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Enhancing transport and decay models for faecal indicator organisms in nearshore coastal waters☆

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A R T I C L E I N F O

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A B S T R A C T

Pathogens in nearshore coastal waters have far-reaching public health and economic implications. Faecal indicator organisms (FIOs) are commonly monitored and modelled to indicate pathogen levels in waterbodies. FIO decay modelling is an integral part of numerical hydro-epidemiological models to simulate the die-off of FIOs in the water bodies. This paper identifies the limitations of one of the comprehensive and widely used FIO decay models, developed by Stapleton et al. and enhances the model by remedying the limitations. The identified limitations are: (i) the decay rates for dark or highly irradiated environments are not accurately presented, and (ii) the effect of salinity is not included. Two enhanced models have been developed, namely (i) the ClipStap model, devised by imposing a minimum decay rate to the Stapleton model, and (ii) the RevStap model, devised by extrapolating the decay rate-irradiation slope at a reference irradiation (260 W/m^2) down to lower irradiation regions. The enhanced models reproduced the literature-reported dark decay rates better and significantly improved the agreement between the modelled and measured decay rate. The enhanced decay models were tested by including them in a hydro-epidemiological model for a data-rich case study, namely Swansea Bay, UK. Results show that the RevStap model improved FIO prediction in some cases. Besides the enhanced models, this research attributes the diurnal variations of FIO to the combined action of riverine FIO inflows, tide action, and FIO decay. These insights on the effect of irradiation and diurnal FIO variations are critical for assessing the impact of water quality on human activities and nearshore ecology.

1. Introduction

Contamination of nearshore coastal waters has far-reaching consequences, such as public health, reduced tourism, contaminated food from aquaculture and the associated economic loss (DeFlorio-Barker et al., 2018; Bussi et al., 2017; Given et al., 2006; Weiskerger and Phanikumar, 2020). The US marine economy annually provides 2.4 million jobs and contributes $397 billion to the country's Gross Domestic Product (NOAA, 2023). Domestic overnight trips to coastal areas in Great Britain contributed £4.6 billion in 2022 (Visit England, 2023). Faecal Indicator Organisms (FIOs), a class of contaminants, are highly correlated with illnesses such as gastro-intestinal infections and eye infections (Pruss, 1998; Pandey et al., 2014). The revised EU Bathing Water Directive (rBWD; European Commission, 2006) stipulated the maximum allowable FIO concentrations in European bathing waters. FIOs in nearshore coastal waters come from various sources including river and stream flows, sewage treatment work (STW) discharges, combined sewage overflows (CSOs) and diffuse sources, such as animal waste (e.g. Crowther et al., 2001; Ahn et al., 2005; Wyer et al., 2010; Passerat et al., 2011; Zhang et al., 2012; King et al., 2021). Furthermore, climate change, population growth and urbanisation are expected to increase stormwater runoffs and CSOs (Semadeni-Davies et al., 2008; Jalliffier-Verne et al., 2017; Perry et al., 2024), which in turn increases the FIO concentrations in coastal waters. Understanding the transport and fate of FIOs is indispensable for planning wastewater treatment efforts and other water quality management strategies, as well as early warning systems that inform the public and other key stakeholders, e.g., the mariculture industry, of impending water quality to protect public health.

FIO modelling is indispensable for studying nearshore coastal waters and water quality management. FIO models have two board classes: (i) data-driven models and (ii) hydro-epidemiological models. Data-driven models (e.g. Crowther et al., 2001; Zhang et al., 2012; Lam and Ahmadian, 2023) are computationally efficient and can provide timely water

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quality predictions for early warning systems. Mechanistic hydro-epidemiological models (e.g. Harris et al., 2004; Lee and Qu, 2004; Gao et al., 2013a; Schippmann et al., 2013) are computationally more expensive but provide more insights into source apportionment as well as the transport and fate of FIOs. Hydro-epidemiological models have been widely applied to predict FIO concentrations at sensitive receivers, review water quality monitoring methods, establish source-receptor connectivity and evaluate the impacts of water management strategies on water quality (Lin et al., 2008; Ahmadian et al., 2013; Schippmann et al., 2013; Abu-Bakar et al., 2017). An important aspect in hydro-epidemiological models is modelling FIO decay. FIO decay rate depends on many factors, such as temperature, solar irradiation, and suspended solid concentration (Byappanahalli et al., 2012). Several FIO decay models (Bowie et al., 1985; Crane and Moore, 1986; Weiskerger and Phanikumar, 2020), with various levels of complexity, have been developed and applied in different waterbodies, such as coastal areas, estuaries, and rivers. Yet, there is no consensus regarding the best model for any given scenario, and most models have limitations when used in real-world studies. This study aims to enhance FIO decay modelling to improve FIO simulation. For this purpose, the FIO decay model developed by Stapleton et al. (2007a), which has been demonstrated a good performance in simulating field sampled FIO concentrations (Willis et al., 2010; Gao et al., 2013b; King et al., 2021), but also showed some limitations, was selected. The new revised decay model demonstrates clear improvements to the existing models. This research also proposes an explanation for the observed diurnal variations of FIOs in bathing waters (Wyer et al., 2018). This research provides an understanding of FIO transport and decay and improves FIO modelling, which are necessary for assessing the impacts of coastal development on public health and the effectiveness of water quality improvement strategies. The remainder of this article is organised as follows: Section 2 shows the limitations of the Stapleton model and presents the enhanced Stapleton models. Then, enhanced Stapleton models are incorporated into hydro-epidemiological models for Swansea Bay, the UK, in section 3. Section 4 presents the hydro-epidemiological model results with different enhanced Stapleton models. Finally, Sections 5 and 6 present the discussion and conclusions, respectively.

2. Enhancement of the FIO model

This section highlights the limitations of the Stapleton model and the enhancement made to the model. Section 2.1 gives a brief introduction to FIO decay models. The limitations of the Stapleton model concerning irradiation and salinity effects and the corresponding enhancements are presented in Sections 2.2 and 2.3, respectively.

a. FIO decay models

While FIO decay is a complex process depending on a myriad of environmental factors (Byappanahalli et al., 2012), it has been usually modelled as a first-order decay process (Bowie et al., 1985; Crane and Moore, 1986; Weiskerger and Phanikumar, 2020):

$$C(t) = C_0 \exp(-kt) \tag{1}$$

where $C(t)$ and C_0 are the concentration at time t and the concentration at $t = 0$ respectively; k is a decay rate. The decay rate may be expressed in T_{90}, the time required for FIO concentration to reduce by 90 %:

$$T_{90} = \frac{2.303}{k} \tag{2}$$

The FIO decay rate k is a function of different environmental variables such as irradiation, salinity, and turbidity. Previous research on FIO decay models mainly concerned the function of k (Weiskerger and Phanikumar, 2020). Stapleton et al. (2007a, b) developed a decay model from a field water sample study in Severn Estuary and Bristol Channel. King et al. (2021) compared the Stapleton et al. (2007a, b) model and the widely-used Mancini (1978) model for Swansea Bay, UK and concluded that the Stapleton model better predicted FIO concentrations. Nevertheless, the Stapleton model has limitations, and this paper aims to enhance the original Stapleton model. This section introduces the Stapleton model, its limitations, and the enhancements that have been attempted to improve these limitations. The enhanced decay models were incorporated in a hydro-epidemiological model and applied to a case study, Swansea Bay, UK, in Sections 2.4 and 2.5 to assess its performance in a practical case. The model results are presented and discussed in Sections 3 and 4. Section 5 concludes this research work.

b. The Stapleton model and its limitations

The Stapleton model, which incorporates the effect of irradiation and turbidity on FIO decay, is as shown in Equation [3–7]:

$$T_{90} = T_{90,2} + \left(T_{90,1} - T_{90,*1}\right) \tag{3}$$

$$log_{10} T_{90,2} = 0.0047 t_b + 0.677 \tag{4}$$

$$T_{90,1} = \frac{\ln 10}{60 K_B I} \tag{5}$$

$$T_{90,*1} = \frac{\ln 10}{60 K_B I_{exp}} \tag{6}$$

$$t_b = 139.479 \times log_{10}(S_s) - 244.736 \tag{7}$$

where T_{90} is the time required for FIO concentration to reduce by 90 % (in hour); t_b is turbidity (NTU); I is irradiation (W/m^2); $I_{exp} = 260\ W/m^2$ is a reference irradiation adopted from the experiment in Stapleton et al. (2007b); S_s is total suspended solid (TSS) concentration (mg/L); K_B is an FIO species dependent constant. Stapleton et al. (2007a) suggested that $K_B = 1.1 \times 10^{-5}$ for *Enterococci*. While Stapleton et al. (2007a) did not suggest a K_B value for *E. coli*, such a K_B value can be adopted from Alkan et al. (1995) to be $K_B = 1.3 \times 10^{-5}$. The decay model has two limitations: (1) the model does not apply to very low (nearly zero) or very high irradiations; (2) the model does not consider the effect of salinity. The first limitation arises from Equation [3–7] giving $T_{90} \to \infty$ as $I \to 0$ (i.e. bacteria do not decay in the dark). This is not reasonable since FIOs are known to decay, although at a slower rate, in the dark (Stapleton et al., 2007b). Most of the literature (Johnson et al., 1997; Jin et al., 2003; Hipsey et al., 2008) reported dark decay rates between 0.02 and 0.092 h^{-1} in freshwater and seawater. In addition, Stapleton et al. (2007b) did not conduct any experiment with irradiations greater than 260 W/m^2 and the mathematical form of the model gives a zero T_{90} (i.e. $k = \frac{2.303}{T_{90}} \to \infty$) at $I = 570.5\ W/m^2$ as shown in Fig. 1. $T_{90} = 0$ implies an immediate FIO die-off, which is not reasonable. The second limitation arises because Equations [3–7] do not include salinity. King et al. (2021) circumvented the first limitation by imposing artificial upper (260 W/m^2) and lower (15 W/m^2) limits of irradiation to the model, which could affect the accuracy of the predictions in some practical circumstances. This research enhances the model by remedying these limitations.

c. Enhanced Stapleton models

Two models were developed and tested to resolve the limitations above. The models are referred to as the Clipped Stapleton (ClipStap) and the Revised Stapleton (RevStap) models. Both models use simple adjustments to represent the measured data more accurately. Simple adjustments are favoured compared to complex models to broaden the applicability and limit reliance on further local data for readjustment, as well as the limited data availability in Stapleton et al. (2007b). The first enhanced model, namely ClipStap, was developed by incorporating a

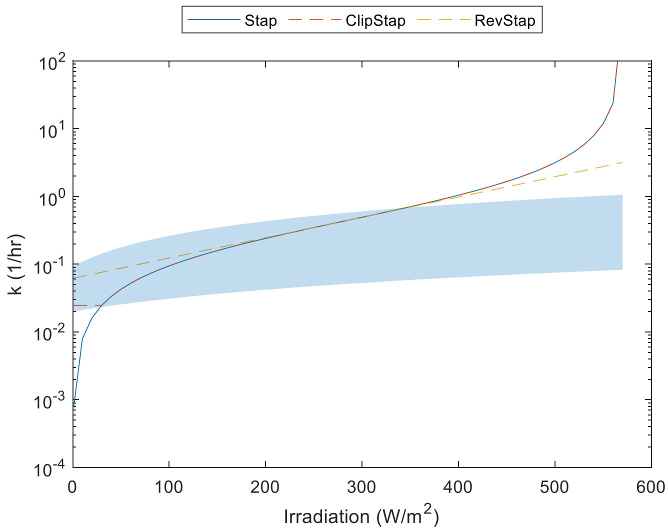


Fig. 1. FIO decay rates from the three models with $C_{sal} = 0$ and the range of decay rates obtained from the literature (the blue shade). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

minimum decay rate that corresponds to the dark decay rate onto the Stapleton model based on the approach implemented by Hipsey et al. (2008). It was found that the required minimum decay rate corresponds to the original Stapleton modelled decay rate at irradiation of 30 W/m^2 at TSS concentration (S_s) of 84.82 mg/L (value adopted from King et al., 2021). The second enhanced model, the Revised Stapleton (RevStap) model, was also developed by providing remedies to limitations due to solar intensity and exclusion of salinity.

The RevStap model was developed by first extrapolating the slope $\frac{d[\ln k]}{dI}$ at $I = 260\ W/m^2$ to $I = 0\ W/m^2$ and secondly adding the salinity term from Mancini (1978), Huang et al. (2015) and Weiskerger and Phanikumar (2020). The resulting model is:

$$k = k_{DI} + k_s = \frac{\ln(10)}{T_{90,2^*}} + 0.0008333C_{sal} \quad [8]$$

Where $k_{DI} = \frac{\ln(10)}{T_{90,2^*}}$ and $k_S = 0.0008333C_{sal}$ are the irradiation-induced and salinity-induced FIO decays, respectively; C_{sal} is salinity. $T_{90,2^*}$ is evaluated as follows:

$$log_{10}(T_{90,2^*}) = 0.0047t_b + 0.677 + 0.003(I^* - I) \quad [9]$$

where $I^* = 260$ W/m^2. Fig. 1 compares the original Stapleton, Clipped, and Revised Stapleton models. The Revised Stapleton model agrees with the Stapleton model when $I > 150\ W/m^2$, yet giving a nonzero k when $I = 0$. Figs. SI–1 and Table 1 show that the ClipStap and RevStap models significantly improved the agreement between the modelled and measured decay rates, particularly for the decay in low solar radiation conditions ($\boldsymbol{k_{dark}}$), as reflected by their smaller root mean square error (RMSE) values.

The FIO decay rates estimated from the Stap, ClipStap and RevStap models were compared to the range of decay rates reported in the literature, as shown in Fig. 1. The FIO decay rates for the three models were estimated without the inclusion of salinity, namely $C_{sal} = 0$. To estimate the literature-based decay rate, the reported dark decay rate k_d and linear coefficient for the effect of irradiation α were collected from the literature, as shown in Tables SI–1 and SI-2. The maximum and minimum values of reported k_d and α were selected and are shown in Tables SI–3. The values in Tables SI–3 were applied to estimate the range of literature-based irradiation-induced decay rate k_{DI} with the expression in Weiskerger and Phanikumar (2020) as below:

$$k_{DI} = k_d + \alpha I \quad [10]$$

Fig. 1 shows that the ClipStap and RevStap models reproduced the reported dark decay rates well, suggesting the enhancements to the Stapleton model are reasonable. In particular, the RevStap model also provides improved modelling for high irradiation scenarios, preventing the decay rate from going to infinity at $I = 570.5\ W/m^2$.

Table 1
Root Mean Square Error (RMSE) between the modelled decay rates and the decay rate measured by Stapleton et al. (2007b) at $I = 1\ W/m^2$ (k_{dark}) and $I = 260\ W/m^2$ (k_{irr}).

Decay model	$\boldsymbol{k_{dark}}$ (1/hr)	$\boldsymbol{k_{irr}}$ (1/hr)
Original Stapleton	0.1125	0.1180
ClipStap	0.0936	0.1180
RevStap	0.0591	0.1173

3. Hydro-epidemiological modelling with the enhanced FIO models

It is important to validate the enhanced FIO decay models in simulating real-world conditions where the salinity concentration and solar radiation are constantly changing, despite them being reasonably constant in the lab where the equations are developed. In this section, the enhanced FIO decay models were validated by incorporating them in hydro-epidemiological models for a data-rich site, namely Swansea Bay, UK. The site configuration is presented in Section 3.1, and the hydro-epidemiological setup is given in Section 3.2.

a. Test site: Swansea Bay, UK

Swansea Bay is located on the Bristol Channel on the southwest coast of the UK, as shown in Fig. 2a. The Bay is subject to primary FIO inputs from various sources (Fig. 2b), including three rivers, namely River Tawe, Neath and Afan, three Sewage Treatment Work (STW) discharges, and also smaller FIO inputs from the drains and streams located along the beaches (King et al., 2021). Bristol Channel and Severn Estuary have a maximum tidal range of more than 14 m, the second highest in the world (British Crown and OceanWise Ltd, 2015). The freshwater river and stream flows are insufficient to generate considerable density stratification (Uncles, 1981; Evans et al., 1990; Ahmadian et al., 2013), but horizontal salinity gradients are present (Collins and Banner, 1980). A semi-submerged barrage is located at the River Tawe outlet, which only overtops at tide levels higher than the barrage weirs (3.05 m above Ordinance Datum). However, the River Neath and Afan are tidal up to about 10 km and 1 km upstream from the coast, respectively.

Swansea Bay was chosen for this study because various sources of FIO affect the water quality in the bay and the availability of data and two designated bathing water sites, namely Swansea Beach and Aberafan Beach, along the Bay (NRW, 2023a). The designated beaches are popular tourist attractions and attract many visitors, particularly during the bathing season (City and County of Swansea, 2023; Welsh Government, 2023). There has also been a proposal to build a tidal lagoon in the bay (Hendry, 2016; Swansea Council, 2021), which could significantly change the flow patterns (Což et al., 2019). As part of the Smart Coasts – Sustainable Communities (SCSC) project (Wyer et al., 2013), the FIO concentration along the Swansea transect was sampled at 30-min intervals during the bathing season (May to September) of 2011. The FIO concentrations along the Swansea and Aberfan transects were sampled in autumn (November) 2012. The rBWD requires FIO samples to be taken at a minimum depth of 0.5 m (Bedri et al., 2016) at the Designated Sampling Point (DSP) for each bathing water site. However, the high tidal range and gentle beach slope in the Bay give large tidal areas, exposing a distance up to 1500 m from shore during spring tides. This prevents readings being taken at each bathing water site at one fixed location. Therefore, the sampling point at each site moved with the tidal

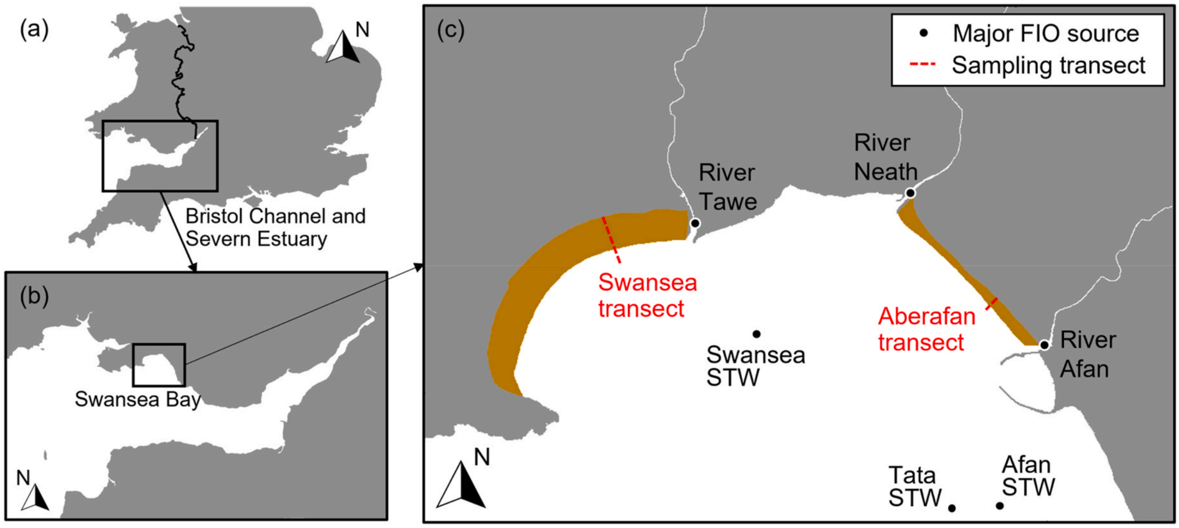


Fig. 2. (a) Location of Bristol Channel and Severn Estuary in UK; (b) location of Swansea Bay in Bristol Channel and Severn Estuary; (c) primary FIO sources and sampling transects in Swansea Bay.

water level along a transect so that the water depth at the sampling point was sufficiently large. Fig. 2c shows two transects for Swansea and Aberafan Beaches. The sampling points along the Swansea transect in the SCSC sampling campaign are shown in Figs. SI–2. The samples were collected in sterile 1 L containers (Aurora Scientific) and stored in a refrigerator before analysis. The samples were then analysed for intestinal *Enterococci* and *E. coli* with standard membrane filtration techniques. The bay is also subjected to FIO inputs from rivers and STWs. As part of the SCSC project, 28 primary and 19 secondary surface water and sewage discharges were recorded at 15-min intervals during the 2011 bathing season. Within the same project, all primary sources were estimated from October to November 2012 at 15-min intervals with the principle in Kay et al. (2008). Estimations were not made for combined sewer overflow (CSO) spills. This set of measured and estimated data has been applied in the hydro-epidemiological modelling of King (2019). In this research, hydro-epidemiological modelling of the Bay was conducted during autumn (November) 2012 and summer (July–August) 2011 to test the FIO decay models under different irradiation conditions, since the autumn was expected to have a lower irradiation than the summer.

b. Modelling setup

TELEMAC-3D (Leroy, 2019) hydro-epidemiological models were developed for Swansea Bay, Severn Estuary and Bristol Channel. TELEMAC-3D can model the effects of density between fresh river water and saline sea water, which were shown to be crucial for accurate representation of the fate and transport of pollutants (Bedri et al., 2013; Gaeta et al., 2020; Lam and Ahmadian, 2024). This paper's numerical domain and meshing are the same as in Lam and Ahmadian (2024) and are shown in Figs. SI–3 for completeness. The bottom friction was modelled with Manning's formula (Chow, 1959) with a constant Manning coefficient $n = 0.025$ throughout the model domain following King et al. (2021).

Fig. 3 shows the *E. coli* source locations, including River Tawe, Neath and Afan, the streams along the beaches, and three offshore sewage treatment discharge points. The flow rates and *E. coli* concentrations in River Tawe, Neath and Afan were modelled as boundary conditions, and other streams and discharges were modelled as source points. The measured and estimated *E. coli* concentrations in the SCSC project mentioned above were imposed at these boundaries and source points. The initial salinity in the numerical domain and the salinity at the open sea boundary were set to be 34 kg/m^3 (Collins and Banner, 1980). A salinity of 0 kg/m^3 (fresh water) was imposed on the river boundaries and the point sources. Although the wind effect is important in some circumstances (e.g. Schernewski et al., 2014; Huang et al., 2022), it was found insignificant in this area (Lam and Ahmadian, 2024) and therefore was not included in this study. Water temperature was set at 15 °C to match values measured in previous studies (Aberystwyth University and University College Dublin, 2018; White et al., 2014). The Smagorinski (1963) model was applied in this study as it has previously shown a good performance in the region (Guo et al., 2020; King et al., 2021) and was recommended for large scale marine areas with large-scale eddies (Gourgue et al., 2013; Bedri et al., 2015). The original Stapleton (Stap), ClipStap and RevStap decay models were tested. The upper limit for irradiation $I_{ul} = 570\ W/m^2$ was imposed to the Stap and ClipStap models to avoid k going to infinity (i.e. $T_{90} = 0$) when $I \geq 570.5\ W/m^2$. The lower limit for irradiation $I_{ll} = 1\ W/m^2$ was imposed to the Stap model to avoid k going to zero when $I = 0$.

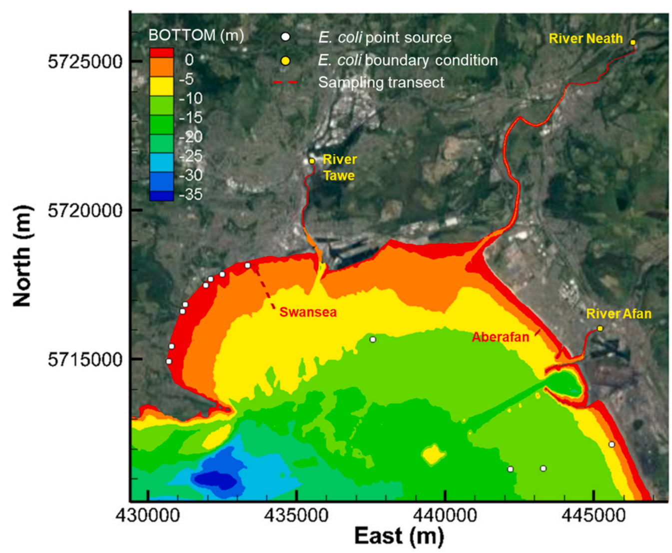


Fig. 3. *E. coli* sources and sampling transects in the numerical domain.

The freshwater inflows diluted the salinity in the Bay once the simulations were started. A sufficiently long precursor run period, with *E. coli* concentrations from freshwater inflows set to zero, is required to ensure that the salinity concentrations reach a stationary state where long-term salinity variation is negligible. The needed precursor run period was found to be 38 days (with time step $\Delta t = 1.00$ s) for the average change in salinity to reach -0.0086 kg/m^3 per tidal cycle, which was considered small and not affecting the overall results. Once the stationary state was reached, *E. coli* inputs from freshwater inflows were modelled. Numerical convergence was tested at the model runs with $\Delta t = 1.00$s, $\Delta t = 0.50$s and $\Delta t = 0.25$s and showed that a time step of $\Delta t = 0.50$s was sufficiently small for accurate modelling of *E. coli* concentrations. The simulated *E. coli* concentrations were obtained at moving sampling points along the Swansea and Aberafan transects with water depth approximately 1 m as in Lam and Ahmadian (2024) because the difference between *E. coli* concentrations sampled at water depths 0.5 m, 1.0 m, 1.5 m, and 2.0 m were found negligible.

4. Results

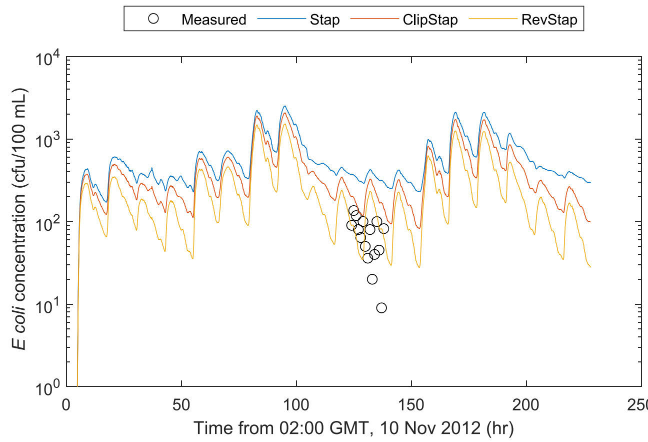
a. Model validation

The numerical setup in this research was similar to the setups in in Lam and Ahmadian (2024), King et al. (2021) and Guo et al. (2021), and was validated in this paper against the water level measured at the Mumbles tidal station (BODC, 2023) and measured velocity using ADCP during SmartCoast project (Ahmadian et al., 2013) as shown in Figs. SI–4. The modelled water levels at Mumbles (location shown in Figs. SI–3) gave root mean square error (RMSE) = 0.3556 m and R^2 = 0.9789 for Simulation 2011, and RMSE = 0.3715 m, R^2 = 0.9827 for Simulation 2012. The depth-averaged flow velocity gave RMSE = 0.0706–0.2069 m/s and R^2 = 0.7064–0.9116 at point L1-L5 for Simulation 2012, with exceptions of y-velocity at points L1 and L5 because of their small values (90 % of the data has a magnitude less than 0.09 m/s). The simulated salinity concentrations were also shown to be relatively constant across the depth in the deep part of the Bay in Figs. SI–5, which is consistent with expected well-mixed conditions and previous research (Uncles, 1981; Evans et al., 1990; Ahmadian et al., 2013).

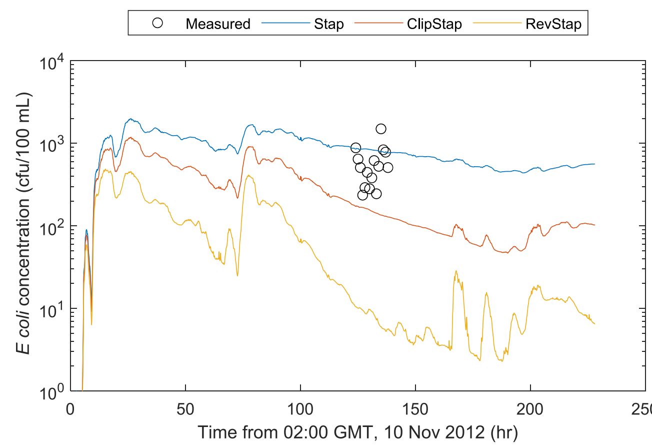
b. Predictions by the FIO decay models

Fig. 4a shows that the RevStap model improved *E. coli* prediction accuracy at the Aberafan transect in Simulation 2012. Figs. 4b and 5a show that the Stap and ClipStap models better predicted the Swansea Bay transects in Simulations 2011; 2012. Nevertheless, the advantage of the RevStep model for high irradiations is demonstrated in Fig. 5. Fig. 5a shows rapid drops in *E. coli* concentrations to below 0.01 cfu/100 mL for the Stap and ClipStap models at 273.1, 319.2, 344.2, 367.3, 394.2 and 441.3 h after start of the simulation, i.e. 03:00 GMT, Jul 12, 2011. These drops correspond to times of high irradiation (>500 W/m^2) in Fig. 5b. These rapid drops were not observed for the RevStap model. An explanation for the drops is that the decay rates for the Stap and ClipStap models increase rapidly when irradiation reaches 500 W/m^2 as shown in Fig. 1, causing rapid *E. coli* decay. The explanation is confirmed by the absence of such drops for the RevStap model in Fig. 5a. Such rapid drops in *E. coli* concentrations were not observed in Simulation 2012 because of the lower irradiation in Nov 2012 compared to the summertime of 2011. The maximum irradiation in the November 2012 simulation was 340.1 W/m^2, unable to trigger the rapid drops in *E. coli* concentrations for the three models.

The diurnal variation of *E. coli* concentration was observed at Aberafan Beach with the RevStap model, as shown in Fig. 4a. In contrast, such variation was not observed as clearly at Swansea Bay in Fig. 4b with any FIO decay models. The fact that the diurnal effect was more identifiable at one site than the other while receiving the same irradiation hinted that the diurnal variations are not solely caused by the irradiation-induced change in decay rates during the day. The prominent modelled diurnal variation at Aberafan Beach could be explained by a combination of significant impacts of *E. coli*, transported through the FIO source (River Afan), tidal action and FIO decay rates. Fig. 6 shows the *E. coli* concentration near the Aberafan sampling transect. At low tides, the river *E. coli* inflow travelled along the shoreline, and the discharged *E. coli* reached the sampling point, as previously demonstrated by Lam and Ahmadian (2024). After approximately 6 h, high tides were developed. The hydrostatic pressure created by the higher tidal levels resisted the river flow, so the associated *E. coli* did not reach the sampling point. The *E. coli* that arrived at the sampling point during the previous low tides decayed to lower concentrations. The lower *E. coli* concentrations at high tides were thought to be caused by the reduction of *E. coli* transport during high tides and bacteria decay. This process was repeated following the tidal cycles, and the observed *E. coli* variations are shown in Fig. 4a. The interactions above played a smaller role for the Swansea transect, explaining the absence of observed diurnal variations at the transect in Fig. 4b. The smaller role of such interaction is caused by the weaker source-receptor connection between the Swansea transect and its primary FIO source (River Tawe) compared to the stronger connection between the Aberafan transect and its primary FIO source (River Afan). Lam and Ahmadian (2024) demonstrated that contaminants from River Tawe need 10.8 h to reach the Swansea transect, while contaminants from River Afan need 4.3 h to reach the Aberafan transect. The weaker source-receptor connectivity between the Swansea transect and River Tawe causes the source-tide interaction to play a weaker role in its *E. coli* concentrations, causing less diurnal variations at the transect.



(a)



(b)

Fig. 4. Measured and modelled *E. coli* concentrations at the (a) Aberafan and (b) Swansea sampling transects in Nov 2012.

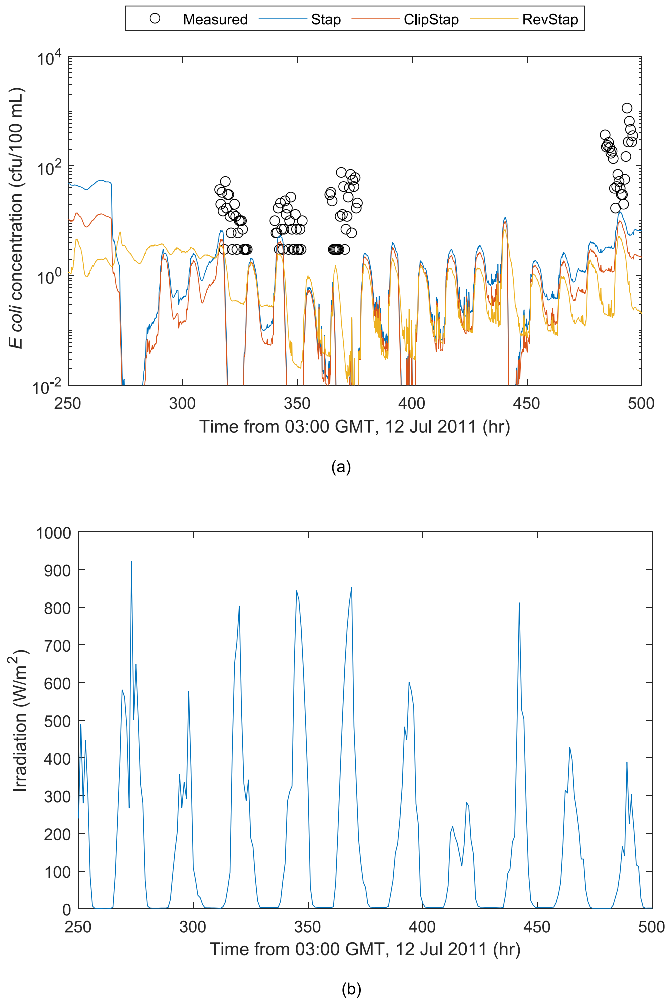


Fig. 5. (a) Measured and modelled *E. coli* concentrations at the Swansea sampling transect; and (b) irradiation within 250–500 h from 03:00 GMT, Jul 12, 2011.

5. Discussion

Novel FIO decay models, ClipStap and RevStap, were proposed to remedy the two limitations in the Stapleton et al. (2007a, b) model. While the novel models, especially the RevStap model, have improved the agreement between modelled and reported dark decay rates, these models did not constantly improve FIO prediction results. It was suggested that FIO transport and decay in nearshore coastal waters are complex processes, and improvements in both hydrodynamic and FIO decay modelling, particularly the effect of irradiation on the FIO decay rate, are needed to improve prediction. While the Stapleton model has been successfully applied in King et al. (2021) for Swansea Bay, UK, the mathematical forms of the original and improved Stapleton models are different from the form presented by Mancini (1978) and Weiskerger and Phanikumar (2020), which is based on $k_{DI} = k_d + \alpha I$. In addition, Stapleton et al. (2007a, b) model was developed from experiments under $I = 0\ W/m^2$ and $I = 260\ W/m^2$ only. More experimental work is needed to confirm and further enhance the mathematical relationship between irradiation and the FIO decay rate. Furthermore, more long-term time series of bacteria, both offshore and at sampling locations, is required to support the testing of such numerical models. Notably, Hipsey et al. (2008) developed a complex decay model that considers the effect of irradiation of different bandwidths on FIO decay. While such complex models provide advantages in terms of accuracy in some places where bandwidth data is available, a simpler model would be preferred for engineering applications, as irradiation data of different bandwidths may not always be available. This research proposes enhancement to an existing FIO decay model for nearshore coastal waters, and highlights shortfalls of existing knowledge on the effect of irradiation on bacteria decay.

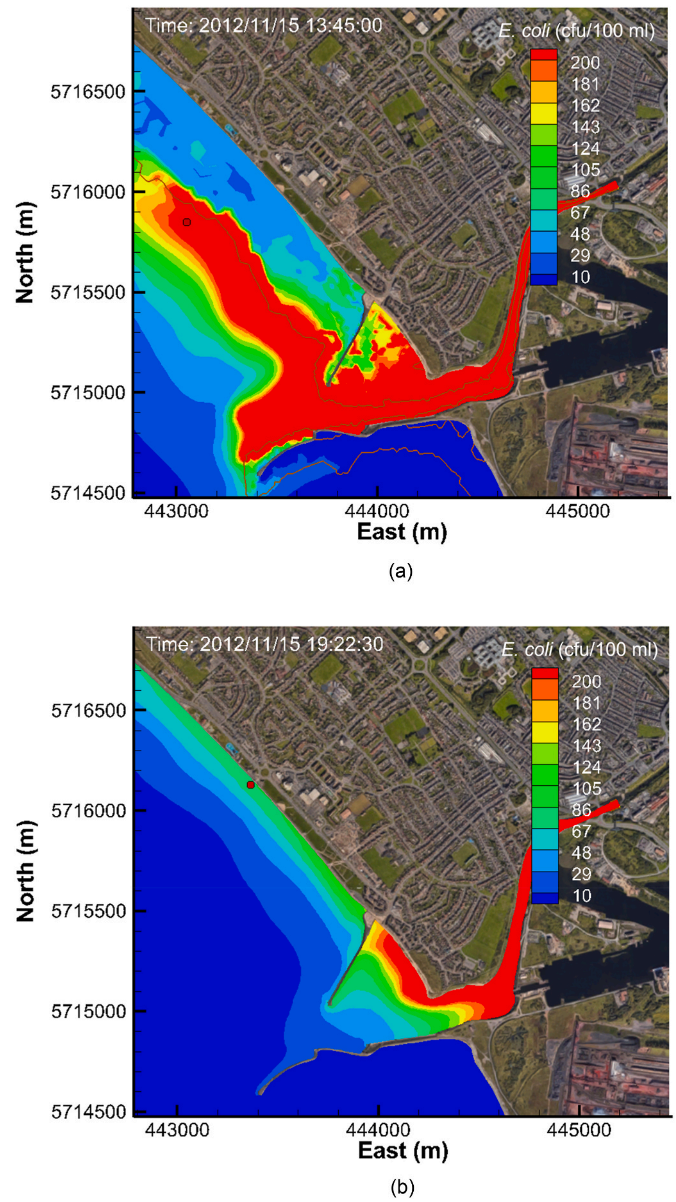


Fig. 6. *E. coli* concentrations near the Aberafan sampling transect at (a) a low tide and (b) a high tide. The circles represent the numerical sampling points along the transect at the respective time instants. The brown line shows the locations where water depth = 0.02 m. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

This research also attributes the observed diurnal variations of FIO concentrations (Wyer et al., 2018) to a combination of river and steam FIO inputs, tide action, and FIO decay. Ahmadian et al. (2013) implicitly attributed such variations solely to irradiation by developing hydro-epidemiological models with different decay rates for day-time and night-time. Their approach was successful for offshore sampling locations further away from the river inflows. In this research, the FIO sampling points are at the bathing water sites and it was showed that the effects of river/stream discharges under different tidal conditions have also become important. This research demonstrated that the physical and microbiological processes behind the diurnal variations of FIOs can be affected by the impact of the sources and are in turn site dependent. To confirm the effects of tide and discharges on diurnal FIO variations, field FIO data during night-times are necessary. It is because tide-driven FIO variations are expected to have two cycles per day and irradiation-driven variations are expected to have one cycle per day. Unfortunately, no *E. coli* sampling was conducted at night-time in the SCSC research project (Wyer et al., 2013), and future night-time field study, despite practical difficulties, is needed.

Understanding the diurnal FIO variations and the effect of irradiation on FIO decay is necessary for predicting FIO concentrations, issuing warnings to events of poor water quality, as well as evaluating the environmental impact of development projects and the effectiveness of water management strategies. Most of the currently available AI and mechanistic models are either not able to predict the diurnal variations of FIO (Gao et al., 2013a; Schernewski et al., 2014; King et al., 2021) or were trained with data having sampling intervals of the order of days (He and He, 2008; Zhang et al., 2012; Thoe et al., 2015; Zhang et al., 2018). The traditional classification for a given bathing water site is based on infrequent (two to four samples per month; NRW, 2023b) and single point sampling of *E. coli* and *Enterococci* (rBWD, European Commission, 2006). While Wyer et al. (2013) and Lam and Ahmadian (2024) have developed short-term (30-min sampling interval) data-driven FIO prediction models, further work is needed to further develop, validate, and understand fully nonlinear models. This research gives insight to the diurnal FIO variations, which is critical to assess the impact of water quality on human activities and ecology such as recreational water use, aquaculture, and aquatic lives, as well as interpreting data-driven model results.

6. Conclusion

While the Stapleton model has been successful in application in real-world cases, two limitations were identified in this research: (i) the FIO decay rates under dark or highly irradiated environments are not accurately modelled, and (ii) the effect of salinity is not included. Two modifications were attempted to remediate the limitations. The first modification, the ClipStap model, imposed a minimal decay rate of 0.025 (1/hr) to the Stapleton model. The second modification, the RevStap model, extrapolated the decay rate-irradiation slope at $I = 260\ W/m^2$ down to lower irradiation regions. The novel models, especially the RevStap model, were more successful in reproducing the decay rates reported in literature for dark environments and significantly improved the agreement between the modelled and measured decay rate. The RevSap model has provided improved modelling for high radiation. The decay models developed integrated into a mechanics-based hydro-epidemiological model, namely TELMAC 3-D, and applied to a data-rich case study. It was found that the ClipStap and RevStap improved the predictions in the Aberafan transect but not in all scenarios. Nevertheless, we believe that there is still potential for the RevStap to be improved using complementary data and through further testing. Besides the enhanced FIO decay models, this research attributes the observed diurnal variations of FIO concentrations to the combined effect of river FIO inputs, tide action, and FIO decay for sampling points at bathing water sites. This research improves hydro-epidemiological modelling, which is important for the impact assessment of water quality on human activities and ecology, by enhancing decay simulation, highlighting the existing shortfall in knowledge on the effect of irradiation and giving insight into the diurnal FIO variations.

CRediT authorship contribution statement

Man Yue Lam: Writing – review & editing, Writing – original draft, Validation, Software, Methodology, Conceptualization. **Reza Ahmadian:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: The study is carried out as a part of TidAl Range schemes as configurable Grid-scale Energy sTorage facilities (TARGET) under a grant from the Engineering and Physical Sciences Research Council (EPSRC), grant number EP/W027879/1 and the EERES4WATER (Promoting Energy-Water nexus resource efficiency through renewable energy and energy efficiency) project, which is co-financed by the Interreg Atlantic Area Programme through the European Regional Development Fund under EAPA 1058/2018. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2025.126055.

Data availability

Numerical data is available upon request. Unfortunately, the authors do not have the permission to share the field data.

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