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# The Use of Transport Time Scales as Indicators of Pollution Persistence in a Macro-Tidal Setting

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**Abstract:** An understanding of water exchange processes is essential for assessing water quality management issues in coastal bays. This paper evaluates the impact of water exchange processes on pollution persistence in a macro-tidal semi-closed coastal bay through two transport time scales (TTS), namely residence time and exposure time. The numerical model was calibrated against field-measured data for various tidal conditions. Simulated current speeds and directions were shown to agree well with the field data. By considering different release scenarios of a conservative tracer by the refinement of an integrated hydrodynamic and solute transport model (the EFDC), the two TTS were used for interpreting the water exchange processes in a semi-closed system, and for describing the effects of advective and dispersive processes on the transport and fate of pollutants. The results indicate that the magnitudes of river inflows to the bay, tidal ranges, and tracer release times significantly influence the residence and exposure times. Return coefficients were shown to be variable, confirming the different effects of returning water for the different conditions that were studied. For the tested river flow magnitudes and tide conditions, the exposure times were generally higher than the residence times, but particularly so for neap tide conditions. The results, therefore, highlight the risks associated with pollutants leaving a specified domain on an outgoing tide but re-entering on subsequent incoming tides. The spatial distributions of the exposure and residence times across the model domain confirmed that for the case of Dublin Bay, river inputs have a potentially greater impact on water quality on the northern side of the bay.

**Keywords:** pollution persistence; health risk; bathing waters; macro-tidal estuary; transport time scales; residence and exposure time



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## 1. Introduction

The need to better understand the behaviour of pollution loads in our coastal waters is increasingly being recognised [1] and remains critical to decision-making that underpins our better management of the changing pressures in these environmental systems. The quality of water in these systems is typically influenced by the presence of both diffuse (often transported from catchments through river systems that discharge to coastal waters) and point source pollutants. Water exchange processes are central to the transport and fate of various physical, chemical, and biological water quality indicators that serve as quality metrics for the recreational and commercial activities in these waters. Semi-closed coastal systems, such as bays, estuaries, and lagoons, are characterised by complex and dynamic transport and mixing processes. The nature of these systems, which are spatially and temporally variable, presents challenges to understanding and quantifying these important processes.

Transport time scales (TTS) are sometimes adopted by water managers and engineers as indexes for interpreting the flow in these semi-closed systems and for describing the effects of advective and dispersive processes on the transport and fate of pollutants [2]. The effects of pollutants on receiving water systems are not only determined by pollutant loads but also retention times, which are related to TTS, which also play a role [1]. TTS are key parameters in understanding the impacts of hydrodynamic processes on the fate and transport of pollutants and are important when studying water exchange processes that influence the environmental impacts of pollutants in receiving waters. Different TTS are reported in the scientific literature to describe distinctive aspects of the water exchange process. Commonly used TTS are defined in Table A1 (Appendix A) and include the residence time [3], exposure time [4,5], flushing time [3,6], turnover time [7,8], influence time [9], and water age [10].

Among these TTS, some, such as the flushing time and turnover time, describe global or bulk time scales [11], while the residence time and exposure time provide space- and time-dependent quantitative measures of the rate at which water masses or pollutants enter and/or are removed from a controlled domain [9]. The focus of TTS studies in semi-closed water bodies, as reported in the scientific literature, has shifted from global and bulk timescales to TTS, which are generally considered more informative and suitable for characterising spatial variability [11], with residence and exposure times being commonly used in this regard.

The residence time in a controlled domain defines the time taken for water parcels, solutes, or particulate matter to leave that domain (for the first time). By contrast, the exposure time is a measure of the total time spent by a water parcel or materials in a controlled domain over a defined study period, including the time intervals of subsequent re-entries (and further exits) of parcels and materials to and from the domain [4,5]. Exposure times are, therefore, often considered in terms of residence and return retention times [2] to reflect the returning effects of water parcels or materials. The residence time, conversely, reflects a more limited measure of the time expended by a water parcel or materials in a semi-closed system and, by definition, excludes the time spent by water parcels or materials in the domain following the initial exit from the domain [5]. This distinction in definition can result in substantial differences between the calculated residence and exposure times, particularly when applied to tidal systems where water exchanges promote the exit of water parcels or materials from the domain on ebbing tides and their re-entry to the domain on incoming or flooding tides. In pollution assessments of tidal systems where multiple tidal cycles would typically be considered, the exposure time arguably represents a more useful and realistic risk index. The exposure time can be decomposed into the residence time and return retention time [2] to quantify the returning effects, which is important in understanding the transport and fate of pollutants in a tidal water body.

While some studies have considered both residence and exposure times in parallel as a way of separating and quantifying the returning water effects in a controlled domain [4,5,12], these have typically focused on deeper waters with a variety of tidal ranges. The combination of a shallow water depth with a relatively high tidal range reflects characteristics of coastal domains that, while less common, are important. However, studies of TTS in shallow bays and estuaries with macro-tidal ranges are limited in the scientific literature, and observations and conclusions from deep-water study sites are not considered transferable to such systems. Given that the exchange processes in these shallow water/ high tidal range settings are likely to be considerably different from those in deeper waters, studies of shallow domains are, therefore, needed to advance our understanding of the complex water exchange processes in these systems for improved decision-making pertaining to the more effective and efficient water quality management of these systems.

Here, we present a numerical investigation in which residence and exposure times are simultaneously investigated in Dublin Bay (Ireland) to characterise TTS in a shallow bay with macro-tidal ranges. The effects of river discharges from adjacent catchments, different tidal ranges, and the release times of conservative tracers were considered in determining exposure and residence times, and the spatial distribution of TTS was considered to assess the effects of these influences in different regions of the bay. The return coefficients were evaluated for these different modelling scenarios to quantify the effects of the returning water on the water exchange processes and TTS. The findings of the study will be of interest to managers of estuarine and coastal waters and are, we believe, transferrable to other shallow-water coastal zones in which large tidal fluctuations have a significant influence on hydrodynamic characteristics.

## 2. Materials and Methods

An integrated hydrodynamic and solute transport model (the EFDC) was used in this study. The numerical model was first calibrated and validated against measured field data for various tidal conditions and was subsequently refined such that the TTS of concern (exposure and residence times) could be determined for test conditions in Dublin Bay. The key test conditions were those that drive the water exchanges in Dublin Bay and included simulations of different tidal conditions, river flow magnitudes, and tracer release times (described fully in the sections that follow). The results analysis was based on a comparison of the model outputs for these different test conditions.

### *2.1. Study Area*

Dublin Bay, located on the east coast of Ireland, is a C-shaped water body bounded to the west by Dublin City and to the east by the Irish Sea, with its northern and southern extents being defined by Howth Head and Dalky, respectively (Figure 1).

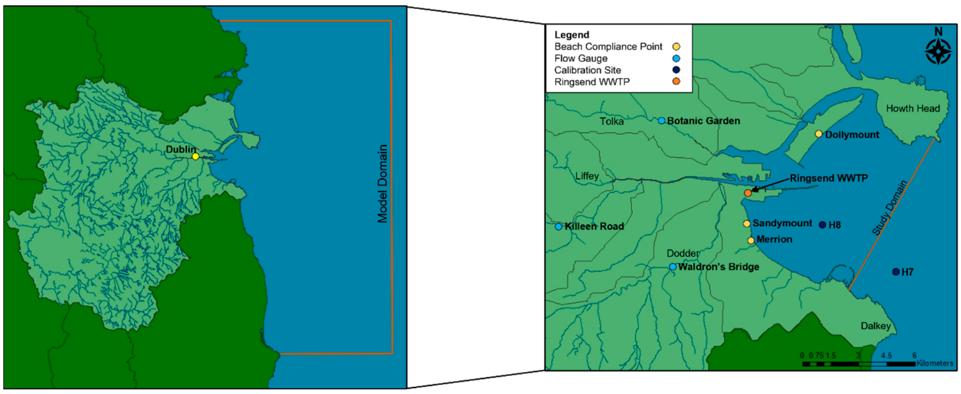


**Figure 1.** Study area: Dublin Bay and the modelled domain.

The bay, which was designated a 'biosphere reserve' by UNESCO in 2015 in recognition of its unique ecological habitat and biological diversity, extends for c. 10 km from its mouth in an easterly direction and covers an area of approximately 100 $km^2$. The bay receives major freshwater inflows from the Rivers Liffey and Tolka, with the River Dodder joining the Liffey a short distance upstream of its outfall to Dublin Bay. The inner portion of the bay that includes the Liffey Estuary covers an area of c. 5 $km^2$ and extends eastwards (for c. 4 km) from the mouth of the river to the extremities of the North and South Bull Walls. Apart from a central navigation channel through this estuary to the Dublin Port, where water depths are c. 8 m, the depths are relatively shallow, being typically less than three metres. Beyond the estuary, the water depth gradually increases, reaching a depth of c. 10 m at the boundary of the study domain (Figure 1). The bay is macro-tidal with a maximum tidal range of more than 4.0 m and with average mean spring and neap tides of 3.6 m OD and 1.9 m OD, respectively [13,14]. The River Liffey is controlled by an upstream hydroelectric plant and dam, resulting in a reasonably regulated inflow. The

Ringsend wastewater treatment plant (WWTP), located on the southern side of the estuary, currently provides preliminary, primary, secondary, and tertiary wastewater treatment to a population equivalent (PE) of 1.9 million and treats c. 40% of Ireland's sewage. UV disinfection is implemented at the plant during the bathing water season (May to September each year) to aid compliance with the EU Bathing Water Directive (2006/7/EC) standards for faecal indicator bacteria (Escherichia coli and intestinal enterococci (IE) in the designated bathing waters around the bay [15].

Notwithstanding the current capacity of the Ringsend WWTP, times of heavy rainfall generate inflows to the plant that exceed the treatment's limits, resulting in overflows to Dublin Bay. While upgrade works to increase the plant's treatment capacity to a 2.4 m PE (with a firm capacity of a 2.1 m PE) are underway, risks of overflows are likely to exist until the upgrades are operational. In the context of these issues, Dublin Bay represents a suitable testbed to explore the water exchange processes and TTS that are central to understanding water quality issues and the associated pressures these have on the biodiversity and ecosystems in the bay and its surrounding areas.

The study area of Dublin Bay is highly dynamic, and TTS, including both residence and exposure times, are likely to be influenced by multiple variables, including the release time of pollutants, the prevailing tidal conditions in the bay, and the antecedent upstream catchment conditions, which will influence river flows to the bay. To fully capture this variability, the modelling scenarios and numerical experiments included an assessment of spring and neap tides and different inflow conditions from the Rivers Liffey, Dodder, and Tolka, which discharge directly to Dublin Bay.

### 2.2. Calculation of Residence and Exposure Times and Return Coefficients

Residence time is defined as the time taken from a given point for a water parcel or materials to reach the outlet or leave a study domain for the first time [16]. Following from this definition, the remnant function adopted in this study was developed to represent the fraction of the initial mass of material whose residence time is greater than or equal to $t$ [4,5,17,18]. The remnant function, $r(t)$, is defined as:

$$r(t) = \frac{M(t)}{M(t_0)} \quad (1)$$

where $M(t_0)$ is the total amount of material in the study domain at the initial time, and $M(t)$ is the amount of material that stays continuously (without leaving) in the domain after the initial time, $t$. The residence time, $T_r$, or exposure time, $T_e$, can then be defined as:

$$T_r = \int_{t_0}^{+\infty} r_r(t)dt \text{ or } T_e = \int_{t_0}^{+\infty} r_e(t)dt \quad (2)$$

where $T_r$ and $T_e$ are the residence time and exposure time, respectively, and $r_r(t)$ and $r_e(t)$ are the respective remnant functions.

As already mentioned, the residence time is useful for characterising the time scale of the water domain if the water parcel is not going to return to the water domain after reaching the outlet (typical for water parcels entering rivers and lakes). However, in estuaries, transitional and coastal domains subject to tidal cycles, water parcels may return to the water domain after leaving. The exposure time represents an alternative time scale parameter that reflects these more complex exchanges [4,5,9,19–21]. Both residence and exposure times are considered in this study.

Exposure and residence times in the Dublin Bay study domain were estimated using a numerical model in which a passive conservative tracer was used to define the transport processes. At the commencement of the simulations, the conservative tracer concentration (Equation (3) in the domain of interest (shown in Figure 1) was set to 1, and to 0 elsewhere. The residence time of the water inside the bay was determined as the time taken by a water parcel to leave the study domain for the first time. Residence times were calculated (using

Equations (2) and (4) by setting tracer concentrations (S) to zero, when the water parcels reached the study domain boundary at the mouth of the bay (following the approach in [5]):

$$S(x,y,t_0) = \begin{cases} 1(x,y) \in \Omega \\ 0(x,y) \notin \Omega \end{cases} \quad (3)$$

$$r(t) = \frac{\int_\Omega H(x,y,t) \cdot S(x,y,t) d\Omega}{\int_\Omega H(x,y,t_0) \cdot S(x,y,t_0) d\Omega} \quad (4)$$

where $H(x,y,t)$ and $S(x,y,t)$ are the water depth and tracer concentration, respectively, at location $(x,y)$ and at time $t$, and $\Omega$ defines the domain of interest.

The return coefficient was defined in previous studies to quantify the re-entry of the water to the studied domain for a given transport time scale and is adopted in this study to represent the effects of the returning water parcel and tracer to the region of interest, $\Omega$, after leaving the domain:

$$C_r = \frac{T_e - T_r}{T_e} \quad (5)$$

where $C_r$ is the return coefficient representing the contribution of returning water in the calculation of exposure times.

*2.3. Hydrodynamic and Dispersion Model*

Exposure and residence times in the Dublin Bay study domain were estimated using the Environmental Fluid Dynamics Code (EFDC) [22]. The model, which utilises a boundary-fitted curvilinear grid in the horizontal direction and sigma grids in the vertical direction, is well established and is used extensively in 1-D, 2-D, and 3-D studies of coastal domains [1,2,22–24]. While some stratification of the water column was previously reported in the Liffey Estuary [13], levels were low, with salinity and temperature changes between the bed and surface being less than 1.5 PSU (between 32 and 34 PSU) and 1 °C, respectively. Furthermore, these low levels of stratification were localised, being observed near the Ringsend WWTP effluent discharge point, and a cooling water outflow from a local power plant. East of the estuary, a well-mixed water body with no evidence of stratification was observed. Given that the current study is concerned with bulk water exchanges across the Dublin Bay domain (Figure 1), the 2-D depth-averaged EFDC model was considered appropriate to simulate the transport of a passive, conservative tracer in Dublin Bay for the hydrodynamic and exchange processes pertaining to the different tide and flow conditions that were tested. Wind can impact transport processes in small coastal bays by influencing horizontal circulation patterns in estuarine and coastal waters [10]. However, studies in Dublin Bay [13,14] indicate that prevailing winds serve to push pollutant plumes in the dominant wind direction. Rather than change the overall circulation patterns and directions, wind effects in Dublin Bay are, therefore, likely to affect circulation and water exchange processes locally, and in the context of a study examining the bulk characteristics of water exchange, such effects are expected to be minimal. The simulations facilitated the calculation of the relevant residence and exposure times and return coefficients. The governing equations of momentum and continuity, together with the transport equations for conservative tracers, are summarised as:

$$\begin{aligned} &\partial_t(mHu) + \partial_x(m_y Huu) + \partial_y(m_x Hvu) + \partial_z(mwu) - (mf + v\partial_x m_y - u\partial_y m_x)Hv \\ &\quad = -m_y H\partial_x(g\xi + p) - m_y(\partial_x h - z\partial_x H)\partial_z p + \partial_z(mH^{-1}A_v\partial_z u) + Q_u \end{aligned} \quad (6)$$

$$\begin{aligned} &\partial_t(mHv) + \partial_x(m_y Huv) + \partial_y(m_x Hvv) + \partial_z(mwu) + (mf + v\partial_x m_y - u\partial_y m_x)Hu \\ &\quad = -m_x H\partial_y(g\xi + p) - m_x(\partial_y h - z\partial_y H)\partial_z p + \partial_z(mH^{-1}A_v\partial_z v) + Q_v \end{aligned} \quad (7)$$

$$\partial_z p = -gH(\rho - \rho_0)\rho_0^{-1} = -gHb \quad (8)$$

$$\partial_t(m\xi) + \partial_x\left(m_y Hu\right) + \partial_y(m_x Hv) + \partial_z(mw) = 0 \quad (9)$$

$$\partial_t(mHS) + \partial_x\left(m_y HuS\right) + \partial_y(m_x HvS) + \partial_z(mwS) = \partial_z\left(mH^{-1}A_v\partial_z S\right) + Q_s \quad (10)$$

where $u$ and $v$ are the horizontal velocity components in the curvilinear, orthogonal coordinates, $x$ and $y$, $m_x$ and $m_y$ are the square roots of the diagonal components of the metric tensor, and $m = m_x m_y$ is the Jacobian or square root of the metric tensor determinant; $H$ is the total water depth, and $\xi$ is the water surface elevation. In the momentum equations (Equations (6) and (7), $f$ is the Coriolis parameter, $A_v$ is the vertical turbulent or eddy viscosity, and $Q_u$ and $Q_v$ are momentum source–sink terms, which are later modelled as the sub-grid scale horizontal diffusion. The relative hydrostatic pressure in the water column is defined by $p$, $S$ is the conservative tracer concentration and $\rho$ and $\rho_0$ are the actual and reference water densities, respectively.

The numerical solution of the EFDC model involves a finite volume–finite difference spatial discretisation on a C-grid-staggering of the discrete variables, which is further described in [22].

The model domain in the current study extends for a distance of 40 km in the east–west direction and a distance of 63 km in the north–south direction (Figure 1). The domain was discretised using an extensive bathymetric data set to form a mesh of 10,772 elements, ranging in size from 900 m at the open-sea boundary to 65 m in the inner part of the estuary.

The model was driven by time-varying water levels along the open-sea boundaries of the modelled domain. Locating the open-sea boundaries a significant distance from the primary study area ensured that their influence on the numerical solution was minimised. The water levels along the open-sea boundaries were based on the major diurnal and semi-diurnal tidal constituents generated from the DHI global tidal database [25] but which were improved for the nearshore area of interest in an extensive calibration routine where the amplitude and phase lags of the two largest constituents (M2 and S2) were correlated to tidal records at five reference gauges [13]. Wave effects on the hydrodynamic processes were not considered in the current model due to the unavailability of high-resolution spatial wave data in the study area that would be required to assess such effects on the model dispersion, such as in the study of Truong et al. [26].

The horizontal eddy viscosity, $A_H$, was calculated using the Smagorinsky sub-grid scale scheme, which can be written as [27]:

$$A_H = C\Delta x\Delta y\left[\partial u_x^2 + \partial v_y^2 + \frac{1}{2}\left(\partial u_y + \partial v_x\right)^2\right]^{\frac{1}{2}} \quad (11)$$

where $C$ is the horizontal mixing constant (with typical values between 0.10 and 0.20) and where $\Delta x$ and $\Delta y$ define the model's grid size in the $x$ and $y$ directions, respectively. In this study, a value for $C$ of 0.2 was adopted.

Hydrodynamic simulations were performed for both spring and neap tide conditions. The hydrodynamic characteristics of the model were calibrated by adjusting the bed friction such that simulated current speeds, directions, and water depths were well correlated to measurements at locations H7 and H8 (Figure 1) for both spring and neap tidal cycles. Data confirming that simulated current speeds, directions, and water depths are in good agreement with the measured data are shown in Figure 2 for location H7 and in Figure 3 for location H8. In addition, typical velocity distributions in the study domain are shown in Figure A1 (Appendix A) for high water levels under neap and spring tide conditions.

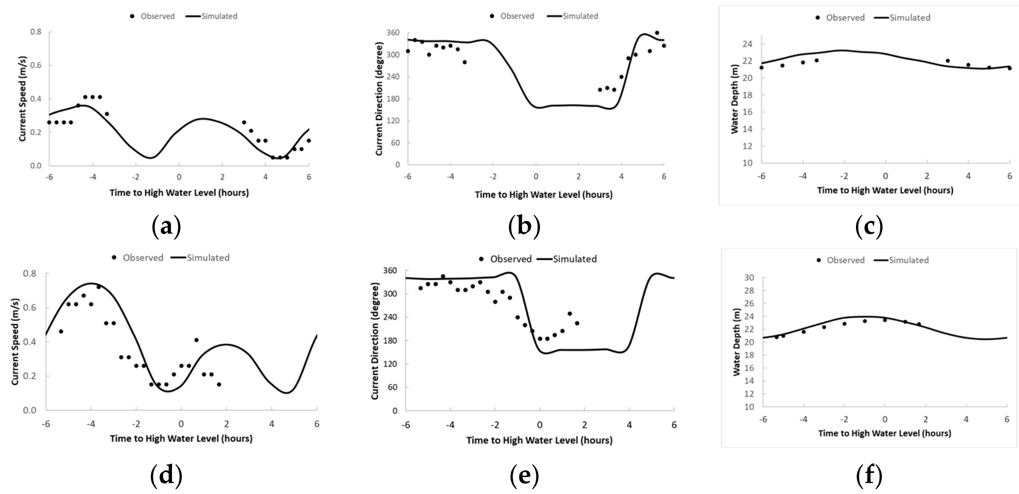


**Figure 2.** Comparison of the current speed and direction at site H7 for neap and spring tides: (**a**) the current speed at neap tide; (**b**) the current direction at neap tide; (**c**) water depth at neap tide; (**d**) the current speed at spring tide; (**e**) the current direction at spring tide; (**f**) water depth at spring tide.

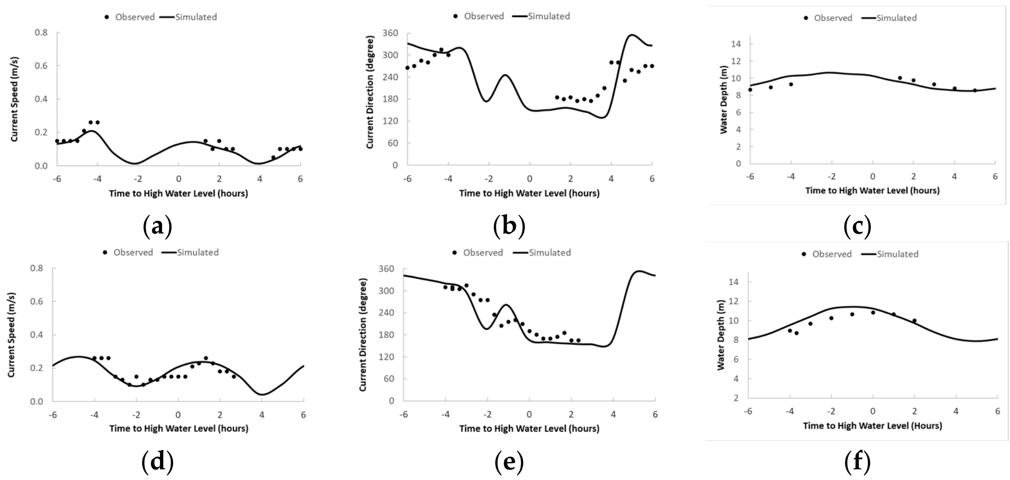


**Figure 3.** Comparison of the current speed and direction at site H8 for neap and spring tides: (**a**) the current speed at neap tide; (**b**) the current direction at neap tide; (**c**) water depth at neap tide; (**d**) the current speed at spring tide; (**e**) the current direction at spring tide; (**f**) water depth at spring tide.

To fully assess the residence and exposure times in Dublin Bay, a matrix of 20 scenarios, reflecting a range of tidal conditions and covering a spectrum of river inflows, was considered. Tracer releases for spring tide at high water levels (SH), spring tide at low water levels (SL), neap tide at high water levels (NH), and neap tide at low water levels (NL) underpinned our investigation of tidal conditions. In combination with these tidal conditions, river inflows (Table 1) to the bay corresponding to the minimum, median, maximum, first quartile, and third quartile flows (flow magnitudes 1 to 5, respectively) were simultaneously examined, producing the tested scenarios in Table 2. Stream flow magnitudes were based on the Irish Environmental Protection Agency's daily averaged hydrometric records from the furthest downstream monitoring stations in the River Tolka (Station No. 09037, Botanic Gardens) and River Dodder (Station No. 09010, Waldron's Bridge) for the period from January 2000 to September 2019 (see Figure 1). The Liffey is regulated by an upstream hydroelectric plant and dam, and only minimal seasonal variations of flows leaving the dam are observed. Flows from the Liffey catchment into the modelled domain were, therefore, based on an averaged flow of 12.4 m$^3$/s from the dam [13] but which was augmented by flow inputs from the Camac River (a tributary river that joins the Liffey between the dam and its outfall to the Irish Sea, which is gauged by the EPA at Station No. 09035, Killeen Road, on the Irish hydrometric network).

**Table 1.** River flow magnitudes used in model simulations.

| | Minimum ($m^3$/s) | Q1 ($m^3$/s) | Median ($m^3$/s) | Q3 ($m^3$/s) | Maximum ($m^3$/s) |
|---|---|---|---|---|---|
| Liffey | 12.48 | 12.63 | 12.79 | 13.02 | 24.30 |
| Tolka | 0.19 | 0.53 | 1.04 | 2.09 | 74.90 |
| Dodder | 0.31 | 0.93 | 1.44 | 2.47 | 54.73 |

**Table 2.** Summary of model simulations (S and N refer to spring and neap tides, H and L refer to high and low water levels, and numbers 1 to 5 define the minimum to maximum river flows).

| Flow | SH | SL | NH | NL |
|---|---|---|---|---|
| Minimum | SH-1 | SL-1 | NH-1 | NL-1 |
| Q1 | SH-2 | SL-2 | NH-2 | NL-2 |
| Median | SH-3 | SL-3 | NH-3 | NL-3 |
| Q3 | SH-4 | SL-4 | NH-4 | NL-4 |
| Maximum | SH-5 | SL-5 | NH-5 | NL-5 |

## 3. Results and Discussion

The method of calculating the residence time and exposure time used in this study was based on integrating the remnant function (Equation (2) from the initial time $(t_0)$ to infinity $(t_{+\infty})$. To address this, previous studies [4,5,12] have suggested integrating the remnant function over a finite time period, albeit this period needs to be sufficiently long for the tracer to leave the domain of interest. Figure 4 shows the change of the remnant function for both the residence and exposure times for a fixed simulation period of 300 days under scenario SH-3. The results show a sharp drop in the remnant function for both the residence and exposure times in the initial period of the simulation but in a significantly different way. The remnant function for the residence time was observed to drop monotonically without oscillations, but for the exposure time, the decrease was characterised by significant oscillations, reflecting the repeated departure and return of the water and tracer to and from the study domain under flooding and ebbing tides. Figure 4 also shows that the cumulative residence time (c. 2 days) and cumulative exposure time (c. 8 days) become constant at their maximum values after c. 30 days and 180 days, respectively, indicating the simulation period of 300 days provides adequate time for a water exchange equilibrium to be reached.

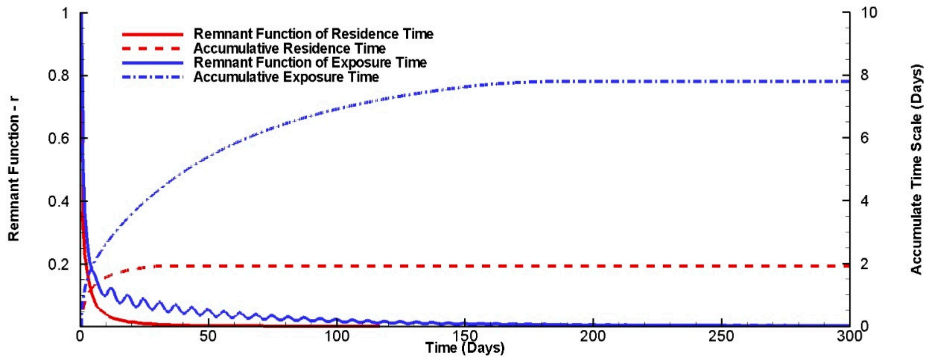


**Figure 4.** Variation of the remnant function for residence and exposure times and accumulative residence and exposure times.

Figure 5 shows the variation of simulated residence and exposure times for different river flow magnitudes, tidal conditions, and tracer release times in Dublin Bay. The data confirm that the exposure times are considerably higher than the residence times for all the scenarios, again reflecting the tidal dynamics where water leaves the study domain on an ebbing tide but re-enters the domain on incoming tides. Of note, however, is the relative difference between the calculated exposure and residence times, which is shown to be considerably higher for spring rather than neap tide conditions, indicating that a smaller proportion of water and tracers exits the bay under neap tide conditions and returns on the incoming tides. Influences of changed river flow conditions on residence and exposure times are also evident in Figure 5, with both time scales generally decreasing with an increasing flow magnitude but being more significant under neap than spring tide conditions. Residence and exposure times were also shown to be influenced by tracer release times, with calculated residence times decreasing in magnitude in the order of NL > NH > SL > SH but with exposure times decreasing in the order of SL >NL > SH > NH. These patterns of decrease contrast sharply with those reported for micro-tidal estuarine settings [5] and hyper-tidal estuaries [12], with differences being attributed to significant variability in the return coefficients determined for the different tidal conditions that were tested in the current study. This variability is shown in Figure 6 and confirms the significant differences in the calculated return coefficients for neap and spring tide conditions in the macro-tidal setting of Dublin Bay with a shallow water depth. The higher return coefficients for spring tides (having a value of 0.6 for the SL scenario but increasing to 0.8 under the SH scenario) indicate that although the capacity for both mixing and tracer transport from the bay is enhanced under spring tide conditions, most of the water and tracers that leave the bay on an ebbing spring tide returns on the next flooding tide. The opposite is the case for neap tides (with return coefficients of 0.1 and 0.3 for the NL and NH scenarios, respectively), where there is a reduced capacity for both the transport of water from the bay and its return on subsequent incoming tides. The significant variations of the return coefficients for neap and spring tides observed here for Dublin Bay differed from other studies of micro- and hyper-tidal estuaries with relatively deep water and where the tidal range-to-water depth ratios were relatively small [4,5,12]. While the high tidal range-to-depth ratio in Dublin Bay contributes to these observed differences, further studies are required to fully understand the effects of the range-to-depth ratio on the returning effects of tides in different tidal settings. The results, however, have potentially significant implications for water quality management in macro-tidal estuarine and coastal settings, in that considering both the tidal condition (range) and the release time of the tracers (or pollutants) is shown to be critical in water quality management strategies where minimising exposure times important for public health protection measures is a central pillar of the strategy. For example, the exposure times in Figure 5 are shown to be lowest for the NH (neap tides coinciding with high water levels) condition for all magnitudes of the river flow inputs, suggesting that this represents the potentially most favourable setting for reducing the impacts and risks of pollution releases into the bay. However, given the differences in the water exchange and pollutant transport processes that will be evident across an area the size of Dublin Bay, further investigation is needed to fully understand the pollutant release times at different locations in the domain.

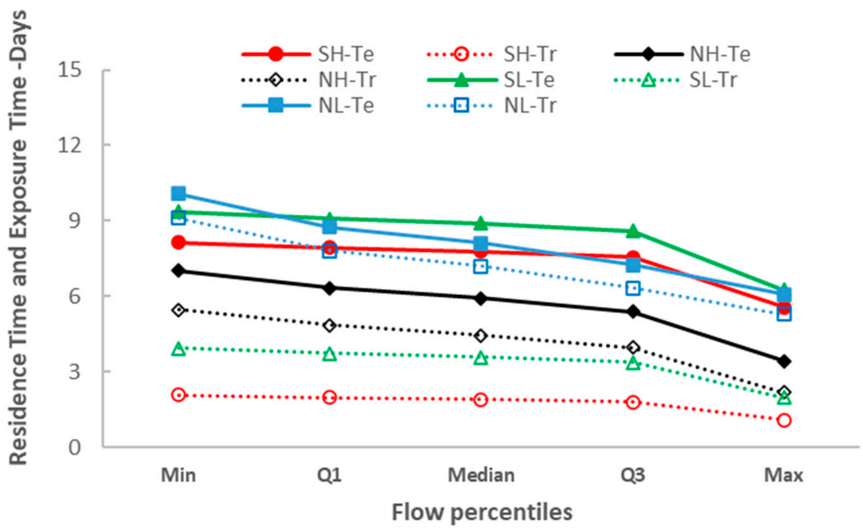


**Figure 5.** Variation of residence time exposure times for different river flow magnitudes, tidal conditions, and tracer release times ($T_r$ and $T_e$ define the calculated residence and exposure times, respectively).

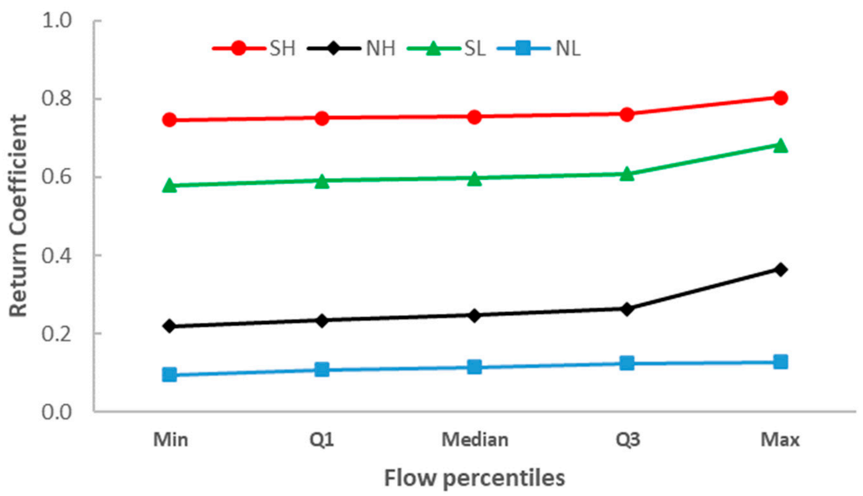


**Figure 6.** Variation of calculated return coefficients for different river flow magnitudes, tidal conditions, and tracer release times.

The spatial distribution of the calculated exposure and residence times in the study domain indicates that there are regions in Dublin Bay where risks to public health from the persistence of pollution are potentially greater than in other areas of the bay, although the average residence time and exposure time of the bay are not very high. Methods of health risk assessment can be found in Nguyen and Huynh [28]. These areas, which are generally concentrated along the southwestern coastline for spring tide conditions, and which include the Merrion and Sandymount Strands, are shown in Figures 7 and 8 (for the minimum, median, and maximum river flows). Areas of the bay where the tendency exists for tracers to persist are shown to extend into the coastal waters in the north-western extent of the study area for neap tides (Figures 9 and 10). These regions of higher transport time scales are typically those where hydrodynamic mixing and the transport processes are more limited and where the potential for dilution of pollutants is diminished. The distributions of the exposure and residence times for the SH condition in Figure 7 confirm that the influence of inputs from the river systems on transport time scales is limited to the Liffey and Tolka estuaries under low flow conditions (Figure 7a,b,d,e) but is shown to extend northwards to the Dollymount Strand at high flows (Figure 7c,f). Importantly, however, transport time scales are shown to reduce as the river flows increase (Figure 5). Considerably less variability in the transport time scales on the southern side of Dublin Bay (incorporating the Sandymount and Merrion Strands) was observed for increasing river flows, with the data highlighting the propensity for pollutants in this region of the bay to persist for considerably longer periods. Similar patterns were observed in other modelled conditions, including the SL (Figure 8), NH (Figure 9), and NL (Figure 10) scenarios. The spatial difference in the TTS suggests that for hydrodynamically active parts of the bay, the river flow conditions and tidal ranges can be used to inform the water quality management strategies in these areas. However, for less dynamic regions of the domain, the flow and tidal conditions are shown only to have a limited impact, suggesting that water quality management strategies might best be focused on controlling local pollutant loadings or altering the discharge point where these loadings enter the bay.

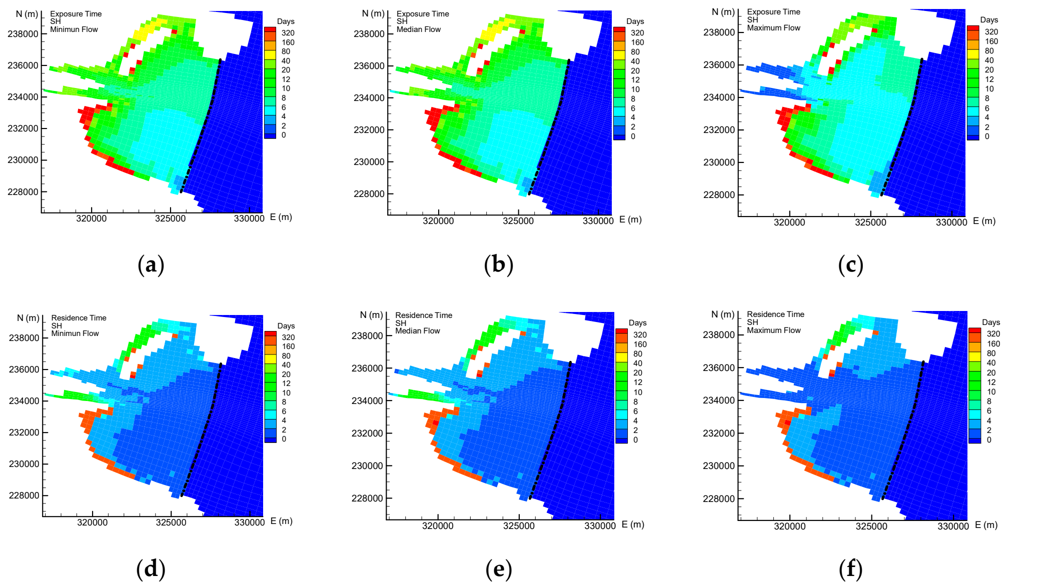


**Figure 7.** Distribution of exposure (**a**–**c**) and residence times (**d**–**f**) across the study domain for the SH scenario for the minimum, median, and maximum river flows.

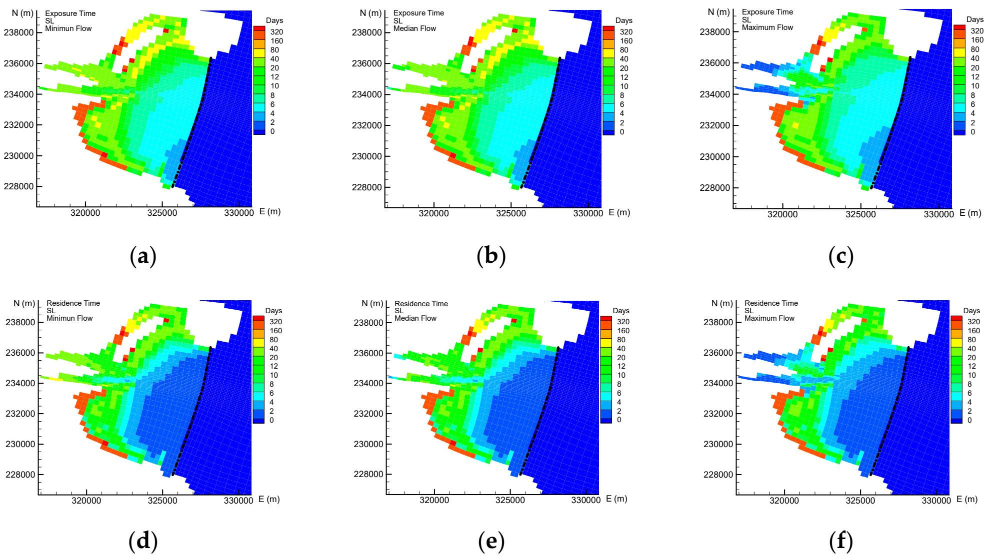


**Figure 8.** Distribution of exposure (**a**–**c**) and residence times (**d**–**f**) across the study domain for the SL scenario for the minimum, median, and maximum river flows.

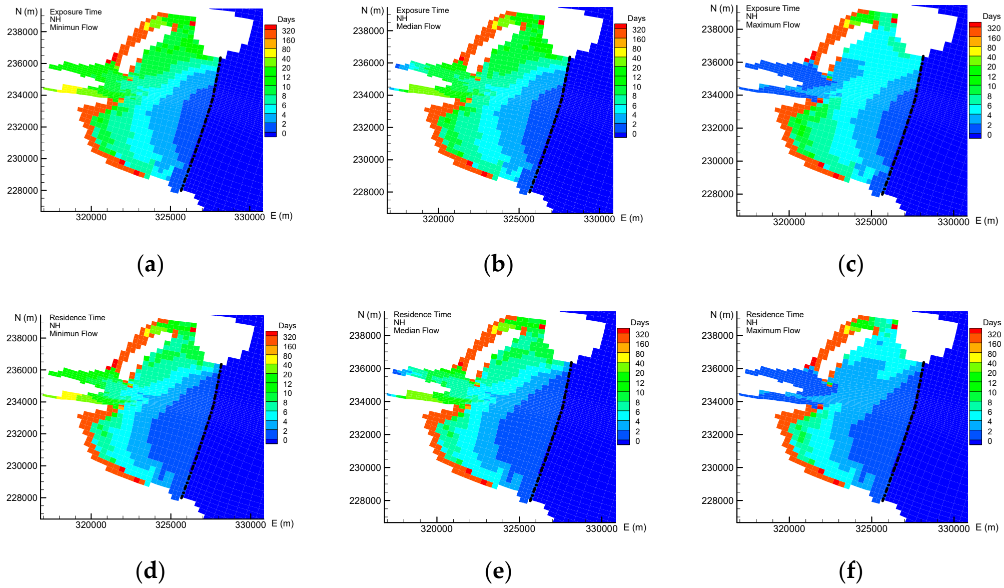


**Figure 9.** Distribution of exposure (**a**–**c**) and residence times (**d**–**f**) across the study domain for the NH scenario for the minimum, median, and maximum river flows.

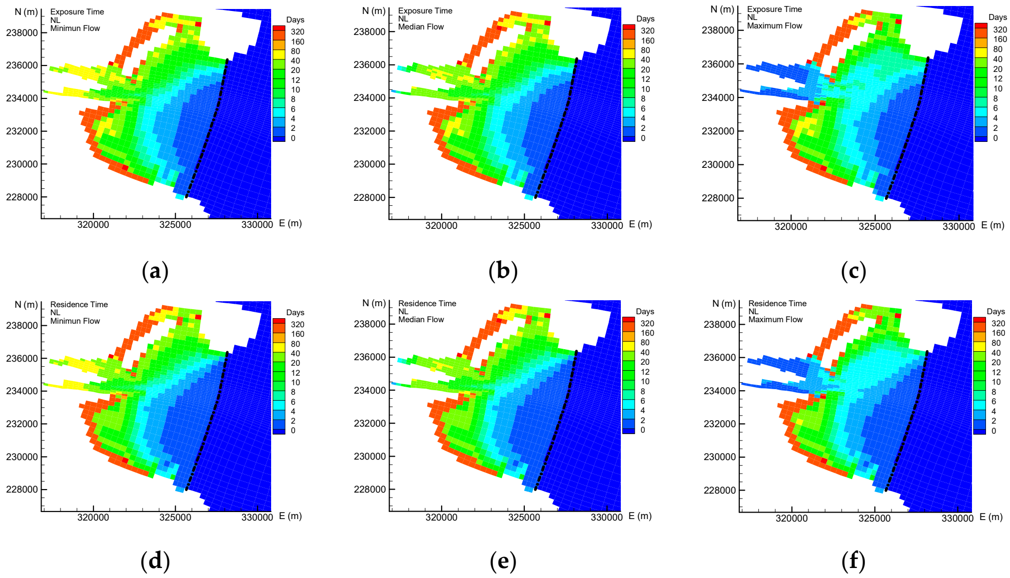


**Figure 10.** Distribution of exposure (**a**–**c**) and residence times (**d**–**f**) across the study domain for the NL scenario for the minimum, median, and maximum river flows.

These findings, therefore, contribute to a wider understanding of pollution patterns and persistence in macro-tidal settings and have implications for the management of coastal waters in the context of compliance with legislative and regulatory frameworks. For example, compliance with the recreational bathing water in Dublin Bay is governed by prescribed standards in the EU Bathing Water Directive [15]. In the context of this directive, the results indicate that concentrations of faecal indicator bacteria (FIB) in the nearshore waters on the northern side of the bay (the Dollymount Strand) will be typically lower than on the southern side of the bay (the Merrion and Sandymount Strands). Furthermore, the exposure times in Figure 7a–c are dramatically higher compared to the residence times in Figure 7d–f. Comparing Figure 7 to Figure 8 also shows that residence and exposure times under the SH scenario are lower, albeit less so for the exposure times than those for the SL condition. Under the SL scenario (Figure 8), the returning effects of the tide are shown primarily to affect only the outer extents of the bay, but under the SH condition

(Figure 7), the return effects are observed in the whole bay. The river flow effects under neap tide conditions (Figures 9 and 10) were much more obvious and significant than those under spring tide conditions (Figures 7 and 8). Somewhat surprisingly, the exposure time under the NH (Figure 9a–c) condition is smaller than that under the spring tide condition (Figures 7a–c and 8a–c) almost everywhere in the bay, except the high retention time region along the coastline of the Sandymount and Merrion Strands and the northeast coastline. The exceptions were attributed to situations where water could 'flood' and dilute the area near the coast under spring tide conditions. However, the extremely mild slope of the intertidal zone of the Sandymount and Merrion Strands led to a less dynamic exchange process between the shallow region and the deeper area. Under the NL condition (Figure 10), the difference between the exposure time and residence time was much less obvious than in all the other scenarios. The data in Figure 10 for this test condition is therefore consistent with the low return coefficients in Figure 6.

## 4. Conclusions

This study investigated transport time scales (TTS) in a macro-tidal setting with a relatively shallow depth, focusing specifically on the variability of exposure and residence times in Dublin Bay on the east coast of Ireland. The study utilised the EFDC integrated hydrodynamic and solute transport model, which was refined for application to Dublin Bay, calibrated and validated against measured data, and subsequently used to determine exposure and residence times for the different tracer release conditions, tidal ranges (neap and spring tides), and flow conditions of rivers that discharge into the Dublin Bay. The study advances the current understanding of water exchange in macro-tidal settings with a shallow water depth and provides findings for water quality control and management in such water bodies. Specific to Dublin Bay, these findings provide an understanding of pollutant persistence risks in different regions of the bay, data that could be extended to bacterial, pathogen, and virus persistence in assessments of risks to public health and wellbeing. The broad range of conditions that were simulated ensured that the full extent of the water exchange processes that prevail in Dublin Bay was reflected in the study. Of significance are the differences in the water exchanges and pollution persistence that were observed for different locations of the bay, indicating that a 'one size fits all' water management strategy is not applicable. Rather, the results suggest that good management of water quality should be underpinned by a more granular approach where mitigation measures specific to a particular location are implemented. The main conclusions are as follows:

(1) The average residence and exposure times in Dublin Bay were shown to be influenced by the magnitude of river flow inputs, particularly under high flow conditions. The reduced averaged residence and exposure times associated with higher river discharges highlight the potential for TTS in the bay to exhibit a seasonal signal.

(2) The influence of river flows on the residence and exposure times is shown to be typically limited to the nearshore areas of the Liffey and Tolka estuaries. Under high flow scenarios, impacts were shown to extend into more northern areas of the bay, affecting the Dollymount Strand, but the southern area of the studied domain, which included the Merrion and Sandymount Strands, remained unaffected.

(3) Dublin Bay is a macro-tide bay, with its highest tidal ranges exceeding 4 m, and the contribution of tidal effects to water exchange processes is significant. The large tidal range also explains the increased significance of rivers only under high flow conditions on calculated TTS. While previous studies in micro-tidal settings point towards reduced exposure and residence times for tracers released at high water levels, regardless of the tidal range, the findings in this study differ. The tidal range is shown to influence both the TTS studied, with residence time values following the order of NL > NH > SL > SH but exposure time values being ordered SL >NL > SH > NH, in which the changed order is attributed to significant differences in the return coefficients under different tidal ranges.

(4) The return coefficients across the tidal conditions and tracer releasing times that were considered were shown to vary between 0.1 and 0.8 but were less sensitive to changes in the magnitude of the river discharges to the bay. The return coefficients strongly influenced the calculated exposure times in Dublin Bay, with the low return coefficient contributing to the lowest exposure time for the NH scenario. Conversely, large return coefficients were associated with the more extended exposure times (longer than those for the NH scenario) calculated for the SH scenario.

(5) The time of the tracers' release significantly affects the residence and exposure times. Under the same tidal range, releasing tracers at higher water levels results in lower residence and exposure times. However, differences in the return coefficients under high tidal range conditions mean that lower exposure times cannot by themselves be guaranteed (higher exposure times were, for example, observed for the SH rather than the NH scenario), and this highlights the importance of carefully considering the release times for the pollution management of coastal and estuarine waters in a macro-tidal setting with a relatively shallow depth.

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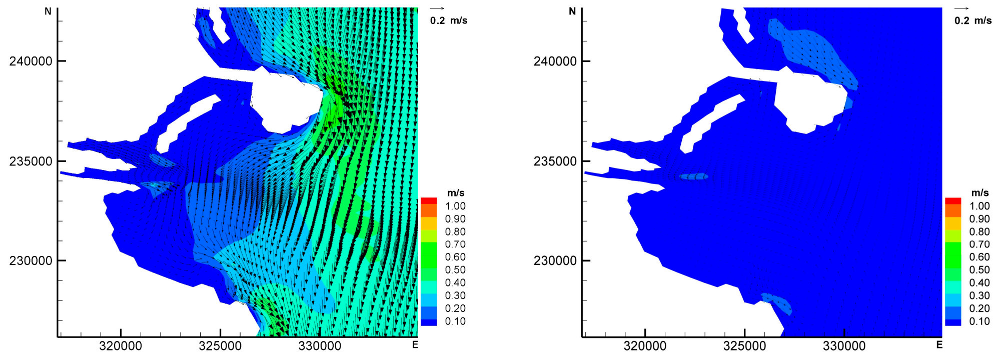
## Appendix A

**Table A1.** Definitions of transport time scales (TTS) reported in the literature.

| TTS | Definition | References |
|---|---|---|
| Flushing time | Time required to replace the existing freshwater in the whole estuary or segment of it at a rate equal to the river discharge. | [3,6] |
| Turnover time | The time required for the freshwater discharge to completely replace the fresh water in the estuarine volume. | [7,8] |
| Residence time | The residence time of a water or a tracer parcel is the time taken by this parcel to leave the controlled domain for the first time from its starting point. | [4,5,9] |
| Exposure time | The exposure time measures the total amount of time spent by a water parcel in the controlled domain, considering all subsequent re-entries to the domain. | [4,5] |
| Influence time | The influence time is the time needed for external particles to reach the observation point. | [9] |
| Age | The age is the time a water parcel has spent since entering the estuary through one of the boundaries. | [3] |




(**a**) (**b**)

**Figure A1.** Simulated current speeds at high water levels for spring (**a**) and neap tide (**b**) conditions.

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