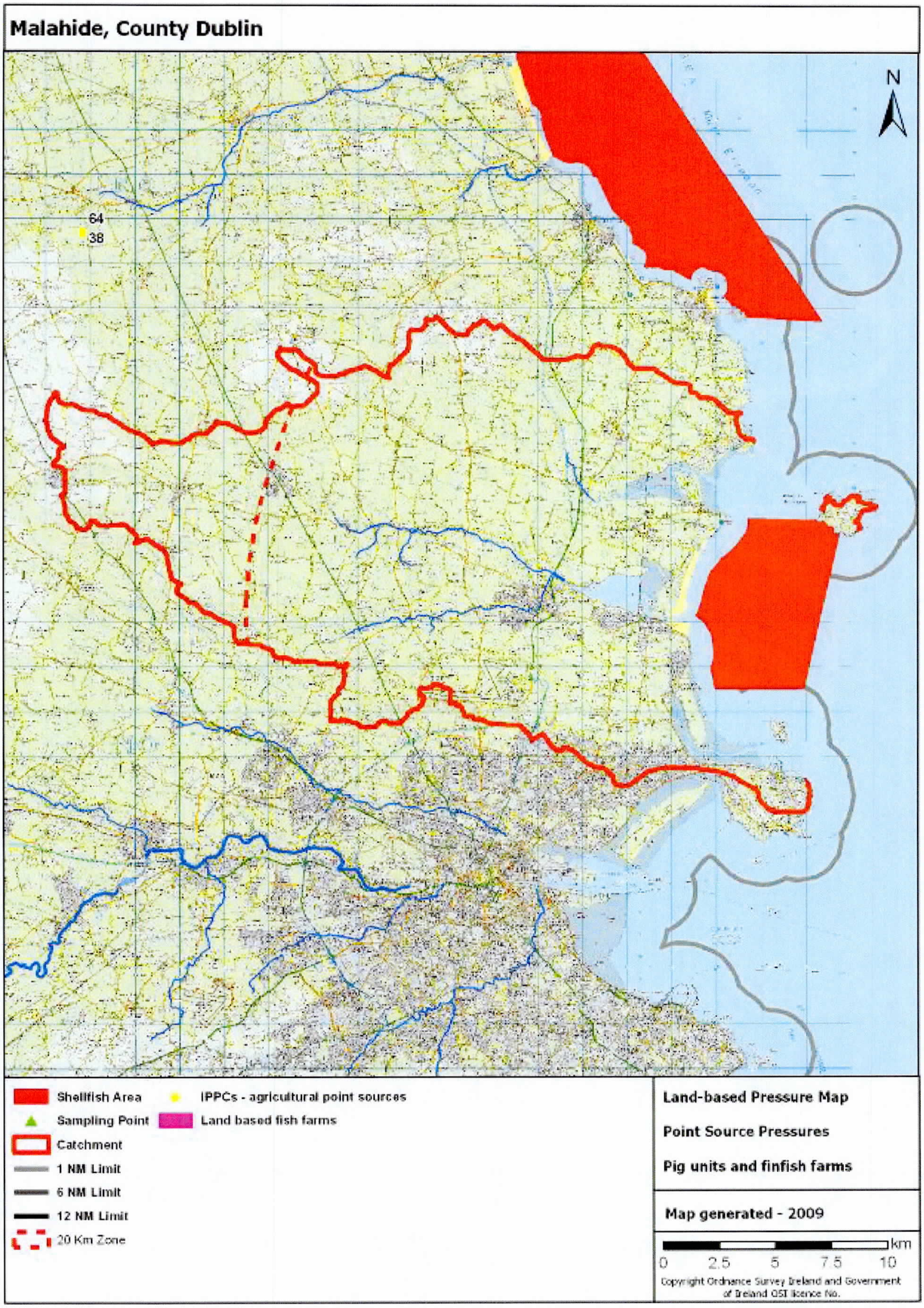
Map generated - 2009

0 2 4 6 8 km

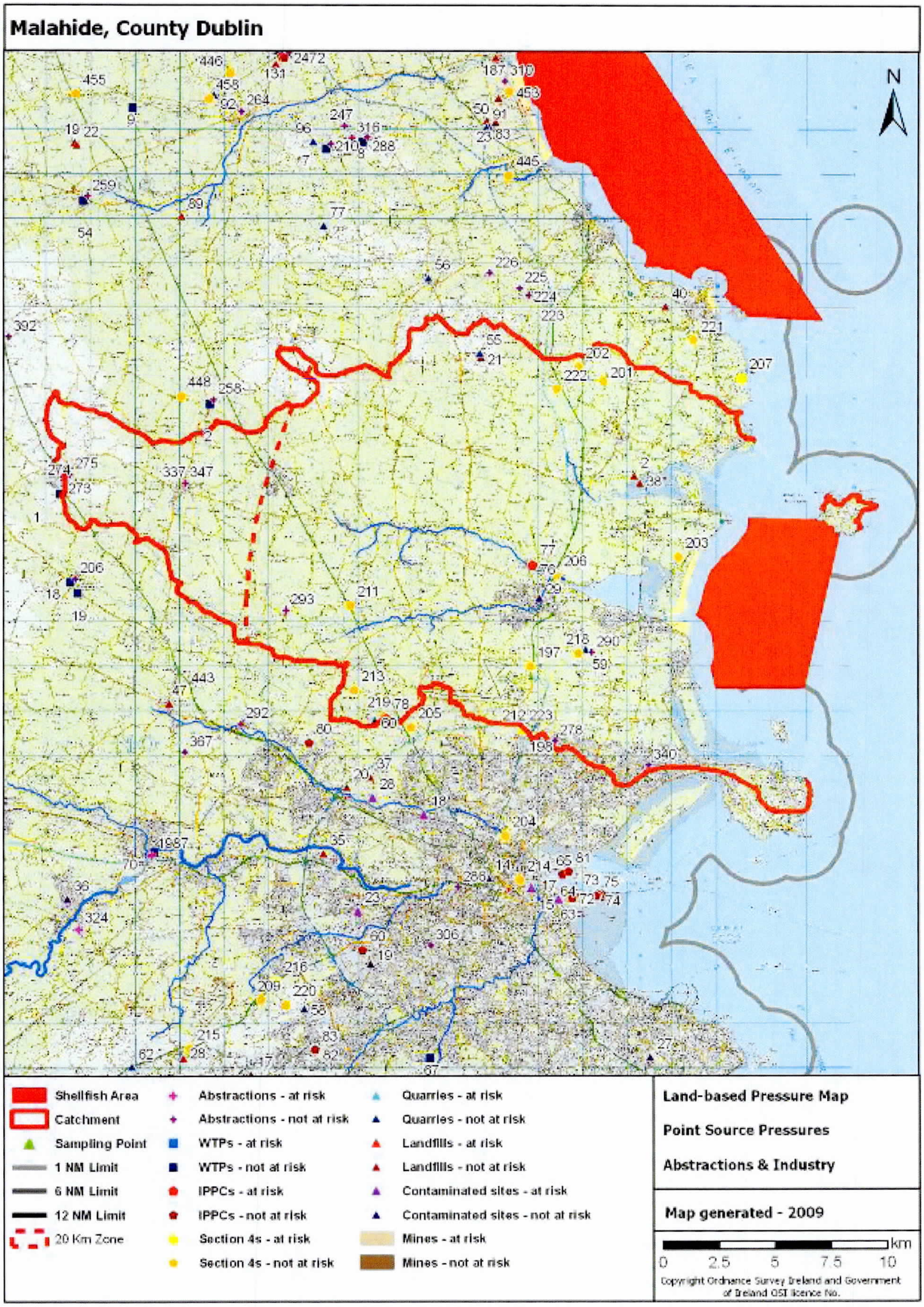
MAP 18 - Municipal waste water systems



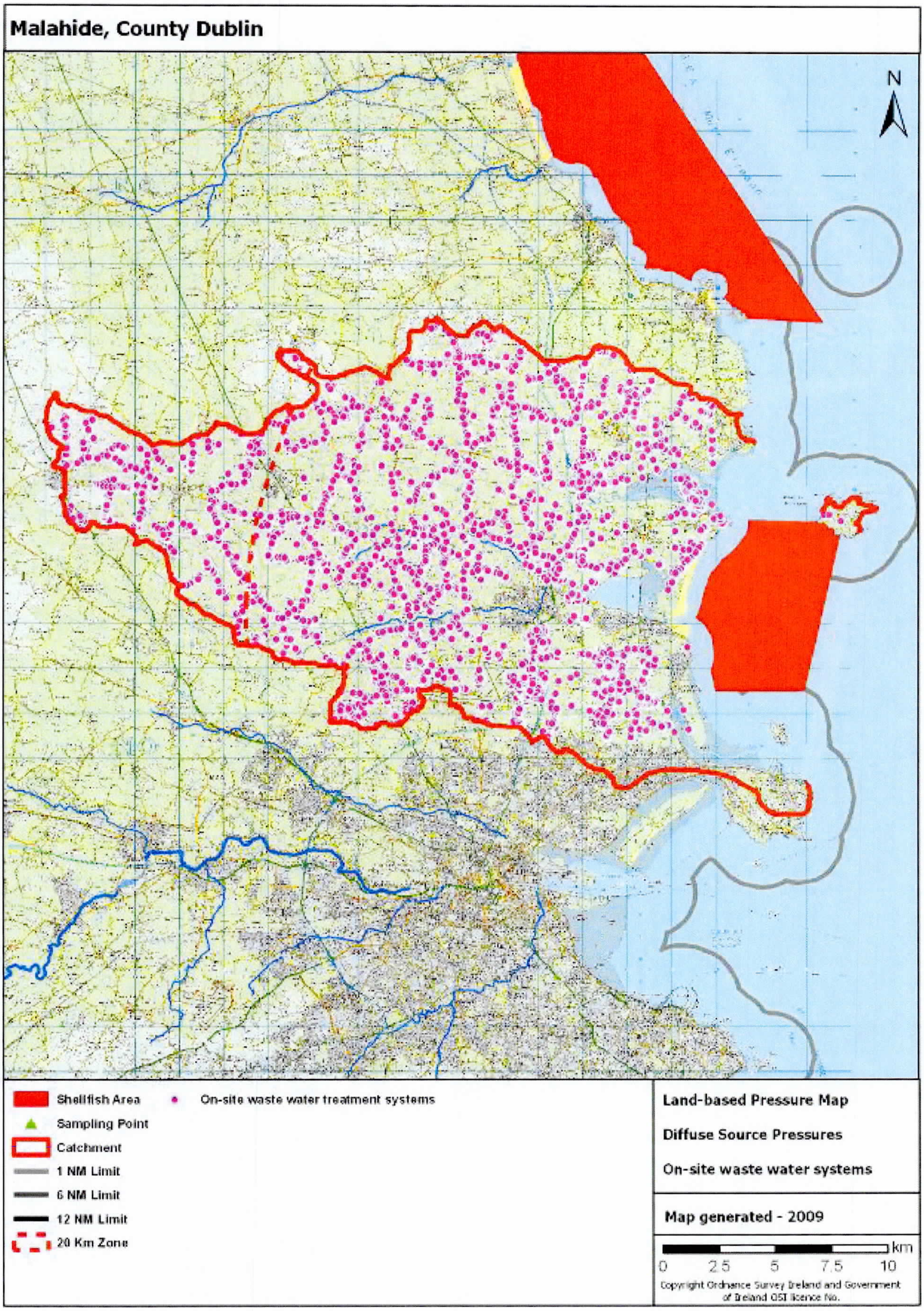
MAP 19 - Pig units and finfish farms (None in this catchment)



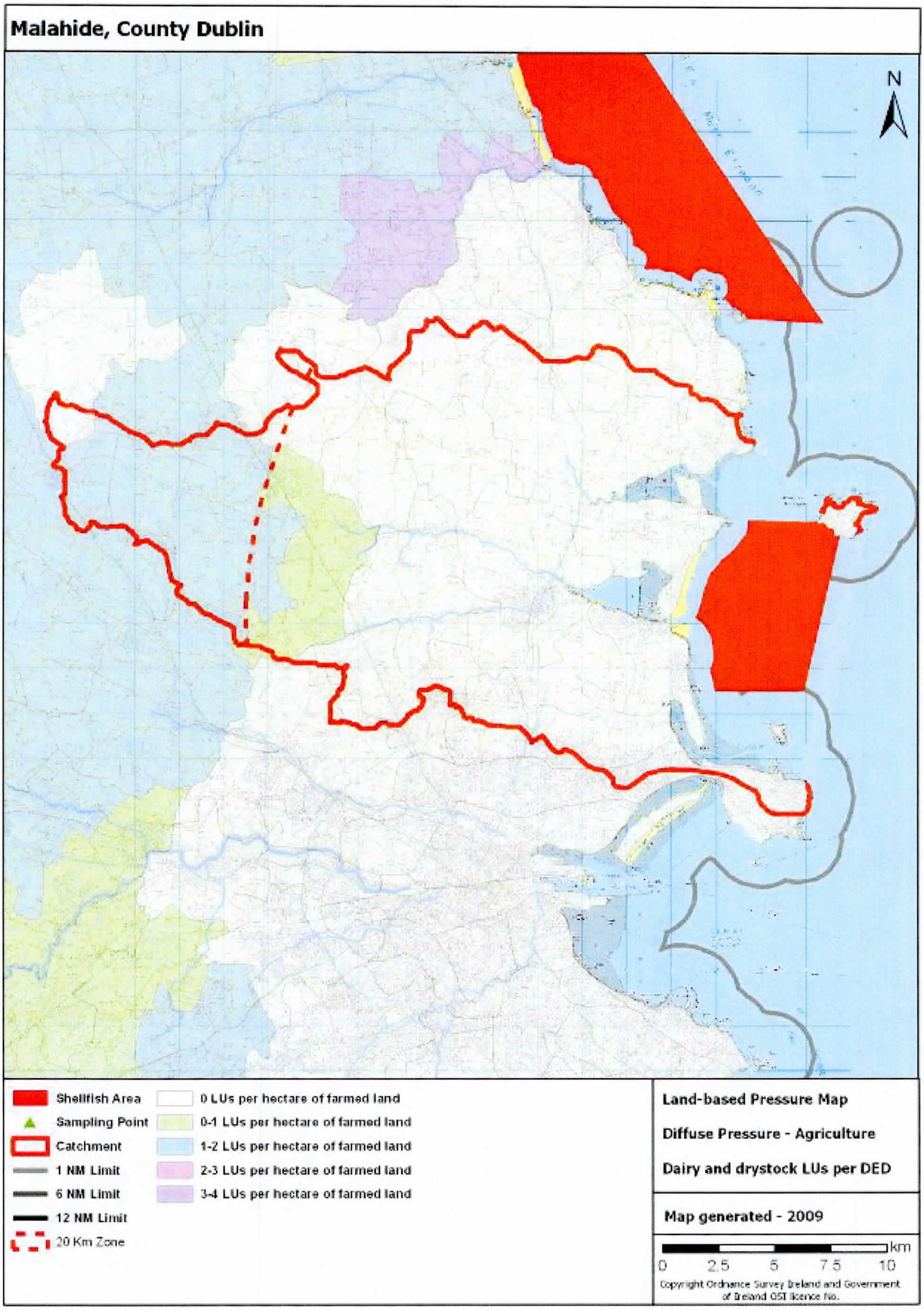
MAP 20 - Industrial point source pressures



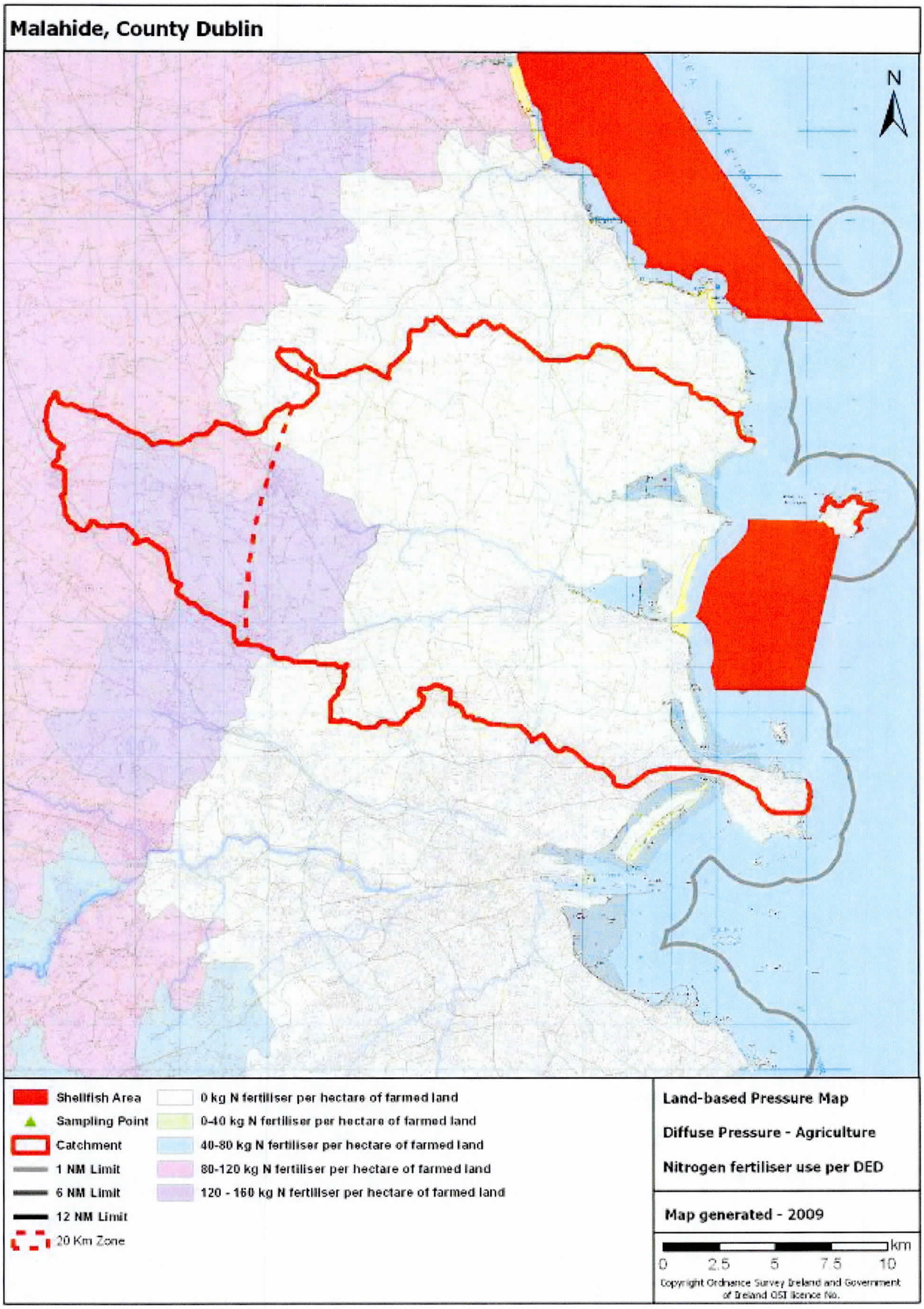
MAP 21 - On-site waste water systems



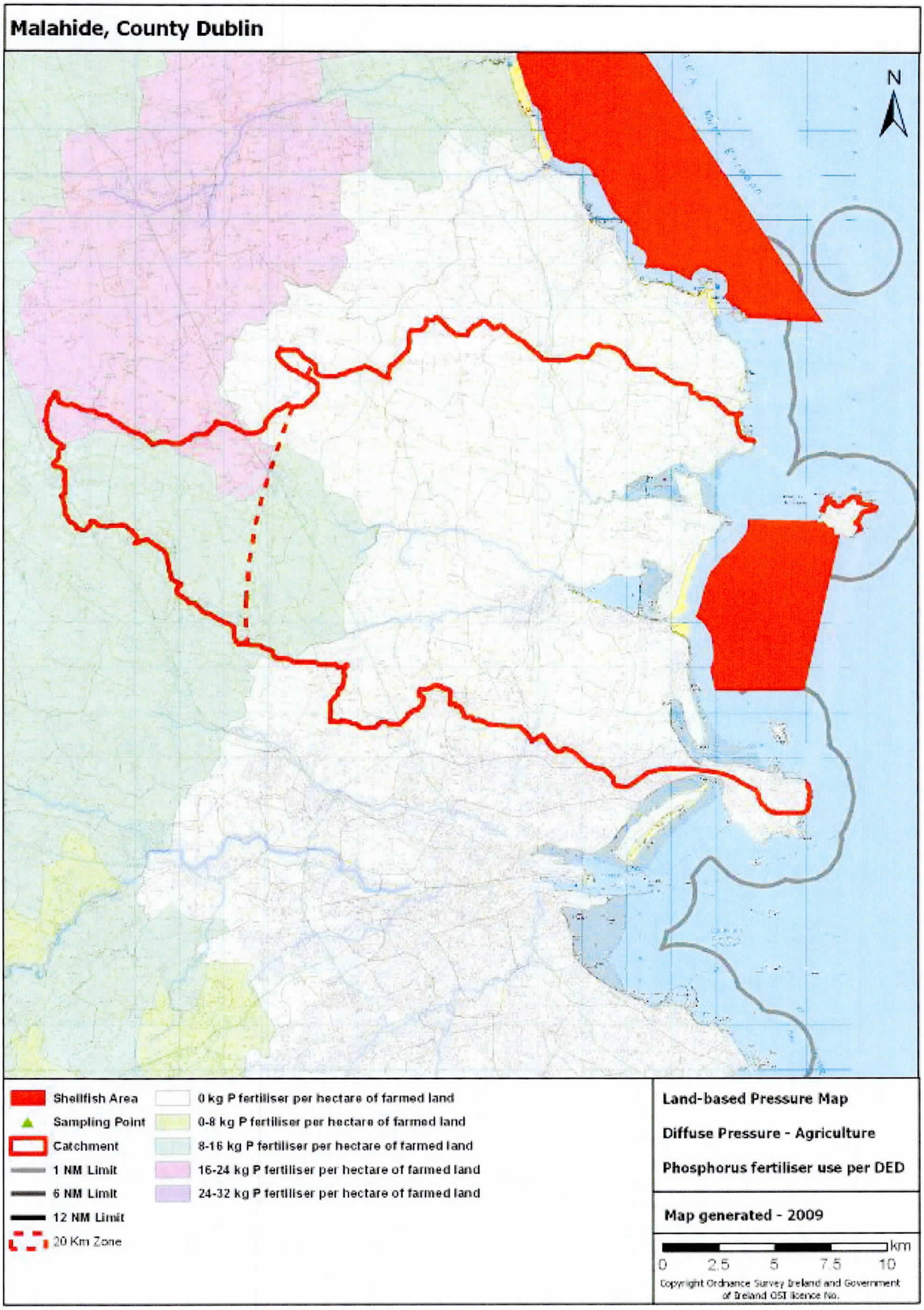
MAP 22 - Dairy and drystock livestock units



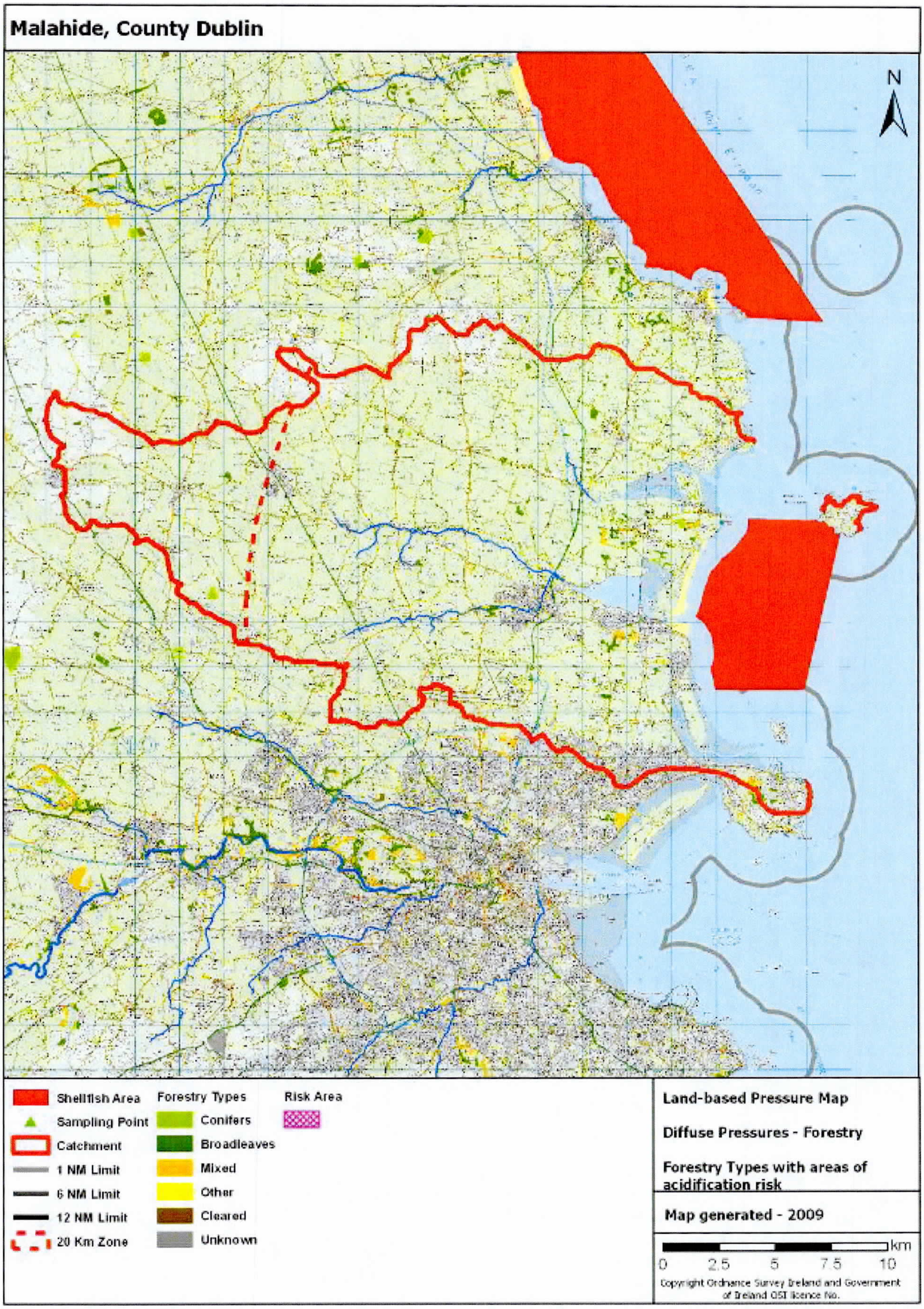
MAP 23 - Nitrogen fertiliser usage



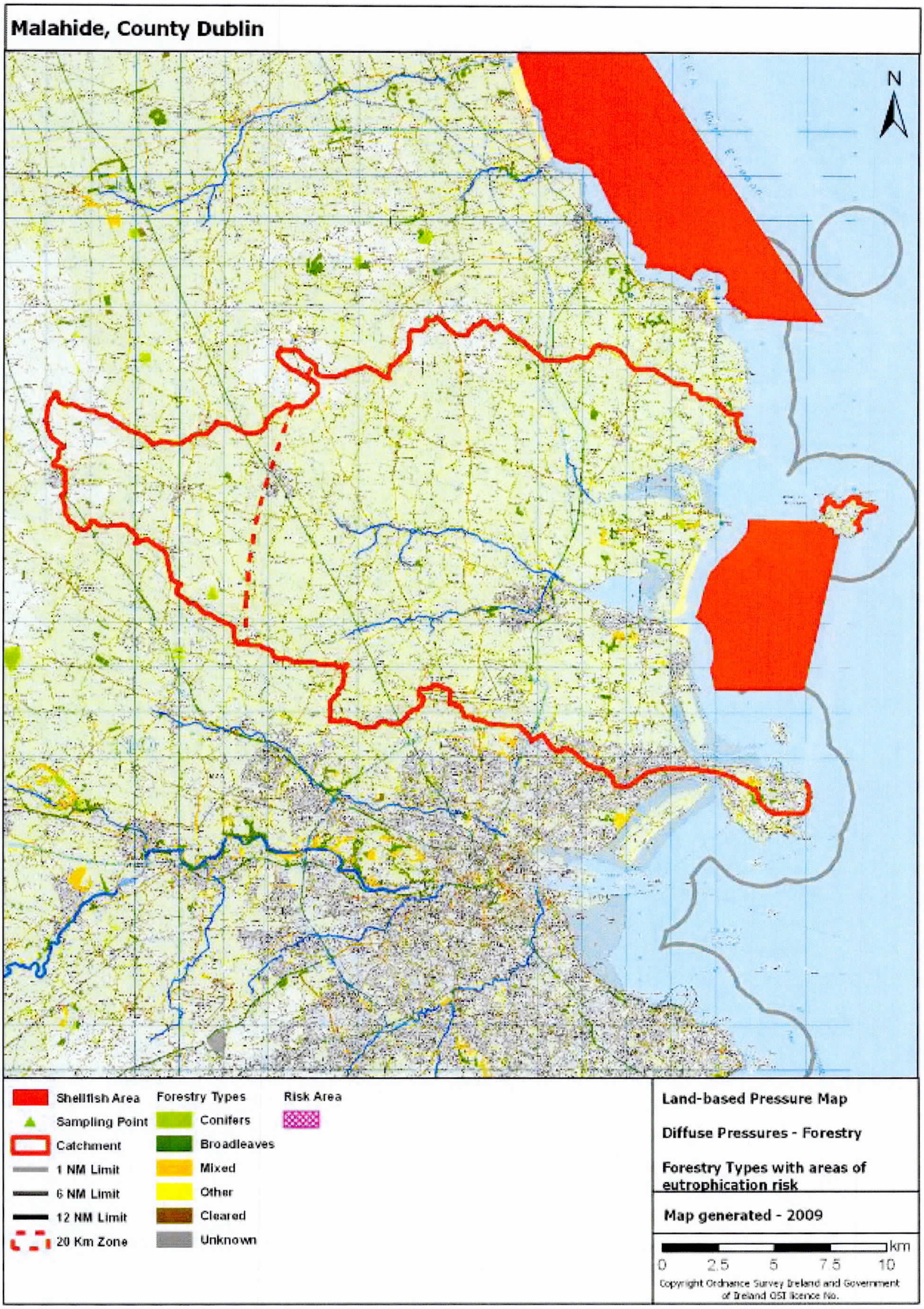
MAP 24 - Phosphorus fertiliser usage



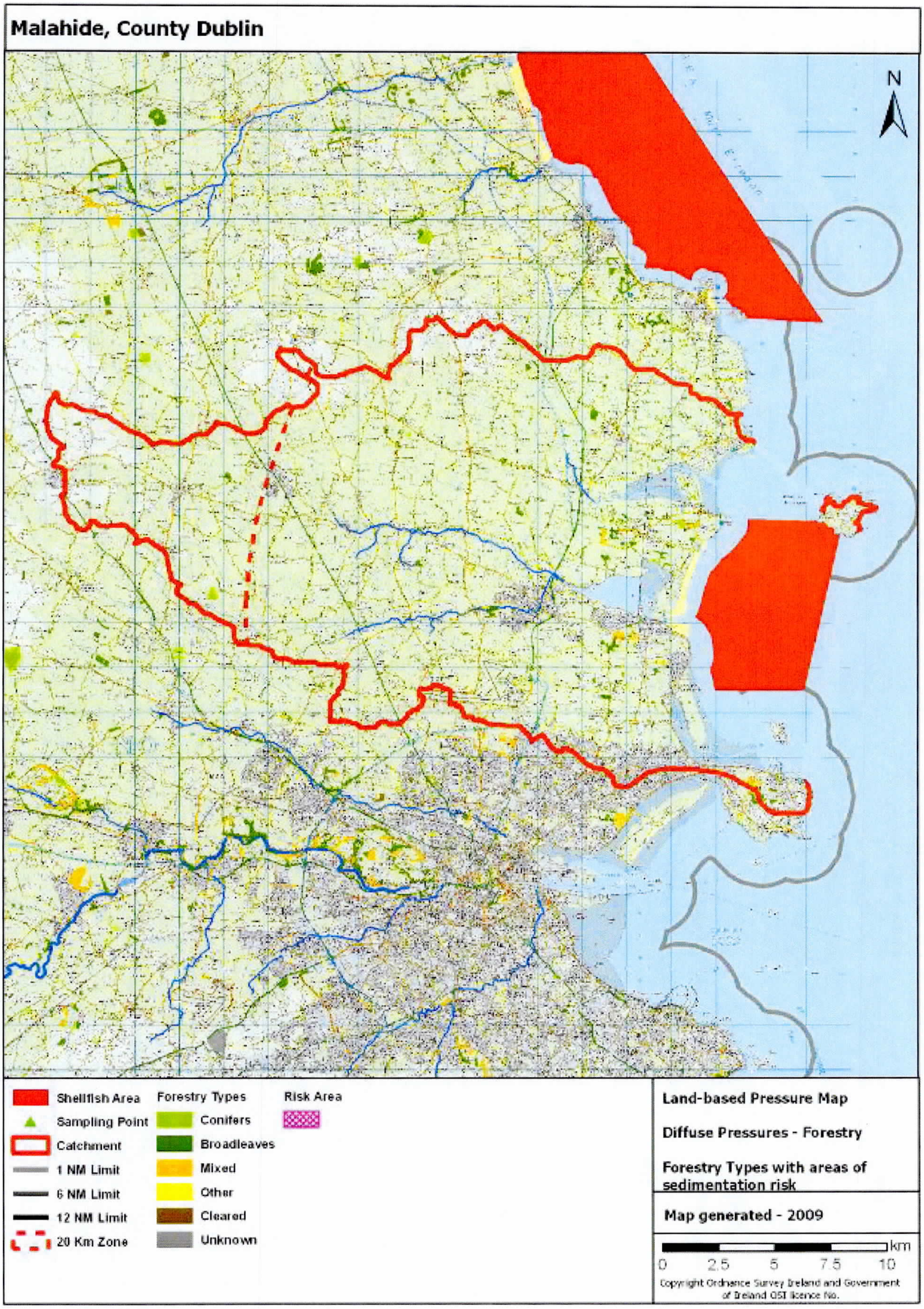
MAP 25 - Forestry types with acidification risk areas



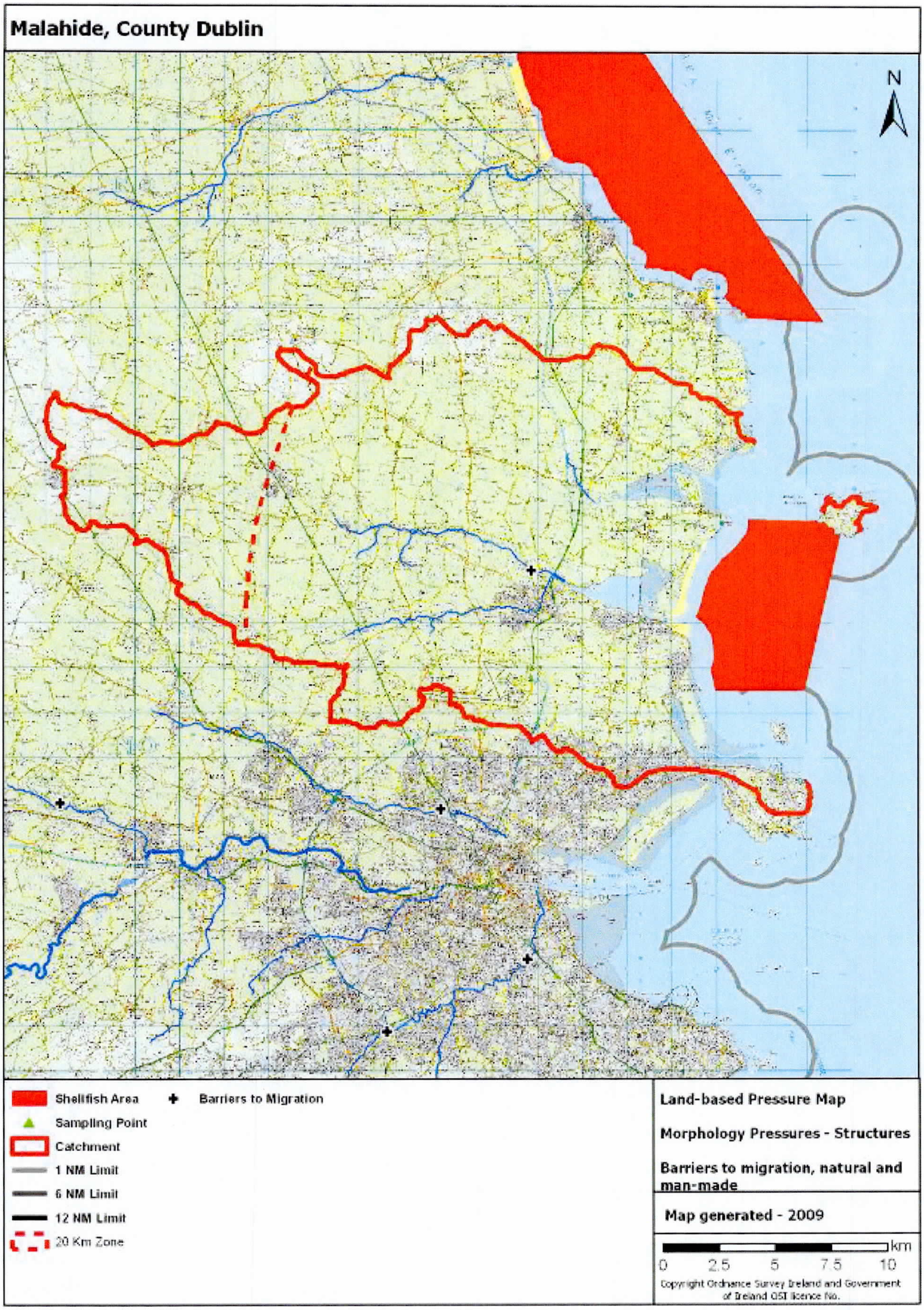
MAP 26 - Forestry types with eutrophication risk areas



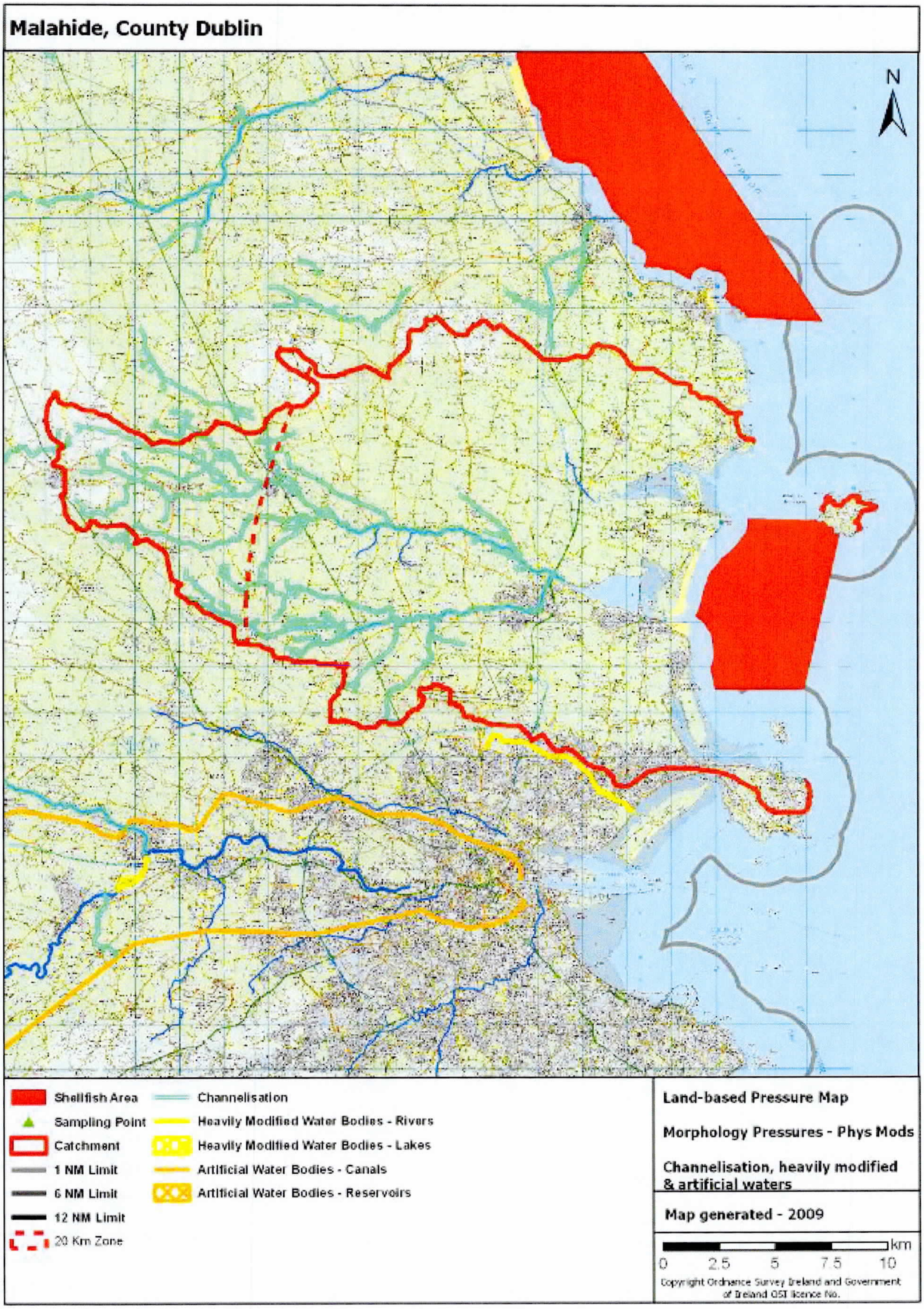
MAP 27 - Forestry types with sedimentation risk areas



MAP 28 - Freshwater structures



MAP 29 - Freshwater physical modifications



5.0 PRESSURES

This section of the characterisation report provides a tabular overview and inventory of the marine and land-based pressures in the vicinity of the designated shellfish area and within the contributing catchment up to a distance of 20 kilometres from the shellfish area. The pressure data has been derived from existing inventories. The pressures considered most likely to be related to any measured impacts on shellfish water quality parameters in this shellfish area have been estimated in order to focus management efforts towards the protection and improvement of the water quality in this shellfish area.

The available information considered when determining the likelihood of the pressures to cause impacts includes:

- pressure type

The pressure types, be it marine or land-based, point, diffuse or morphological, vary in terms of: their likelihood to impact on shellfish water quality; the water quality parameters they are likely to affect; and the severity of the impacts. The results of monitoring can therefore provide an indication of which pressure types are likely to be causing impacts.

- pressure magnitude

The magnitude of the pressures acting on a shellfish area can affect the overall potential impact. For marine pressures, the magnitude depends on the number and scale of the pressures but also on the exposure of the shellfish area to the pressures which in turn depends on how open or sheltered the shellfish area is and on water circulation. For land-based pressures, the magnitude depends on the number and scale of the pressures but also on the remoteness of the pressures from the shellfish areas which in turn depends on the distance of the pressures from the shellfish area, the topography of the catchment and the presence of lakes downstream of pressures which can act as pollution sinks.

- WFD risk designations

A series of risk assessments relating to the main pressures on waters were carried out during the WFD implementation process to identify pressures 'at risk' of impacting the surrounding water environment. These were originally carried out in 2004 and 2005 in accordance with Article V of the directive but many of them were subsequently updated in 2008 to feed into draft River Basin Management Plans. A lot of information about the pressures was collected to undertake these assessments and some of that information is summarised in this section where it is useful in screening which pressures are most likely to impact on shellfish water quality. In all cases, the most up-to-date risk assessment information available was used. Full details of the WFD risk assessments can be found at www.wfdireland.ie.

Whilst the risk designations under the WFD provide a useful screening tool for pressures, their relevance in terms of any water quality issues measured in Shellfish Waters has to be assessed in further detail to identify key pressures at a particular site.

For example, the main issue to be addressed in the Malahide Pollution Reduction Programme is microbial contamination of the shellfish growing waters. Available monitoring data does not suggest, for example, metal contamination of shellfish.

Table 4 lists all of the pressures considered in the development of the characterisation report and indicates their presence or absence within the shellfish area, within the marine waters in the vicinity of the shellfish area or within the contributing catchment. Those pressures that are present are discussed later in this section.

TABLE 3 - Summary of pressures

Pressure type	Pressure type	Pressures	Present
Marine	Point	Marine finfish farms	No
	Morphology	**Fishing gear activity**	**Yes**
		Structures and associated activities	
		Ports	**Yes**
		Flow/Sediment manipulation structures	**Yes**
		Piled structures	**Yes**
		Causeways	No
		Physical modifications	
		Shoreline reinforcement	**Yes**
		Embankments	No
		Reclaimed Land	**Yes**
		Capital dredging	No
		Maintenance dredging	**Yes**
		Aggregate removal	No
		Disposal at sea	No
		Marine heavily modified waters	**Yes**
Land-based	Point	Urban wastewater systems	
		Urban waste water treatment systems	**Yes**
		Combined sewer overflows	**yes**
		Agricultural and aquacultural point sources	
		Pig units	No
		Freshwater finfish farms	No
		Industrial point sources	
		Abstractions	**Yes**
		Water treatment plants	Yes
		IPPCs	**Yes**
		Section 4s	**Yes**
		Quarries	**Yes**
		Landfills	**Yes**
		Mines	No
		Contaminated lands	**Yes**
		Other	No
	Diffuse	**On-site waste water treatment systems**	**Yes**
		Agriculture	
		Livestock density	**Yes**
		Nitrogen fertiliser usage	**Yes**
		Phosphorus fertiliser usage	**Yes**
		Forestry	**Yes**
	Morphology	Structures	
		Barriers to migration	**Yes**
		Physical Modifications	
		Channelisation	**Yes**
		Heavily modified waters	No
		Artificial waters	No

5.1 Marine Pressures

Marine pressures are considered up to a distance of 5 kilometres from the shellfish area. Marine pressures situated further away or in adjacent waterbodies are also mentioned if they are considered significant. Marine pressure types include point source pressures (marine finfish farms) and morphological pressures including fishing gear activity, structures (ports, bridges, piers, slipways etc) and physical modifications (shoreline reinforcement, embankments, dredging etc). The potential impacts associated with these pressures are as follows:

- Point source pressures

Marine finfish farms can be associated with increased nutrient levels in waters, arising from fish excretion and excess feed input.

- Morphological pressures

Fishing activity can be associated with increased suspended sediment levels arising from disturbance of the seabed. The potential severity of the impacts varies depending on the type of fishing gear used and the extent, frequency and duration of the activity. The impact of boats is dealt with in association with marine structures below.

Structures (such as ports, harbours, bridges, slipways and piers) alter natural processes such as flow and silt movement and can therefore affect levels of suspended sediment in marine waters. The activities associated with these structures, for example shipping and boating, are associated with effects on the levels of general physico-chemical parameters, faecal coliforms, metals and chemicals.

Physical modifications (such as shoreline reinforcement, embankments and dredging) can alter natural processes such as flow and silt movement and can therefore affect levels of suspended sediment. However, once these modifications are established or the activities have ceased, the surrounding environment can acclimatise and impacts do not necessarily continue.

The following tables summarise the nature and extent of marine pressures up to a distance of 5 kilometres from the designated shellfish area. The likelihood for these pressures to impact on shellfish water quality parameters is discussed. The potential severity of the impacts of marine pressures is most closely associated with the activity type, magnitude and proximity and therefore the discussions in this section focus on these factors.

5.1.1 Point source pressures

There are no marine point source pressures in the vicinity of this designated shellfish area.

5.1.2 Morphology pressures

An assessment of the risk posed to marine waters from marine morphology pressures was carried out during the WFD implementation process. The results of this assessment show that the marine waters in and around this shellfish area are considered to be 'not at risk' from morphological pressures.

Fishing gear activity

TABLE 4 - Fishing gears

Fishing gear types	Type	Present	Comment
Pots	Static	Yes	Large areas within and adjacent to shellfish area
Tangle Nets	Static	No	NA
Bottom Set Gill Nets	Static	No	NA
Draft Nets	Static	No	NA
Drift Nets	Static	No	NA
Line Fishing	Static	Yes	Widespread throughout the area
Box Dredge	Mobile	No	NA
Cockle Dredge	Mobile	No	NA
Hydraulic Dredge	Mobile	Yes	Large areas within and adjacent to shellfish area
Scallop Dredge	Mobile	No	NA
Oyster Dredge	Mobile	Yes	Small area within shellfish area
Otter Trawl	Mobile	Yes	Large area within and adjacent to shellfish area
Beam Trawl	Mobile	No	NA
Digging	NA	No	NA
Gathering	NA	No	NA
Rake	NA	No	NA

Table 4 provides a summary of the fishing gear activity occurring within 5 kilometres of the designated shellfish area. Map 15 illustrates these pressures. Boat movements are dealt with below in association with marine structures such as ports and piers.

Static fishing gear types generally would not be expected to impact on shellfish water quality. Mobile fishing gears however disturb the seabed and can therefore affect the levels of suspended sediments in marine waters with the severity of the impacts depending on the frequency, intensity and extent of the fishing activity.

Static fishing gear activity in the area includes widespread line fishing (lines set on the seabed with bated hooks at intervals) and the use of pots (bated traps set on the seabed targeting crustaceans).

The use of mobile gear types includes the use of oyster dredges and hydraulic dredges within and adjacent to the shellfish area (metal blades which dig into the seabed to harvest shellfish) and the use of otter trawls within and adjacent to the shellfish area (nets towed along the seabed). Monitoring in the area does not indicate any water quality issues that are likely to be associated with the use of fishing gears and the morphology status of the water body within which the activity is occurring is 'high' (morphology is one of the elements of overall WFD status). In addition, the WFD assessment has deemed the area to be 'not at risk' from morphological pressures. Therefore, it is unlikely that fishing activity is affecting shellfish water quality in this shellfish area.

Structures and associated activities

TABLE 5 - Marine morphology structures

Marine morphology structures	Direct	0-5km	Comment
Ports	0	2	Howth fishing port, Malahide marina
Flow and sediment manipulation	0	19	Piers, slipways, breakwaters
Piled structures	0	4	Bridges & piers
Causeways	0	0	NA

Table 5 provides a summary of the marine morphology structures located within 5 kilometres of the designated shellfish area. Map 16 illustrates these pressures. Flow and sediment manipulation structures include piers, breakwaters, groynes, flow deflectors and training walls. Piled structures include bridge and pier supports and wind turbines. Causeways include roads and railway lines. These structures affect flow and sediment movement and can therefore impact on levels of suspended sediments, though these impacts can settle down once the structures are well established in an area. The activities associated with marine structures, including shipping and boating, can affect a wide range of water quality parameters including general physico-chemical parameters such as suspended sediment, dissolved oxygen and nutrient levels. Faecal coliform levels can also be affected as well as the levels of harmful substances such as metals and pesticides. Boat movements can lead to erosion and sedimentation effects as well as pollution from fuels.

There are no marine structures in the direct vicinity of this shellfish area although Howth fishing port is a couple of kilometres south of the area and there are 24 other marine structures within 5 kilometres, including Malahide marina as well as piers and slipways. Monitoring in the area does not indicate any water quality issues that are likely to be associated with these structures or their associated activities, the WFD morphology status of the water bodies within which the activity is occurring is 'high' and the WFD assessment has deemed the area to be 'not at risk' from morphological pressures. Therefore it is unlikely that the structures themselves or their associated activities are affecting shellfish water quality in this shellfish area.

Physical modifications

TABLE 6 - Physical modifications

Physical modifications	Direct	0-5 km	Comment
Shoreline reinforcement	0	24	NA
Embankments	0	0	NA
Reclaimed land	0	13	NA
Capital dredging	0	0	NA

Physical modifications	Direct	0-5 km	Comment
Maintenance dredging	0	2	Shipping channels
Aggregate removal	0	0	NA
Dumping at sea	0	0	NA

Table 6 provides a summary of the physical modifications occurring within 5 kilometres of the designated shellfish area. Map 17 illustrates these pressures. These modifications can affect flow and sediment movement though these impacts can cease once the modifications are established.

There are 24 instances of shoreline reinforcement and 13 areas of reclaimed land within 5 kilometres of this shellfish area. There are also 2 areas where maintenance dredging occurs within 5 kilometres of the shellfish area. Monitoring does not indicate any water quality issues which is likely to be associated with these modifications, the WFD morphology status of the water body within which the activity is occurring is 'high' and the WFD assessment has deemed the area to be 'not at risk' from morphological pressures. Therefore, it is unlikely that these modifications are affecting shellfish water quality in this shellfish area.

TABLE 7 - Heavily modified waters

HMWB name	Distance	Extent	Comment
Broadmeadow estuary	0-5	NA	NA

Table 7 lists the heavily modified marine waters located within 5 kilometres of the designated shellfish area. Map 17 illustrates these pressures. Such modifications can affect flow and sediment movements but the effects can cease once the modifications are established.

The Broadmeadow estuary, which is situated about 3 kilometres west of this shellfish area, has been designated as a heavily modified marine water body. Monitoring does not indicate any water quality issues which is likely to be associated with this modification, the WFD morphology status of the water body within which the activity is occurring is 'high' and the WFD assessment has deemed the area to be 'not at risk' from morphological pressures. Therefore, it is unlikely that this modification is affecting shellfish water quality in this shellfish area.

5.2 Land-based Pressures

The contributing catchment is used to identify the land-based pressures that could potentially be impacting on shellfish water quality and therefore the size of the contributing catchment can be important in determining the magnitude of the pressures. Contributing catchment sizes vary considerably; however, pressures are only considered up to a distance of 20 kilometres from the shellfish area and are, where appropriate, divided into four zones: direct, 0 to 5 kilometres, 5 to 10 kilometres and 10 to 20 kilometres. Pressures within the catchment, but further than 20 kilometres from the shellfish area, are also included if they are considered significant. In addition significant land-based pressures acting in adjacent waterbodies which may have an impact due to tidal influences are also considered where relevant.

Land-based pressure types include point source pressures, diffuse source pressures and morphology pressures. The shellfish water quality parameters potentially impacted by these pressures are as follows:

- Point source pressures can affect the whole suite of shellfish water quality parameters. For example, waste water treatment plants, CSOs and agricultural point sources can impact on the levels of faecal coliforms, nutrients, bacteria and other harmful substances in receiving waters while IPPC licensed industries, mines, quarries and landfills can impact on the levels of polluting substances in receiving waters such as petroleum hydrocarbons, organohalogenated substances and metals. Abstractions are included under this heading and can impact on salinity levels, though not to an extent likely to lead to non-compliances with shellfish water salinity standards, as well as reducing the dilution available for polluting discharges.

- Diffuse source pressures affect many of the shellfish water quality parameters. Agricultural activity and on-site waste water treatment systems (OSWTS) can impact on faecal coliform levels as well as general physico-chemical parameters such as the levels of suspended sediments and dissolved oxygen. Forestry activity can impact on the pH of receiving waters as well as on the levels of suspended solids and nutrients and it is also associated with the use of pesticides which can contain organohalogenated substances.

- Land-based morphology pressures, and associated activities, are not generally associated with impacts on water quality in marine areas. Their impacts are usually associated with the loss of natural freshwater features and habitats and changes to the behaviour of freshwater systems including sediment movement. Channelisation activities however, if occurring close to shellfish areas, can impact on shellfish water quality, particularly the levels of suspended sediment.

The following tables summarise the nature and extent of land based pressures within the catchment up to a distance of 20 kilometres from the designated shellfish area. The likelihood for these pressures to impact on shellfish water quality parameters is discussed. All of the factors discussed at the beginning of this chapter can affect the likelihood for land-based pressures to impact on shellfish waters.

5.2.1 Point Source Pressures

Urban Wastewater Systems

Table 8 lists the urban waste water treatment plants in the catchment up to a distance of 20 kilometres from the shellfish area. Map 18 illustrates these pressures and map references link the map and table. The information in the table was compiled by the WFD Municipal and Industrial Regulation Study in 2008 and includes:

- the distance of the plants from the shellfish area
- the WFD status of the water body within which the plants are located
- the level of treatment available at the plants
- whether the plants are included in the current Water Services Investment Programme 07-09
- the design capacity (in terms of population equivalents (P.E.)) of the plants
- the percentage at which the plants are operating above or below their design capacity currently
- the percentage at which the plants are likely to be operating above or below their design capacity in 2015 based on population projections
- the WFD risk designations associated with the plants and the reasons behind the risk designations

The WFD risk assessment in relation to urban waste water treatment plants was updated in 2008 to feed into the draft RBMPs with a further update currently underway (due for completion by November 2009). The plants were designated as 'at risk' for a variety of reasons including:

- A Insufficient WWTP capacity – existing load
- B Insufficient WWTP capacity – future load
- C Insufficient assimilative capacity for BOD – existing load
- D Insufficient assimilative capacity for BOD – future load
- E Insufficient assimilative capacity for nutrients – existing load
- F Insufficient assimilative capacity for nutrients – future load
- G Historical deterioration in downstream Q value where the Q station is within 3 kilometres of the outfall
- H Downstream Q value is less than 4 where the Q station is within 3 kilometres of the outfall
- I Deterioration in upstream to downstream Q value were the distance between Q stations is less then 3 kilometres
- J Exceedance of bathing water quality within 1 kilometre of the outfall
- K Exceedance of shellfish water quality within 1 kilometre of the outfall
- L Expert opinion

Waste water discharges from waste water treatment plants can contain a wide range of potentially polluting components originating from households, industry and urban areas. These discharges can affect the levels of faecal coliforms, nutrients, dissolved oxygen, suspended sediment, organic wastes and harmful chemicals in receiving waters.

The 2008 risk assessment identified 13 urban waste water treatment plants within the catchment with 9 of them 'at risk' for a range of reasons including insufficient plant capacity, insufficient assimilative capacity in receiving waters and deterioration in downstream water quality. The WFD risk assessment was reviewed by experts in November 2009 with regard to the Water Services Investment Programme and waste water licensing actions. The most significant plants were identified on the basis of proximity, plant performance, population equivalent and level of treatment. In this review, the plants at Malahide, Portrane/Donabate and Swords were identified as key plants in terms of the risk to shellfish water quality in this shellfish area.

Of the plants that are 'at risk', Malahide and Howth are by far the largest with design P.Es. of 20,000 and 30,000 respectively. They are located quite close to the shellfish area and, though they are operating within their design capacities, they are associated with failures of bathing water quality standards in receiving waters. Of the other plants that are 'at risk', Portrane and Lusk in particular are operating well in excess of their design capacity though both are scheduled for upgrade under the current Water Services Investment Programme.

Swords is the largest in the catchment with a design capacity of 60,000 P.E. This plant incorporates secondary treatment with nutrient removal. The Malahide plant has a design capacity of 20,000 P.E. and incorporates secondary treatment with nutrient removal and UV disinfection. The plant is included in the current Water Services Investment Programme 2007-2009. The plant at Portrane/Donabate has a design capacity of 8,000 P.E. and incorporates secondary treatment. The plant is included in the current Water Services Investment Programme and expansion of the scheme to a capacity of 65,000 P.E. is underway.

TABLE 8 - Urban waste water treatment plants

Name	Map Ref	Dist	Status	Treatment level	WSIP 07-09	Capacity P.E.	% surplus existing	% surplus future	At Risk
Balgriffin	258	0-5	Poor	nd	No	100	-800 %	-1,100 %	Yes – D/H
Ballyboghill	259	10-20	Poor	nd	No	250	-28 %	-41 %	Yes – C/D
Colecut	260	5-10	Moderate	nd	No	100	0 %	0 %	No
Lusk	264	0-5	nd	Primary	Yes	2,300	-204 %	-207 %	Yes – A/B/J
Malahide	265	0-5	nd	Secondary, nutrient removal, UV disinfection	Yes	20,000	35 %	28 %	Yes - J
North Dublin Drainage System - Howth	267	0-5	nd	nd	No	30,000	0 %	0 %	Yes - J
Oldtown	268	10-20	Poor	nd	No	500	56 %	52 %	Yes – C/D
Portrane/Donabate	269	0-5	nd	Secondary	Yes	8,000	-	-	Yes – J
Rowelstown	271	10-20	Poor	nd	No	100	0 %	0 %	Yes – H
Rush	272	0-5	nd	No treatment	Yes	7,800	7 %	-4 %	No
Swords	275	5-10	nd	Secondary plus nutrient removal	No	60,000	17 %	11 %	No
Toberburr	276	10-20	Poor	Secondary	No	640	0 %	0 %	Yes – C/D/G/H
Turvey	277	5-10	nd	nd	No	100	0 %	0 %	No

NOTE: A minus figure in the percentage surplus columns means that the plant is working above its design capacity, nd denotes 'no data' where for examples plants are located in areas with no WFD status information

Table 9 lists the Combined Sewer Overflows (CSOs) in the catchment up to a distance of 20 kilometres from the designated shellfish area. Map 18 illustrates these pressures and map references link the map and table. Information provided in the table in relation to the CSOs includes:

- the distance of the CSOs from the shellfish area
- the WFD status of the water body within which the CSOs are located

TABLE 9 - Combined Sewer Overflows

CSO Name	Map Ref	Distance	Status
Seafield Court	1	0-5 km	nd
Southshore Road	192	0-5 km	nd
Rogerstown Road	193	0-5 km	nd
Burrow Road	194	0-5 km	nd
Lissenhall Road	195	0-5 km	nd
St Ita's Hospital	196	0-5 km	nd
Donabate	197	0-5 km	nd
Portmarnock Strand	198	0-5 km	nd
Baldoyle Village	199	0-5 km	nd
Inbhir IDE	2	0-5 km	nd
Moyclare	201	0-5 km	nd
Burrow Road	204	0-5 km	nd
Craigview	205	0-5 km	nd
Claremont	206	0-5 km	nd
Ashbourne	235	10-20 km	Bad
Cuckoo Stream	254	5-10 km	Poor
Floraville	257	0-5 km	Poor
Castlefield Manor	258	0-5 km	Poor
Forest Road	260	5-10 km	Poor
Bridge Street	261	5-10 km	Poor
Glassmore Park	262	5-10 km	Poor
St Donagh's Road	362	5-10 km	nd
James Terrace	381	0-5 km	nd
James Terrace	382	0-5 km	nd
O'Hanlon's Lane	383	0-5 km	nd
No name	822	0-5 km	nd
Oldtown	87	10-20 km	Poor
Ashbourne	88	10-20 km	Bad
Portmarnock Bridge	93	0-5 km	Poor
Hole in the Wall Road	94	0-5 km	Poor
Mayne Bridge	95	0-5 km	nd

NOTE: nd means 'no data' where CSOs are located in areas with no WFD status information

Discharges from CSOs can contain a wide range of potentially polluting components originating from households, industry and urban areas. These discharges, which receive no treatment, can affect the levels of faecal coliforms, nutrients, dissolved oxygen, suspended sediment, organic wastes and harmful chemicals in receiving waters.

The inventory of CSOs compiled during the WFD characterisation process shows that there are 31 known significant CSOs within the catchment. Many of them are located very close to the shellfish area, within water bodies whose status is unsatisfactory.

CSOs are a possible source of the faecal contamination and elevated nutrient levels indicated by monitoring in the area and therefore they could possibly be affecting shellfish water quality in this shellfish area.

Abstractions

TABLE 10 - Abstractions

Name	Map Ref	Type	Distance	Status	Abs Rate $m^3\ day^{-1}$	At Risk (Ratio)
Roadstone	290	Groundwater	5-10	nd	50	No
Kilbridge National School	293	Groundwater	10-20	Moderate	1	No
IPPC 574	340	Groundwater	0-5	nd	295	No

NOTE: nd means 'no data' where abstractions are located in areas with no WFD status information

Table 10 lists the abstractions in the catchment up to a distance of 20 kilometres from the designated shellfish area. Map 20 illustrates these pressures and map references link the map and table. Information provided in the table in relation to abstractions includes:

- the type of abstraction (river, lake or groundwater)
- the distance of the abstraction from the designated shellfish area
- the WFD status of the water body within which the abstraction is located
- the abstraction rate, expressed in cubic metres per day
- the WFD risk designations associated with the abstractions and the reasons behind the designations

The WFD risk assessment in relation to abstractions was updated in 2008 to feed into the draft RBMPs. Abstractions are deemed to be 'at risk' if they account for a significant proportion (>10%) of the resource. For river abstractions, the net abstraction is expressed as a proportion of the Q95 flow (i.e. the flow that is exceeded 95% of the time). For lake abstractions, the net abstraction is expressed as a proportion of the Q50 inflow to the lake (i.e. the long term median inflow). For groundwater abstractions, the net abstraction is expressed as a proportion of recharge volume (i.e. long term average recharge across the groundwater bodies).

Generally it is very unlikely that abstractions would lead to non-compliances with the shellfish standards for salinity in shellfish areas. Abstractions that represent a large proportion of their corresponding resources can decrease available dilution capacity but this is also unlikely to affect shellfish areas.

There are 3 abstractions in the catchment. All 3 are groundwater abstractions and none of them are 'at risk' and, since they don't represent a significant proportion of their corresponding groundwater resources, they are unlikely to affect any aspect of shellfish water quality in this shellfish area.

Integrated Pollution Prevention and Control Industries

TABLE 11 - Integrated Pollution Prevention Control Licenses

Name	Map Ref	Distance	Status	Reasons for risk
Evode Industries Ltd 1 (Construction)	76	5-10 km	Poor	Yes – G/H
Evode Industries Ltd 2 (Construction)	77	5-10 km	Poor	Yes – G/H
Huntstown (Power station)	78	10-20 km	Poor	Yes – C/D/E/F

Table 11 lists the IPPC licensed industries in the catchment up to a distance of 20 kilometres from the designated shellfish area. Map 20 illustrates these pressures and map references link the map and table. Information provided in the table in relation to the licensed industries includes:

- the distance of the industries from the designated shellfish area
- the WFD status of the water bodies within which the industries are located
- the WFD risk designations associated with the industries and the reasoning behind the designations

The WFD risk assessment in relation to IPPC licensed industries was updated in 2008 to feed into the draft RBMPs. The industries were designated as 'at risk' for a variety of reasons which are outlined on page 57.

Discharges from IPPC licensed industries are diverse and can affect the levels of faecal coliforms, nutrients, suspended sediments, dissolved oxygen as well as a wide range of chemicals in receiving waters.

There are 3 IPPC licensed industries within the catchment and all of them have been designated as 'at risk' for various reasons including inadequate assimilative capacity in receiving waters and deterioration in downstream water quality. However, none of them are a likely source of the elevated levels of faecal coliforms and nutrients indicated by shellfish and WFD monitoring, and therefore they are unlikely to be affecting shellfish water quality in this shellfish area.

Section 4 Licensed Industries

TABLE 12 - Section 4 Licenses

Name	Map Ref	Distance	Status	Risk
Abbey Commercial Parks	196	5-10 km	Poor	No
Aer Rianta	197	5-10 km	Poor	No
Country Crest	201	5-10 km	Moderate	No
Department of Education	202	5-10 km	Moderate	No
Donabate Gold Club	203	0-5 km	nd	No
East Vocational Enterprises Ltd	206	5-10 km	nd	No
Emmaus Retreat Centre	208	5-10 km	Poor	No
Hanover's Tavern	211	10-20 km	Poor	No
Irish Asphalt Ltd	213	10-20 km	Moderate	No
Roadstone Feltrim	218	5-10 km	nd	No
Roadstone Huntstown	219	10-20 km	Poor	No
Superdawn Ltd	222	5-10 km	Moderate	No

NOTE: nd means 'no data' where industries are located in areas with no WFD status information

Table 12 lists the Section 4 licensed industries in the catchment up to a distance of 20 kilometres from the designated shellfish area. Map 20 illustrates these pressures and map references link the map and table. Information provided in the table in relation to the industries includes:

- the distance of the industries from the designated shellfish area
- the WFD status of the water bodies within which the industries are located
- the WFD risk designations associated with the industries and the reasoning behind the designations

The WFD risk assessment in relation to Section 4 licensed industries was updated in 2008 to feed into the draft RBMPs. The industries were designated as 'at risk' for a variety of reasons which are outlined on page 57.

Discharges from Section 4 licensed industries are diverse and can affect the levels of faecal coliforms, nutrients, suspended sediments, dissolved oxygen as well as a wide range of chemicals in receiving waters.

There are 12 Section 4 licensed industries in the catchment but none of them have been deemed to be 'at risk'. It is therefore unlikely that these industries are affecting shellfish water quality in this shellfish area.

Quarries, mines, landfills and contaminated lands

TABLE 13 - Quarries, mines, landfills and contaminated lands

Name	Map Ref	Distance	Status	Risk	Notes
Hollywood Quarry	55	5-10 km	Moderate	No	Quarry
Roadstone Feltrim Quarry	59	5-10 km	nd	No	Quarry
Roadstone Huntstown Quarry	60	10-20 km	Poor	No	Quarry
Fingal County Council	2	5-10 km	nd	No	Unlined landfill
Murphy Concrete Manufacturers	21	5-10 km	Moderate	No	Lined landfill
Dublin County Council	38	0-5 km	nd	No	Unlined landfill
Diamond Innovations Irish Operations	24	5-10 km	Poor	No	Contaminated land – chlorine, ammonium
Global Switch Property Dublin Ltd	25	5-10 km	Poor	No	Contaminated land - oil

Name	Map Ref	Distance	Status	Risk	Notes
Arch Chemicals	29	5-10 km	Poor	No	Contaminated land – chloroperidine, ammonia

NOTE: nd means 'no data' where operations are located in areas with no WFD status information

Table 13 lists the quarries, mines, landfills and contaminated lands in the catchment up to a distance of 20 kilometres from the designated shellfish area. Map 20 illustrates these pressures and map references link the map and table. Information provided in the table in relation to the plants includes:

- the distance of the industries from the designated shellfish area
- the WFD status of the water bodies within which the plants are located
- the WFD risk designations associated with the industries

Some of the WFD risk assessments in relation to these point sources were updated in 2008 to feed into the draft RBMPs but some of the assessments date back to the WFD characterisation process in 2004 and 2005. Expert opinion within Local Authorities was used to assign risk designations to quarries and landfills but monitoring data was used for mines and contaminated lands.

Mining and quarrying operations can impact on levels of suspended solids and metals in receiving waters whilst landfills and contaminated sites can be more diverse and impact on the levels of nutrients, suspended sediments and oxygen levels as well as metals and other chemicals.

There are 3 quarries, 3 landfills and 3 contaminated lands within the catchment but none of them have been designated as 'at risk' of impacting their surrounding water environment. Therefore, they are unlikely to be affecting shellfish water quality in this shellfish area

5.2.2 Diffuse Source Pressures

On-site waste water treatment systems

TABLE 14 - On-site waste water treatment systems

Risk	Number	% of total
Total number	5,181	-
Number per km^2 in the catchment	13.76	-
Number per km^2 nationally	1.4	-
Number that are high risk to surface waters from pathogens	4,907	94.71%
Number that are high risk to groundwaters from pathogens	617	11.9%
Number that are high risk to surface waters from phosphorus	4,289	82.78%
Number that are high risk to groundwaters from phosphorus	510	9.84%
High likelihood of inadequate percolation of leachate	4,838	93.37%

Table 14 summarises the numbers of on-site waste water treatment systems (OSWWTS) within the catchment up to a distance of 20 kilometres from the designated shellfish area and outlines how many of them are located in areas of high risk to surface and groundwaters from pathogens and phosphorus and how many of them are located in areas where the likelihood of inadequate percolation of leachate is high. Map 21 illustrates the locations of the OSWWTSs while Maps 6 to 10 illustrate the risk to surface and groundwaters and the likelihood of inadequate percolation, all of which is based on soil, sub-soil and geological characteristics. Generally, systems located in areas where effluent cannot get away underground pose a risk to surface waters while systems located in areas where the effluent moves too quickly through the subsoil pose a risk to groundwaters. OSWWTS effluent can impact on the levels of faecal coliforms, suspended sediments, nutrients and dissolved oxygen in receiving waters. In addition, the use of household cleaning products can introduce a range of harmful chemicals to the water environment.

There are 5,181 systems in the contributing catchment and their density is much higher than the national average. The risk to surface water from pathogens high throughout the catchment as is the likelihood of inadequate percolation. The majority of the systems are therefore located in hydrologically unsuitable conditions. Other factors which affect the likelihood of these systems to impact surface and groundwaters are whether suitable types of systems are selected, whether they are installed correctly, whether they are properly maintained and whether they are situated close to the designated shellfish area or to ditches, drains, watercourses, wells or boreholes. Therefore, it is likely that a substantially smaller number than the total number of systems in the catchment are posing a risk to surface and groundwaters. Monitoring indicates faecal contamination and elevated nutrient levels in this shellfish area which could be arising from this source. These systems therefore could possibly be affecting shellfish water quality in this shellfish area.

Agriculture

TABLE 15 - Livestock units and chemical fertiliser usage

Indicator	Catchment (per ha of farmed land)	National Average (per ha of farmed land)
Livestock units	0.91 LU	1.20 LU
Nitrogen fertiliser usage	126.87 kg	92.09 kg

Indicator	Catchment (per ha of farmed land)	National Average (per ha of farmed land)
Phosphorus fertiliser usage	15.02 kg	9.74

Nitrates Directive limit = 170 kg N per hectare = approx. 2 LU per hectare
Nitrates Directive derogation = 250 kg N per hectare = approx. 3 LU per hectare.

Table 15 provides an estimate of the average number of dairy and drystock livestock units and the average loadings of nitrogen and phosphorus chemical fertiliser per hectare of farmed land within the contributing catchment area. Maps 22, 23 and 24 illustrate this. The figures beneath the table express the nitrate limit (and Ireland's derogation) under the Nitrates Directive in terms of livestock densities. Discharges related to agriculture can affect the levels of faecal coliforms, suspended sediments, nutrients and dissolved oxygen in receiving waters. In addition, the use of pesticides and herbicides can introduce a range of harmful chemicals to the water environment.

Less than 20% of the area of this catchment is farmed land. The estimate of livestock density is lower than the national averages whereas the estimates of fertiliser usage are higher than the national averages. The EPA's diffuse model risk assessment, which investigates the relationship between catchment attributes (percentages of diffuse land cover including agriculture), water chemistry and ecological status, deems the whole catchment to be at risk areas (Map 13). There are many areas of wet soils within the catchment (Map 5) where there is a potential risk of agricultural runoff. As agriculture is a possible source of the faecal contamination and elevated nutrient levels indicated by monitoring in the area, agriculture could possibly be affecting shellfish water quality in this shellfish area.

Forestry

TABLE 16 - Forestry types

Type	Area	Percentage of area
Conifers	0.16 km^2	0.04 %
Broadleaves	1.45 km^2	0.4 %
Mixed	0.79 km^2	0.2 %
Other	0 km^2	0 %
Cleared	0.06 km^2	0.02 %
Unknown	0.02 km^2	0.01 %
Total	2.48 km^2	0.7 %
Nationally	6,795 km^2	10.0 %

Table 16 presents the area and percentage area of the catchment under the various types of forest cover. Maps 25, 26 and 27 illustrate this. Forestry activity can impact on the pH of receiving waters as well as on the levels of suspended solids and nutrients. It is also associated with the use of pesticides which can introduce harmful chemicals to the water environment.

This is 2.48 km^2 of forested land in this catchment and percentage area under forest cover is very low compared to the national average. Unlike agriculture, the location of forestry activity is known and very little forestry activity occurs in close proximity to the shellfish area. The EPA's diffuse model risk assessment, which investigates the relationship between catchment attributes (percentages of diffuse land cover including forestry), water chemistry and ecological status, highlights diffuse risk areas in the catchment (Map 13). However, the more recent risk assessment, undertaken by the WFD Forest and Water study, does not highlight any areas of acidification, eutrophication and sedimentation risk (Maps 25, 26 and 27). Overall, mainly due to the very low levels of forestry in the catchment, forestry is unlikely to be affecting shellfish water quality in this shellfish area.

5.2.3 Morphology Pressures

Structures

TABLE 17 - Natural and man-made barriers

Freshwater morphology structures	Number	Dist	Comment
Barriers to migration	1	5-10 km	Artificial

Table 17 summarises the occurrences of morphological structures within the contributing catchment area up to a distance of 20 kilometres from the designated shellfish area. Map 28 illustrates this. Any impacts associated with barriers, which could include impacts on flow, sediment movement and fish migration, are likely to be localised.

There is 1 artificial barrier to fish migration within the catchment but it is not situated in the vicinity of the shellfish area. It is therefore unlikely to be affecting shellfish water quality in this shellfish area.

Physical Modifications

TABLE 18 - Channelisation

Physical modification	Extent	Comment
Channelisation	129 km	River Broadmeadow

Table 18 summarises the occurrences of channelisation within the contributing catchment area up to a distance of 20 kilometres from the designated shellfish area. Map 29 illustrates this. Channelisation, if it occurs reasonably close to a shellfish area, can affect suspended sediment levels in the shellfish area while it is taking place.

The Broadmeadow river system is extensively channelised with 129 kilometres of channel affected, some of it adjacent to the area. This activity could therefore affect shellfish water quality while it was taking place.

5.3 Summary of Key Pressures

Information from existing data sources has been used to identify all of the pressures acting on the shellfish area and to assess their likelihood to be affecting shellfish water quality in this shellfish area.

The status at this site is impacted by faecal coliforms which is indicative of sewage related key pressures. Dissolved inorganic nitrogen status issues are also identified in the general area.

This summary section highlights:

- **key pressures**

The key pressures are those identified as most likely to be affecting shellfish water quality. The final PRP will confirm and focus on these key pressures.

- **potential secondary pressures**

These pressures are identified as possibly affecting shellfish water quality. The final PRP will either confirm them as key pressures or eliminate them from further consideration.

5.3.1 Key Pressures

1. Urban wastewater systems

The 2008 risk assessment identified 13 urban waste water treatment plants within the catchment with 9 of them 'at risk' for a range of reasons including insufficient plant capacity, insufficient assimilative capacity in receiving waters and deterioration in downstream water quality. The WFD risk assessment was reviewed by experts in November 2009 with regard to the Water Services Investment Programme and waste water licensing actions. The most significant plants were identified on the basis of proximity, plant performance, population equivalent and level of treatment. In this review, the plants at Malahide, Portrane/Donabate and Swords were identified as key plants in terms of the risk to shellfish water quality in this shellfish area.

Of the plants that are 'at risk', Malahide and Howth are by far the largest with design P.Es. of 20,000 and 30,000 respectively. They are located quite close to the shellfish area and, though they are operating within their design capacities, they are associated with failures of bathing water quality standards in receiving waters. Of the other plants that are 'at risk', Portrane and Lusk in particular are operating well in excess of their design capacity though both are scheduled for upgrade under the current Water Services Investment Programme.

Swords is the largest in the catchment with a design capacity of 60,000 P.E. This plant incorporates secondary treatment with nutrient removal. The Malahide plant has a design capacity of 20,000 P.E. and incorporates secondary treatment with nutrient removal and UV disinfection. The plant is included in the current Water Services Investment Programme 2007-2009. The plant at Portrane/Donabate has a design capacity of 8,000 P.E. and incorporates secondary treatment. The plant is included in the current Water Services Investment Programme and expansion of the scheme to a capacity of 65,000 P.E. is underway.

The inventory of CSOs compiled during the WFD characterisation process shows that there are 31 known significant CSOs within the catchment. Many of them are located very close to the shellfish area, within water bodies whose status is unsatisfactory. CSOs are a possible source of the faecal contamination and elevated nutrient levels indicated by monitoring in the area and therefore they could possibly be affecting shellfish water quality in this shellfish area.

2. On-site waste water treatment plants

There are 5,181 systems in the contributing catchment and their density is much higher than the national average. The risk to surface water from pathogens high throughout the catchment as is the likelihood of inadequate percolation. The majority of the systems are therefore located in hydrologically unsuitable conditions. Other factors which affect the likelihood of these systems to impact surface and groundwaters are whether suitable types of systems are selected, whether they are installed correctly, whether they are properly maintained and whether they are situated close to the designated shellfish area or to ditches, drains, watercourses, wells or boreholes. Therefore, it is likely that a substantially smaller number than the total number of systems in the catchment are posing a risk to surface and groundwaters. Monitoring indicates faecal contamination and elevated nutrient levels in this shellfish area which could be arising from this source. These systems therefore could possibly be affecting shellfish water quality in this shellfish area.

5.3.2 Potential Secondary Pressures

3. Agriculture

Less than 20% of the area of this catchment is farmed land. The estimate of livestock density is lower than the national averages whereas the estimates of fertiliser usage are higher than the national averages. The EPA's diffuse model risk assessment, which investigates the relationship between catchment attributes (percentages of diffuse land cover including agriculture), water chemistry and ecological status, deems the whole catchment to be at risk areas (Map 13). There are many areas of wet soils within the catchment (Map 5) where there is a potential risk of agricultural runoff. As agriculture is a possible source of the faecal contamination and elevated nutrient levels indicated by monitoring in the area, agriculture could possibly be affecting shellfish water quality in this shellfish area.