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Evaluating the eutrophication risk of a man-made tidal lagoon

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Abstract

With increased nutrient inputs to estuaries in recent decades exacerbating their susceptibility to eutrophication, assessment of the response of individual estuaries to nutrient enrichment is attracting considerable attention. However, the impact of tidal renewable energy extraction on estuarine nutrient dynamics and the risk of eutrophication has been largely overlooked to date despite the detrimental consequences of eutrophication on ecosystem functioning. It is understood that tidal renewable energy schemes such as the proposed tidal lagoon in Swansea Bay would alter tidal flow characteristics, potentially having knock-on impacts on physical estuarine characteristics and ecological processes in the impounded area within the lagoon. This study examined the existing physical estuarine characteristics in Swansea Bay and evaluated the risk of eutrophication following the operation of the proposed tidal lagoon under the ebb-only and two-way operational modes using a simple risk assessment model. Two surveys were conducted to measure in-situ temperature, salinity, dissolved oxygen, chlorophyll-a and turbidity in surface waters as well as with depth at 12 sampling station selected to achieve maximum coverage of the proposed location of the tidal lagoon. The water column was found to be nutrient enriched and essentially vertically homogenous with no strong evidence of stratification. High dissolved oxygen, low turbidity and high phytoplankton biomass indicated by the chlorophyll-a concentrations were observed. The bay did not show any signs of eutrophication as the phytoplankton biomass did not reach the level typical of harmful algal blooms and oxygen depletion was not observed indicating that eutrophication is not present in the bay. Based on numerical model predictions, the bay was found to exhibit a moderate response to nutrient enrichment with no risk of eutrophication and no net change in its status following the operation of the lagoon under both ebb-only and two-way operational modes. However, the conditions for phytoplankton growth are likely to be more favourable under ebb-only operational mode compared to two-way operational mode due to changes in current velocities in the enclosed basin within the lagoon.

1. Introduction

Estuaries and their surrounding coastal areas are among the most biologically productive ecosystems in the world and their importance in terms of sediment stabilisation, carbon sequestration and providing fish habitats has long been recognised (Baban, 1997). This is partly due to the accumulation and availability of nutrients through continual input from rivers along with adequate light conditions inherent in shallow waters to fuel primary production (McLusky, 1996). Nutrient loading in rivers has increased considerably in recent years as a result of anthropogenic activities such as agricultural and wastewater disposal practices. Runoff from fertilizer usage in agriculture now accounts for more than half of the increase in nutrient loading in the aquatic environment (Newton et al., 2003). Other sources of nutrients to estuaries include effluent discharge from wastewater treatment plants and atmospheric deposition. Such elevated nutrient inputs could give rise to algal blooms in estuaries and a condition known as eutrophication. Eutrophication is an ecological response to the enrichment of a waterbody by nutrients, especially compounds of nitrogen and/or phosphorus, causing an accelerated growth of algae including phytoplankton to produce an undesirable disturbance to the balance of organisms present in the water column and overall water quality (CEC, 1991).

Eutrophication has been identified as one of the major threats to the integrity of estuarine ecosystems due to its attendant changes to ecosystem functioning (Vidal et al., 1999). Increased primary production and increased biomass of primary producers such as phytoplankton are known consequences of eutrophication due to nutrient enrichment (Devlin et al., 2011). Accelerated phytoplankton production limits sunlight availability to benthic aquatic plants, depletes dissolved oxygen in the water column due to decomposition of accumulation biomass resulting in hypoxic or anoxic conditions, and decreases species diversity and abundance (Chislock et al. 2013). This could give rise to shifts in invertebrate communities and permanent changes in aquatic habitats, with negative implications for pelagic and benthic fauna including fish stocks. For example, fine-grained sediments trapped by macroalgae reduces the reproductive success of marine invertebrates, which in turn reduces prey for filter-feeders including fish and water birds (Raffaeli et al. 1998). A high mortality rate of scallops was reported in Warquoit Bay due to an accumulation of algal blooms (Valiela et al. 1992). Beukema (1991) also found a decrease in the abundance of molluscs and crustaceans due to accelerated algal growth. Eutrophication could also lead to algal toxin production, with a wider range of toxic species reported in estuarine environments compared to freshwater environments which could significantly affect the edibility of local seafood (Shumway, 1990). These in turn have a variety of socio-economic consequences which could become significant overtime. Phytoplankton blooms also impede water flows and the movement of boats as filamentous algae intertwine around boat propellers making boating difficult. Other impacts of eutrophication include a decrease in the quality and aesthetic value of water bodies and thus threaten tourism and recreational activities. Nutrient enrichment does not always give rise to the abovementioned negative impacts because the response of waterbodies to nutrient enrichment is also influenced by other factors some of which include water exchange rate, water residence time, water depth, turbidity and stratification, all of which exert a control on productivity (Scott, 1999; de Jonge et al., 2002; Zhang et al., 2019). Given the detrimental ecological and socio-economic impacts and risks associated with eutrophication, assessing the response to nutrient enrichment in estuarine environments is attracting considerable attention for the effective management of transitional waters, for example in order to comply with the Water Framework Directive (WFD) (2000/60/EC). The WFD requires that "good ecological status" of surface waters be maintained. This is particularly important for those environments that are highly anthropogenically impacted such as Swansea Bay.

Swansea Bay is located in the northern reaches of Bristol Channel along the South Wales coastline (Figure 1). It has been subjected to significant anthropogenic impacts due to industrial activity and land reclamation for over a century. Under the WFD, the bay is recognised as a heavily modified waterbody, with its predicted ecological quality classified as "Bad Potential" (Callaway, 2016). In recent years, Swansea Bay has been proposed as the site for the first of a new fleet of tidal lagoons for tidal renewable energy generation because it experiences one of the world's largest tidal ranges which is up to approximately 8.4 m on a spring tide (Smith and Shackley, 2006). Tidal lagoons extract energy from the rise and fall of the tide and converts it to electricity using hydro-turbines housed within impoundment structures. The hydro-turbines can be operated to generate electricity during flood tide (flood operational mode), ebb tide (ebb operational mode) and on both flood and ebb tides (two-way operational mode). During flood and ebb modes of operation, electricity is generated once per tidal cycle and during two-way, electricity is generated twice per tidal cycle (Waters and Aggidis, 2016). It is expected that a fleet of tidal lagoons could supply 8% of the UK's electricity (Elliot, 2019). The proposed lagoon was granted planning permission by the UK's

Department of Energy and Climate Change (DECC) and its feasibility and scope has undergone extensive examination by the independent Hendry Review commissioned by the UK government and published in 2017 (Hendry, 2017). It has been estimated that the lagoon could generate power for 155,000 homes and save around 236,000 tonnes of CO_2 per annum (Waters and Aggidis, 2016; Petley and Aggidis, 2016). Additional benefits including flood protection, integrated aquaculture, tourism and recreational activities are also suggested (Elliot, 2019). However, uncertainty exists as to its future due to environmental concerns over potential impacts of the lagoon as highlighted in the Hendry review, coupled with its high initial capital costs and its possible inability to generate electricity at a competitive price (Hendry, 2017; Elliot et al., 2018; Panio et al., 2019). More recently, the UK government of the time stated that the £1.3bn Swansea Bay tidal lagoon does not represent good value for money (TLP, 2018). There is no man-made electricity generating tidal lagoon currently and as such there is no lagoon specific operational data on their environmental impacts available. Several modelling studies on tidal lagoons have recently been undertaken but they have largely focussed on hydrodynamic and morphological changes as well as optimising operation and electricity generation (Neill et al., 2018; Thomas et al., 2015; Angeloudis and Falconer, 2017; Angeloudis et al., 2016, 2018; Vouriot et al., 2019; Xue et al., 2019). Angeloudis et al., (2016) found that the current velocities in the impounded water column within the lagoon could be increased due to advective accelerations, turbulent wakes and a stagnation of flow upstream creating recirculation zones. Such a change in the tidal flow characteristics could have significant knock-on effects on the local hydro-environment and ecological processes. A few studies have assessed changes to pollutant dispersion and the potential for carbon sequestration (e.g. Evans and Langley, 2017; Piano et al., 2019), providing some insights into the possible impacts on ecosystem functioning but the likelihood of eutrophication occurring is largely unknown. In this study, we examined the physical estuarine characteristics in Swansea Bay in order to provide a snapshot of its prevailing status and applied a simple risk assessment model to evaluate the risk of eutrophication following the operation of a tidal lagoon in the bay under two different modes of operation: ebb-only and two-way operational modes. To achieve this, model predictions of dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) concentrations as well as phytoplankton biomass chlorophyll concentrations under a no lagoon scenario were compared to those under the two different modes of operation. The phytoplankton primary production under the different scenarios were also assessed and will aid in further reducing the uncertainty and addressing regulatory concerns associated with the eventual development and integration of tidal lagoons.

2. Materials and methods

2.1. Study area

Swansea Bay is located to the north of the Bristol Channel. The Bristol Channel separates the coastline of South Wales, to the north, from that of England, to the south. Swansea Bay has a shoreline of about 29 km in length. It is semi-diurnal, with a spring tidal range of approximately 8.5 m and tidal currents which take the form of a rectilinear, reversing offshore flow with an anticlockwise eddy in the western inner part and an area of divergence on the eastern side of the embayment (Smith and Shackley, 2006). It is relatively shallow, with a water depth that ranges from less than 5 m in the upper reaches of the bay to about 20 m in the outer regions and tidal currents of around 0.52 m s^{-1} (Collins et al., 1980). It is predominantly characterised as a sandy embayment, with fine and medium sand and gravel in the outer regions of the bay and increasing proportions of fine-grained sediment occurring close nearshore to the west and in shallow intertidal areas (Thomas et al., 2015; TLSB, 2016). The catchment area draining into the bay accommodates a large human population (in excess of a quarter of a million) as well as several industries. Historically, the bay was linked to the coal, steel and fishing industries, with the main current industrial activity centred in Port Talbot. The riverine nutrient inputs to the inner bay are from Rivers Tawe and Neath from the north and River Avon from the east, with approximately two thirds derived from Rivers Tawe and Neath which are affected by anthropogenic activities (Chubb et al., 1980). The inner bay also receives treated effluent inputs from a sewage treatment works which serves the Swansea area. Infrastructure development have severely modified the bay and it is characterised by a series of coastal protection measure, fronted by wide intertidal mud and sand flats (Callaway, 2016). Some of these have been recognised as areas of major conservation importance and given national designations including Sites of Special Scientific Interests (i.e. Crymlyn Burrows and Blackpill) and Special Area of Conservation (i.e. Kenfig). The proposed lagoon comprised a 9.5 km long embankment with an impounded area of 11.5km^2 located between River Tawe and Neath (Figure 1). As mentioned above, the lagoon was granted planning consent in 2015 and is estimated to have an installed capacity of 320 MW (Petley and Aggidis, 2016; Waters and Aggidis, 2016). Although proposed as a pathfinder for future lagoons, it has faced major challenges which have stalled progress on the construction and operation of the lagoon.

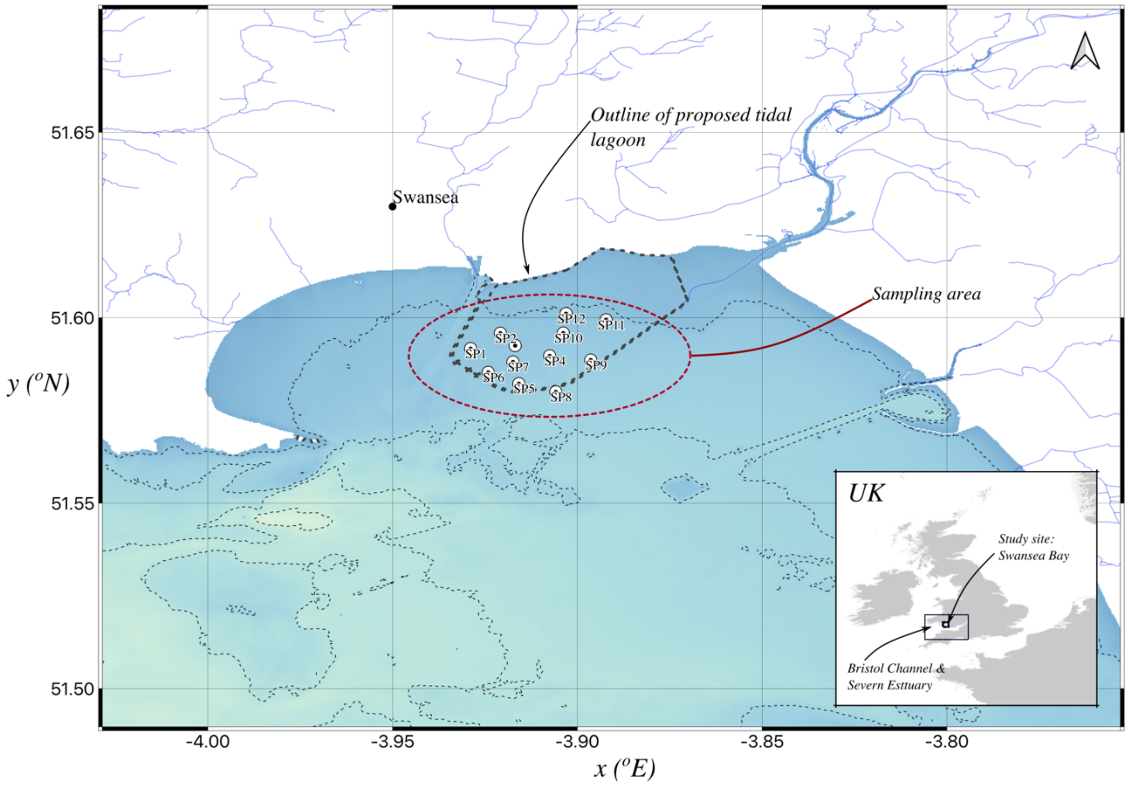

Figure 1. Map showing the location of Swansea Bay and the sampling stations (SP1 – SP12) within the proposed region of the tidal lagoon.

2.2. Field sampling and analysis

Two surveys were conducted using a research vessel equipped with a winch, a Global Positioning System (GPS) and a depth echo-sounder over a two-day period at 12 sampling stations in July 2018. The stations were selected to achieve maximum coverage of the proposed location of the tidal lagoon within Swansea Bay (Figure 1). Each survey was carried out during daylight hours, starting around low water along the neap to spring tidal circle. At each sampling point, an advanced multiparameter sonde (RS Hydrolab, UK) was deployed using a winch to measure in-situ temperature (°C), salinity (PSU), dissolved oxygen (DO) (mg l^{-1}), chlorophyll-a florescence (mg Chl-a m^{-3}) and turbidity, producing a depth profile of the measured parameters. Prior to their usage, the sonde sensors were calibrated as per the manufacturer's instructions and the precisions were ± 0.05 °C temperature, ± 0.1 PSU and ± 0.5 % DO. Water samples were also collected in triplicates for dissolved inorganic nitrogen analysis at each station using a 1.7-L Niskin water sampler near the surface at a depth of ~ 30 cm. The collected water samples were immediately filtered on board using cellulose acetate membrane filters (nominal pore size of 0.45 μm) attached to a sample-rinsed syringe. The filtered samples were stored in the dark on ice in an icebox and returned to the laboratory where they were frozen until analysed using ion chromatography (Dionex ICS-2500 system).

2.3. Tidal lagoon operation

A tidal lagoon creates an artificial height difference of water levels (head) between the impounded water body and the open sea. Water is then released though turbines when appropriate head differences have been created to generate electricity. Figure 2 is an illustration of the generalised operation of a tidal lagoon highlighting the holding, generation, slucing and pumping stages. During flood tide, sea level rises, while both sluice gates and turbine wicket gates are closed, increasing the difference between sea level and lagoon water level (head). When the head has increased enough to reach a starting head level, turbine wicket gates are opened, and power is generated as water flows into the lagoon. As head decreases, and water flow slows, sluice gates are opened to increase flow rate, to achieve a more constant flow rate and therefore more constant power generation. This happens until the head decreases to a minimum head level, at which point both turbine wicket and sluice gates close, and lagoon water level stays constant during a holding stage, as sea level drops during ebb tide. When the head increases to starting head level again, the process repeats, except that water flows out of the lagoon into the sea. Pumping, which is highlighted in Figure 2, uses the turbines to pump water in or out of the lagoon during holding stage, to increase head level. This increases the amount of time over which power production occurs. The overall mode of operation is determined by the length of the various stages of the operational cycle:

- holding: both turbine wicket gates and sluice gates are closed to prevent water leaving or entering the lagoon, and build up head
- generation: energy is produced as turbine wicket gates open and water flows downstream through the turbines
- sluicing: as flow rate of water decreases with head, sluice gates open to allow for extra water flow
- pumping: an optional process that increases head and therefore duration of energy generation.

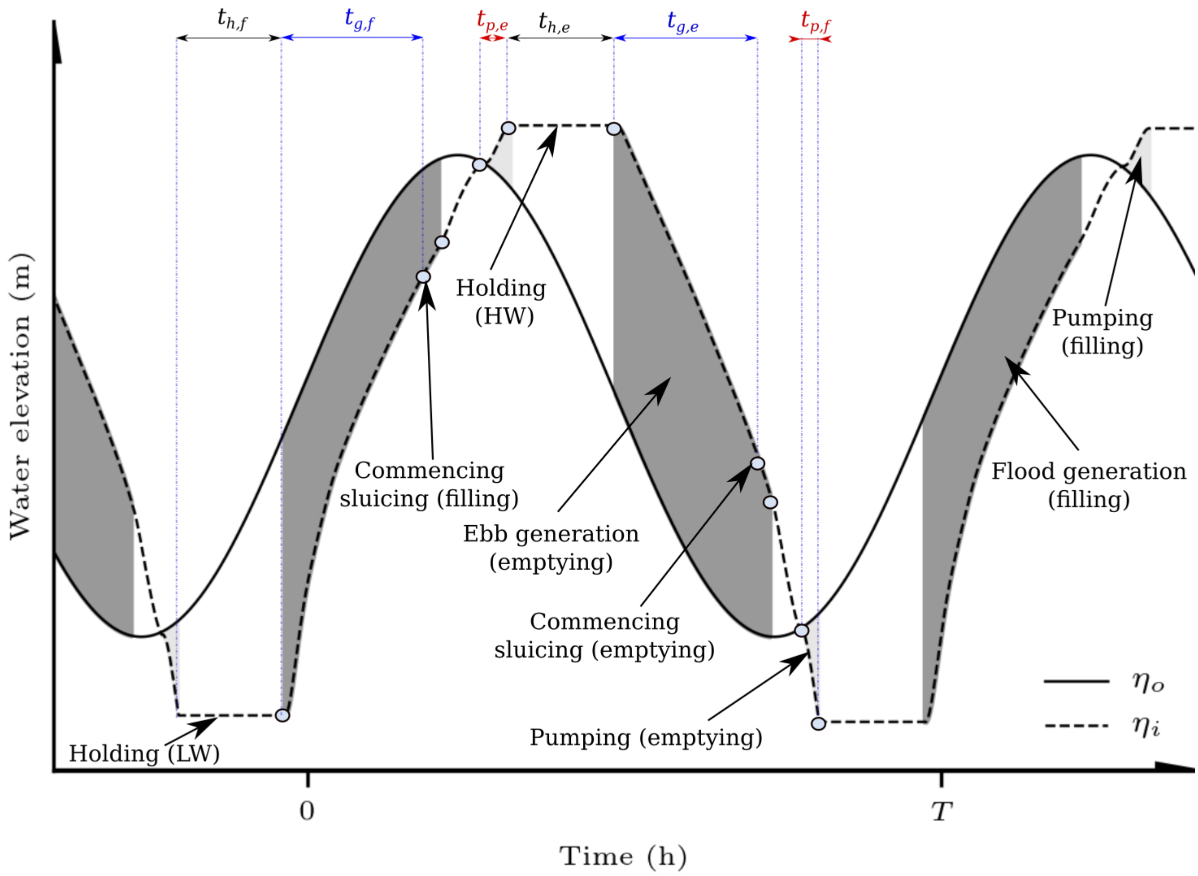

Figure 2. Generalised operation of a tidal lagoon over a tidal period with shaded areas representing energy generation periods (Harcourt et al., 2019). The holding ($t_{h,f}$, $t_{h,e}$), generation ($t_{g,f}$, $t_{g,e}$) and pumping ($t_{p,f}$, $t_{p,e}$) stages are highlighted.

2.4. Model setup

The simple risk assessment model used in this current study has previously been used to assess the susceptibility of estuarine and coastal waters to eutrophication (Tett et al., 2003; Painting et al., 2007; Kadiri et al., 2014; Zhang et al., 2019). It was initially developed by the UK's Comprehensive Studies Task Team. However, it has been refined by Tett et al. (2003) and Painting et al. (2007) to allow for its prediction of maximum phytoplankton chlorophyll biomass (X_{max}) and maximum primary production (P_{max}) which are key parameters for assessing the impacts of nutrient enrichment. The conceptual structure of the model can be found in Painting et al (2007). The maximum phytoplankton chlorophyll biomass is predicted by:

$$X_{max}\ (\text{mg Chl m}^{-3}) = X_0 \times q + S_{eq} \tag{1}$$

where X_0 is the phytoplankton chlorophyll biomass concentration (mg m^{-3}) in the adjacent seawater, q is the phytoplankton chlorophyll yield from dissolved nutrients and S_{eq} is the equilibrium nutrient concentration. The yield q is an ecosystem parameter and CSTT (1997) and Painting et al (2007), based on observational yield data reported by Gowen et al (1992), recommended a yield of 1.05 mg Chl (mmol, N)$^{-1}$ for dissolved inorganic nitrogen (DIN which was used in this study. The yield of 1.05 mg Chl (mmol, N)$^{-1}$ has been previously reported in other studies (Edwards et al., 2003). In the case of dissolved inorganic phosphorus (DIP), the yield q was set at 16.8 mg Chl (mmol, P)$^{-1}$, allowing the phytoplankton yield atomic N:P ratio to correspond with the Redfield ratio of 16:1, which is the ratio required for balanced phytoplankton growth. In the absence of measured chlorophyll concentrations in adjacent seawater, the X_0 obtained from Morris and Mantoura (1980) where chlorophyll concentrations were measured the region outside the bay in the adjoining Bristol Channel was used. The S_{eq} for DIN and DIP was calculated using:

$$S_{eq}\ (\mu\text{M}) = S_0 + (S_i/(E \times V)) \quad (2)$$

where S_0 is the nutrient input concentration from the adjacent seawater (µM), S_i is the total nutrient inputs from sources other than seawater (kmol d^{-1}), E is the water exchange rate (d^{-1}) and V is the volume of water in the basin (m^3), with the basin is defined in this study as the impounded area within the lagoon. The mean volume of water within the lagoon was estimated using a 0-D model developed by Angeloudis et al. (2016) to simulate the operation of the Swansea Bay lagoon. The model is based on backward finite difference method and it has been used in Lewis et al., (2017), Angeloudis et al. (2018) and Harcourt et al. (2019) to simulate operational conditions for tidal lagoon schemes. Table 1 shows the parameter values used in the model to simulate a time series of water volume within the lagoon for the different lagoon operation stages, with the values set to zero for the simulation of the no lagoon scenario. Details on the formulations used in the model and the tidal lagoon specifications are expanded in Angeloudis et al., (2016) and are omitted here for brevity. The water exchange rate, E, was estimated using the method applied in Balls (1994) and Tett et al. (2003) in which the exchange rate is related to the volume of the estuary (V, m^3) and riverine inputs (F, m^3 d^{-1}) by the expression: E (d^{-1}) = $F/V \times C_0/C_0 - C$, where C_0 is the salinity of coastal seawater (taken as 34.5 practical salinity units) and C is the salinity in the bay. The total nutrient inputs used in this study were estimated as the product of annual mean daily river flow for the two main rivers which feed into the inner bay (i.e. Tawe and Neath) and mean annual concentration of each nutrient in the rivers from monthly monitoring data collected by Natural Resource Wales from 2014 to 2016 (Table 2). It is assumed that the available nutrient inputs from the two main rivers feed into the water column within the lagoon and are converted to phytoplankton chlorophyll biomass at the fixed yield of phytoplankton chlorophyll from nutrient inputs. The method used for estimating the nutrient input loads has been reported to produce the most accurate estimates relative to those calculated using other methods (De Vries and Klavers, 1994). The S_0 obtained from Morris and Mantoura (1980) where DIN and DIP concentrations were measured in the region outside of the bay in the adjoining Bristol Channel were used in this study.

The potential maximum primary production was predicted via:

$$P_{max}\ (\text{gC m}^{-2}\ \text{y}^{-1}) = (X_{max} \times \mu(I) \times C{:}Chl \times 365 \times h)/1000 \tag{3}$$

where X_{max} is maximum phytoplankton chlorophyll biomass (mg m^{-3}), $\mu(I)$ is light controlled phytoplankton growth rate (d^{-1}), $C{:}Chl$ is the carbon to chlorophyll ratio over an annual cycle (i.e. 365 days), h is depth of the photic zone of the water column in the enclosed basin (m). $\mu(I)$ is calculated as:

$$\mu(I)\ (\text{d}^{-1}) = \alpha^B \times (I - I_c) \tag{4}$$

where α^B is effective photosynthetic efficiency d^{-1}(μE m^{-2} s^{-1})$^{-1}$ at low illumination, I is the 24-hour mean photosynthetically available radiation (PAR, μE m^{-2} s^{-1}) and I_c is the compensation PAR (μE m^{-2} s^{-1}). I (μE m-2 s-1) was calculated as follows:

$$I\ (\mu\text{E m}^{-2}\ \text{s}^{-1}) = (1 - m_0).m_1.m_2.I_0.((1 - e^{-K_d h})/(K_d.h)) \tag{5}$$

where I_0 is the annual 24-hour mean sea-surface solar radiation (W m^{-2}), m is used to convert the mean solar radiation into a mean PAR (i.e. m_0 is sea albedo, m_1 is total solar energy to PAR photons (μE J^{-1}), m_2 is the fraction of the surface PAR penetrating light), K_d is the diffuse light attenuation (m^{-1}) and h is the depth of the photic zone in the water column. The assigned values of α^B, I_c, m_1 and m_2 recommended by CSTT (1997) and Tett et al. (2003) are given in Table 1. I_0 was set to the default value of 150 W m^{-2} s^{-1} as suggested by Painting et al. (2007). The K_d was calculated using a linear regression model (Devlin et al., 2008), which allows for the estimation of K_d from suspended particulate matter (SPM) concentrations between 0 – 300 mgl^{-1}: K_d = 0.08596 + (0.06729 × SPM) (r^2=0.98). Given that SPM concentrations were not measured in this study, concentration of 8 mg l^{-1} for the no lagoon scenario obtained from TLSB, (2016) was used. In the case of the ebb-only and two-way operational modes, the suspended sediment concentrations of 8.4 mg l^{-1} and 9.6 mg l^{-1}, were used on the basis of a predicted 5% and 20% increase in current speed within the enclosed basin under ebb-only and two-way operational modes (Angeloudis et al., 2016; Xue et al. 2019). A change in current speed would precipitate a change in the resuspension of sediments, affecting suspended sediment concentrations in the water column. The depth of the photic zone was set at 6 m given the bay is relatively shallow and the photic zone is between 5 m and 6 m (Joint, 1980).

Table 1. Parameter values used for the different stages of the lagoon operation in the 0-D model.

	Operational modes	
	Ebb-only	Two-way
Ebb generation time before sluicing (hours)	6.00	2.50
Flood generation time before sluicing (hours)	0.00	2.50
Ebb holding mode duration (hours)	3.50	3.00
Flood holding mode duration (hours)	0.00	3.00
Ebb pumping duration (hours)	0.00	0.00
Flood pumping duration (hours)	0.00	0.00
Pumping head (m)	0.00	0.00

Table 2. Risk assessment model parameters and their values under the no lagoon, ebb-only and two-way operational mode scenarios.

Symbol		No lagoon	Ebb-only	Two-way	Units
	Site-specific parameters				
S_i	total nutrient input from sources except seawater (DIN)	35.7	35.7	35.7	kmol d^{-1}
	total nutrient input from sources except seawater (DIP)	0.74	0.74	0.74	kmol d^{-1}
S_0	seawater nutrient concentration (DIN)	7.3	7.3	7.3	µM
	Seawater nutrient concentration (DIP)	0.5	0.5	0.5	µM
X_0	seawater chlorophyll concentration	2	2	2	mg m^{-3}
V	volume of enclosed basin	68.4	78.9	70	10^6 m^3
E	water exchange rate	0.11	0.09	0.10	d^{-1}
	Standard parameters				
q	chlorophyll yield (from DIN)	1.05	1.05	1.05	mg Chl (mmol N)$^{-1}$
	chlorophyll yield (from DIP)	16.8	16.8	16.8	mg Chl (mmol P)$^{-1}$
α^B	effective photosynthetic efficiency = $\alpha_m{}^X q_a{}^N Q_{max}(1 - \eta)/(1 + b)$	0.006*	0.006*	0.006*	d^{-1}(µE m^{-2} s^{-1})$^{-1}$
C:Chl	carbon to chlorophyll ratio (annual)	40	40	40	
I_c	compensation irradiance = $r_{0a}(1 - \eta) + r_{0h}\eta(1 + b_a))/(\alpha_m\varkappa)$	5*	5*	5*	µE m^{-2} s^{-1}
	Used to calculate standard parameters				
α_m	algal (chlorophyll related), nutrient replete, photosynthetic efficiency	0.042*	0.042*	0.042*	mmol C (mg chl)$^{-1}$ d^{-1} (µE m^{-2} s^{-1})$^{-1}$
b	rate of increase of (microplankton) respiration with growth = $b_a(1 + b_h\eta) + b_h\eta$	1.4*	1.4*	1.4*	
b_a	rate of increase of (autotroph) respiration with growth	0.5*	0.5*	0.5*	
b_h	rate of increase of (heterotroph) respiration with growth	1.5*	1.5*	1.5*	
η	'heterotroph' fraction = microheterotroph carbon biomass/total microplankton biomass	0.4*	0.4*	0.4*	
Q_{maxa}	maximum autotroph nitrogen content	0.2*	0.2*	0.2*	mmol N (mmol C)$^{-1}$
$^xq_a{}^N$	autotroph chlorophyll nitrogen ratio	3*	3*	3*	mg Chl (mmol N)$^{-1}$
r_{0a}	autotroph basal respiration (at zero growth)	0.05*	0.05*	0.05*	d^{-1}
r_{0h}	heterotroph basal respiration (at zero growth)	0.07*	0.07*	0.07*	d^{-1}
	Submarine optics				
I_0	mean annual sea-surface 24-h solar radiation	150*	150*	150*	W m^{-2}
m_0	sea albedo	0.06*	0.06*	0.06*	
m_1	conversion from total solar energy to PAR photons	1.909*	1.909*	1.909*	µE J^{-1}
m_2	fraction of the surface PAR that is penetrating light	0.4*	0.4*	0.4*	
h	Depth of the photic zone	6	6	6	m
K_d	diffuse attenuation for PAR	0.69	0.72	0.81	m^{-1}

*Default values based on data analysis and sensitivity analysis (see Painting et al. (2007) and Tett et al (2003)).

3. Results and discussion

3.1. Physical characteristics

Table 3 shows the range and mean values of temperature, salinity, turbidity, DIN, pH, DO and chlorophyll a in surface water. The surface water temperatures were relatively constant, with a 0.7 °C and 0.5 °C difference between the minimum and maximum values in the first and second surveys, respectively. Such narrow temperature ranges indicate that solar heating did not result in significant heat accumulation in the water column. The salinity ranged from 33.1 – 33.8 PSU and 33.5 – 33.8 PSU in the first and second surveys, respectively (Table 3). These values are in-line with estuarine conditions and those previously reported for the bay in other studies (TLSB, 2014). The salinity values are a function of tidal influx, freshwater inputs from River Tawe and Neath which drain into the bay and the circulation pattern within the bay where tidal currents take the form of a rectilinear, reversing offshore flow with an anticlockwise eddy in the western inner part and an area of divergence on the eastern side of the bay (Collins et al., 1980; Smith and Shackley, 2006). Depth profiles of temperature and salinity at the stations are shown in Figure 3. With the exception of one station, there were no marked changes in temperature and salinity with depth and the distributions were essentially vertically homogenous suggesting that the water column was largely well-mixed, with marginal stratification at one station. The commonly used Simpson-Hunter stratification parameter, Sp (Simpson and Hunter, 1974) provides a useful and convenient physical framework for assessing potential vertical stratification in the water column (Bisagni, 1992; Hu et al., 2003; Holt and Umlauf, 2008). It is a convenient measure of the relative degree of vertical mixing in highly tidally dynamic environments and is given by:

$$S_p = \log_{10}[h / U^3]$$

where h is water depth (m) and U is current speed (ms^{-1}). The boundary between vertically well-mixed and thermally stratified water is given by S_p = 1.5, with the water column considered to be stratified for regions for which S_p is greater than 1.5. With a water depth of 5m and current speed of 0.52 m s^{-1}, S_p was found to be 1.55 (i.e. slightly greater than 1.5) suggesting that areas in the bay may be subject to some degree of stratification as illustrated in the station 1 results. The absence of stratification in the majority of the stations is likely due to wind and wave induced mixing. The vertical mixing in the bay further precludes stratification development as Uncles and Joint (1983) found that the vertical mixing timescale in the bay is too short to allow stratification to occur. Stratification has implications for nutrient availability in the water column, although in a shallow bay such as Swansea Bay, the effect is believed to be less crucial.

The turbidity levels were found to be relatively low, with little or no variation with depth (Table 3 and Figure 4a, b and c). Such low turbidity is likely due to weak tidal currents resulting in the resuspension of small proportions of bottom sediments and low concentrations of suspended sediment in the water column. Syvitski et al (2000) found a correlation between suspended sediment load and basin scale length for a range of waterbodies worldwide. Given the length of the bay, there is likely to be relatively low terrestrial sediment inputs from tributary rivers. Stoner (1977) found that approximately 260 tonnes day^{-1} (95,000 tonnes year^{-1}) of suspended sediment is discharged into the bay from rivers Tawe, Neath and Afan which is low in comparison to the Bristol Channel which receives as high as 9 x 10^6 tonnes of suspended sediment from its tributary (Manning et al., 2010). The construction of a barrage in river Tawe in recent decades may have reduced the suspended sediment inputs into the bay. In addition, sediment from the shallow intertidal areas within the bay itself may be resuspended into the water column, although it is likely to be less than the inputs from the tributary rivers that feed into the bay. Sediments may also be advected into the bay from the adjoining Bristol Channel and kept in suspension particularly during spring tides, and concurrently with strong south westerly winds.

The DO concentrations in surface waters in the bay were high with values ranging from 8.6 to 9.3 mgl^{-1} and 8.7 to 9.6 mgl^{-1} in the first and second survey, respectively (Table 3). These values correspond to oxygen saturation in the first and second surveys of 108% – 124% and 109% to 128% showing that the bay is supersaturated, reflecting high in-situ photosynthetic production of oxygen. High photosynthetic activity often observed in estuarine environments in warm spring and summer months give rise to supersaturation (Cabecadas et al., 1999). The high temperatures, low turbidity and shallow water depth allows for sunlight penetration in the bay to provide adequate conditions for phytoplankton photosynthesis. In addition, there was no significant change in DO concentrations with depth (Figure 4d, e and f). This implies that the water column is well-oxygenated, with no evidence of hypoxic or anoxic conditions at depth. DO concentration is widely used to assess the impact due to nutrient enrichment and the risk of eutrophication in estuaries, with concentrations below 4 – 6 mgl^{-1} considered to be an indicator of oxygen depletion which negatively affects aquatic fauna including fish stocks (Nixon et al., 1995; OSPAR, 2005; Best et al., 2007; Devlin et al., 2011). The DO concentrations in the bay were greater than 6 mgl^{-1}, indicating the absence of oxygen depletion in the bay.

The DIN concentrations were found to be high, with values which ranged from 14.4 – 20.2 µM and 13.4 – 17.4 µM in the first and second surveys, respectively (Table 3). Such high concentrations suggest nutrient enrichment. Historically, the bay has been reported to be nutrient enriched (Humphrey et al., 1980). The enrichment was attributed to the heavy industrialisation of the area bordering the bay and the wide use of the bay as a disposal site for untreated industrial effluents and domestic sewage from outfalls. Although there has been closures of some industries and the decommissioning of a major sewage outfall in recent years which would have reduced nutrient inputs into the bay and improved the overall water quality, the high observed DIN concentrations suggest that the bay remains nutrient enriched. Smith and Shackley (2006) did not observe a significant change in nutrient concentrations in the bay after the closure of the sewage outfall. The enrichment is believed to arise from riverine inputs from the main rivers (i.e. Tawe, Neath and Afan) which feed into the bay. Chubb et al. (1980) identified these rivers as significant sources of nutrient to the bay as some drain catchments extensively used for agriculture and with generally high DIN soil. This is further augmented by effluents from current industrial activity in the area. In addition, the bay receives treated effluents discharged from a new waste-water treatment plant which is located in the region where this study was conducted. It is important to note that nutrient enrichment does not present a problem in itself. Negative effects are only likely if significant proportions of the available nutrients are assimilated by phytoplankton and favourable conditions exist for phytoplankton growth (Kocum et al., 2002).

The high DIN concentrations were complemented with high Chl-a concentrations which increased with depth (Table 3 and Figure 5). These high Chl-a concentrations are indicative of phytoplankton biomass but do not reach the level typical of nuisance algal blooms. The concentrations were within the range previously observed in the bay (1.7 – 26.6 mg Chl-a m^{-1}, Paulraj and Hayward, 1980; 6 – 12 mg Chl-a m^{-1}, Joint and Pomeroy, 1981). It is well known that phytoplankton biomass and its rate of production are controlled by nutrient and light availability as well as water residence time (Underwood and Kromkamp, 1999; Kocum et al., 2002). In this study, the high phytoplankton chlorophyll biomass is a direct consequence of the nutrient enrichment and light availability evidenced by the high DIN concentrations and low turbidity levels in the bay, coupled with the residence time of water in the bay providing favourable conditions for growth. The low turbidity levels allow sunlight penetration for phytoplankton photosynthesis in the water column as the transformation of nutrients into biomass requires solar energy to drive photosynthesis (Underwood and Kromkamp, 1999). During this study, available data indicated that the residence time of water in the bay (around 9 days) is typical of those for bays and bar-built estuaries (Owen et al., 1986; Nixon et al., 1996; Painting et al., 2007). The water residence time is equivalent to a water exchange rate of 0.1 d^{-1} which is believed to allow sufficient time for phytoplankton to grow. Although the timescale for phytoplankton cell divisions were not evaluated in this study, Brand and Guillard (1981) found that phytoplankton divide rapidly, with doubling times in the order of 1 day which is less than the water residence time in the bay. It is important to note that algal blooms and signs of eutrophication were not visibly observed during this study. However, a significant change in nutrient concentrations, turbidity and water residence time could make the bay more susceptible to eutrophication.

Table 3. Temperature, salinity, DO, turbidity, DIN and chlorophyll-a concentrations at the sampling stations in Swansea Bay. DIN measurements were not taken made at depth and turbidity measurements were not taken during survey 2.

	Survey 1		Survey 2	
	Range (surface)	Mean (depth)	Range (surface)	Mean (depth)
Temperature (°C)	19.3 - 21.2	19.9	19.9 - 22.2	20.1
Salinity (PSU)	33.1 – 33.8	33.8	33.5 - 33.8	33.8
DO (mg l^{-1})	7.8 – 8.8	9.1	7.7 – 9.5	9.5
Turbidity	0.1 – 1.8	1.2	-	-
DIN (μM)	14.4 – 20.2	-	13.4 – 17.4	-
Chl-a (mg m^{-3})	2.4 – 6.1	10.6	5.6 – 14.3	15.3

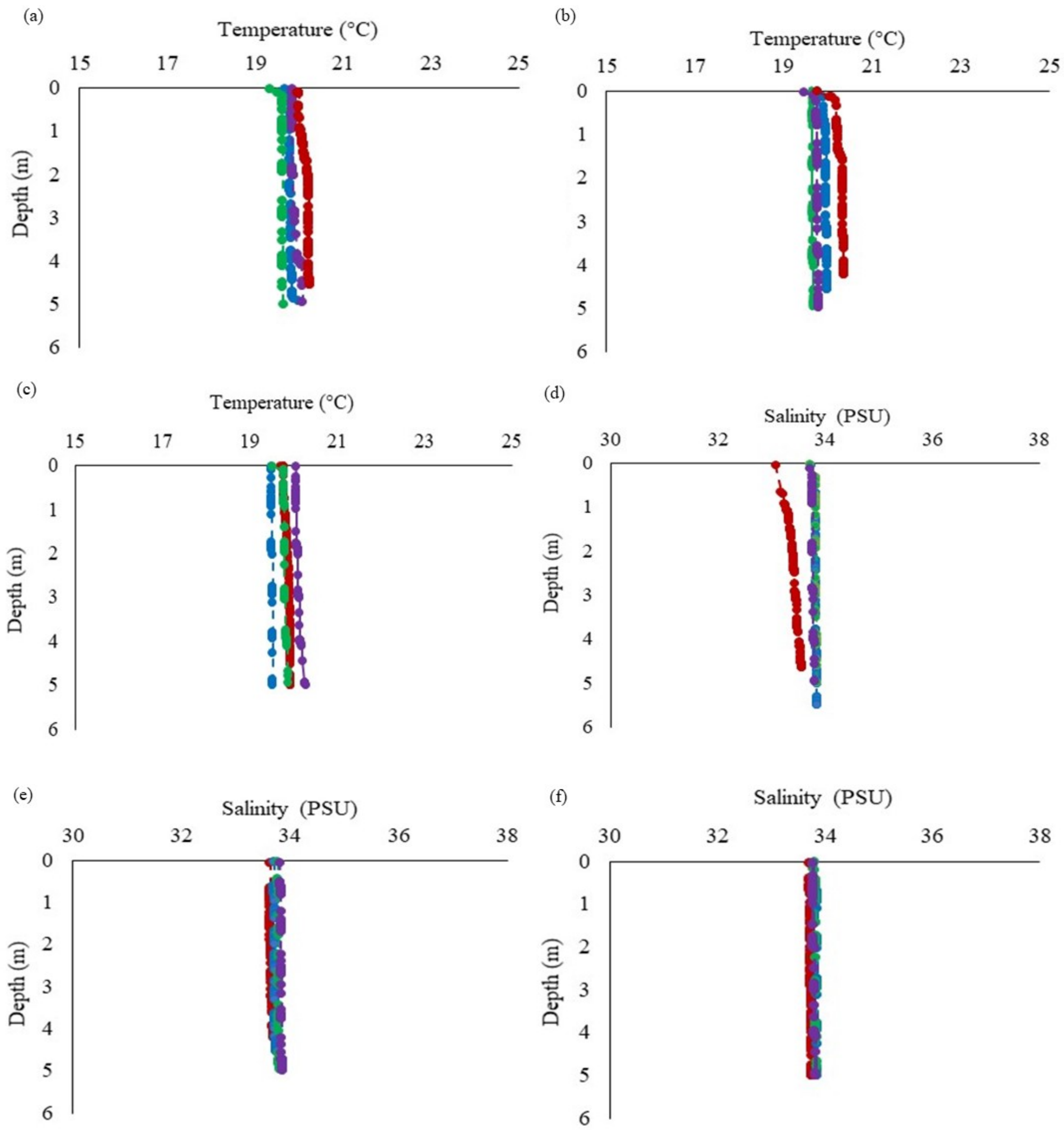

Figure 3. Depth profiles of temperature (a, b, c) and salinity (d, e, f) in Swansea Bay. Stations 1, 6, 9, 11 are shown in 3a and d; stations 2, 3, 5, 7 are shown in 3b and e and stations 4, 8, 10, 12 are shown in 3c and f. These are for survey 1 only. Profiles for survey 2 are not shown as the trends were similar to those for survey 1.

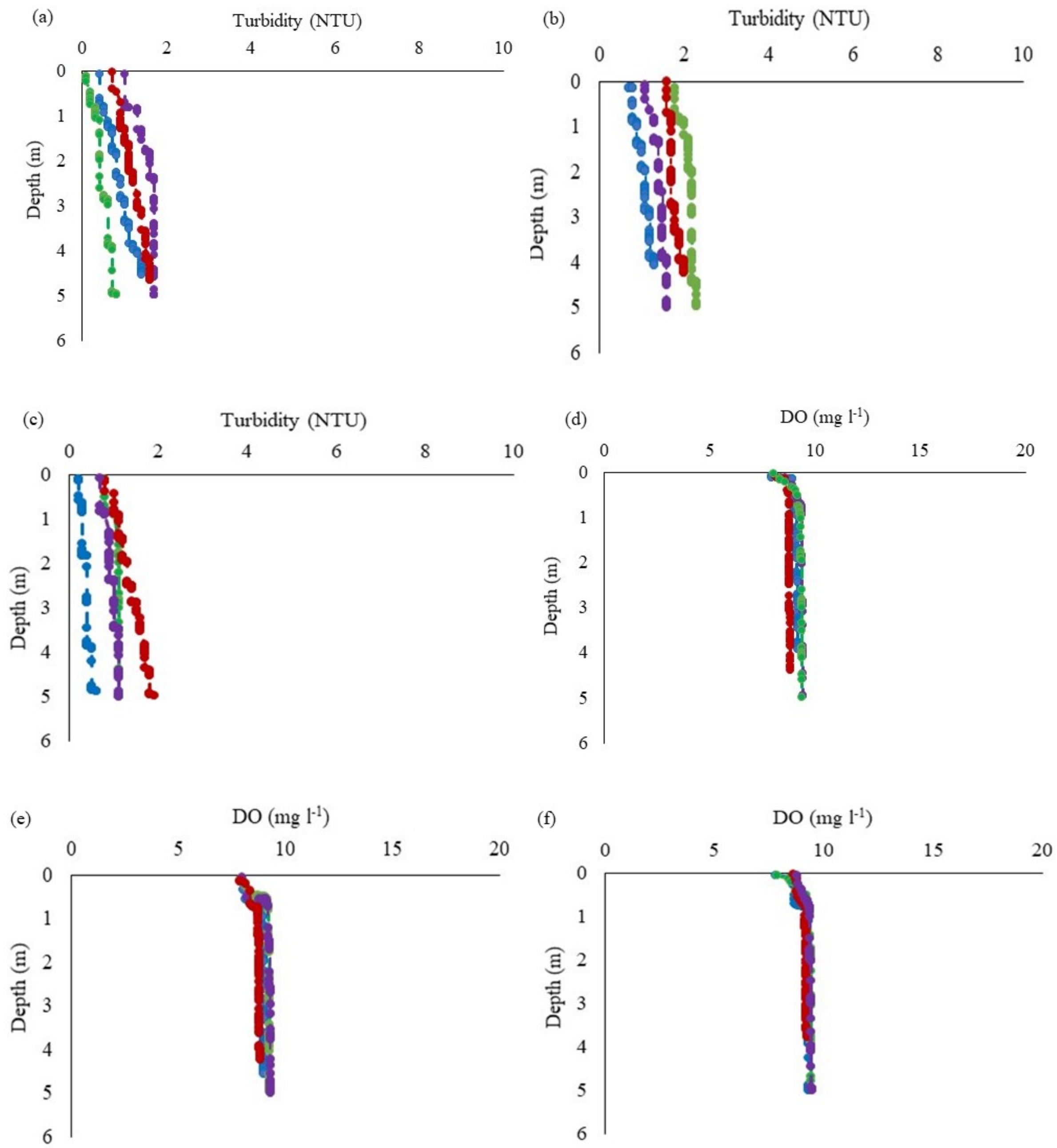

Figure 4. Depth profiles of turbidity (a, b, c) and DO (d, e, f) in Swansea Bay. Stations 1, 6, 9, 11 are shown in 4a and d; stations 2, 3, 5, 7 are shown in 4b and e and stations 4, 8, 10, 12 are shown in 4c and f. These are for survey 1 only. Profiles for survey 2 are not shown as the trends were similar to those for survey 1.

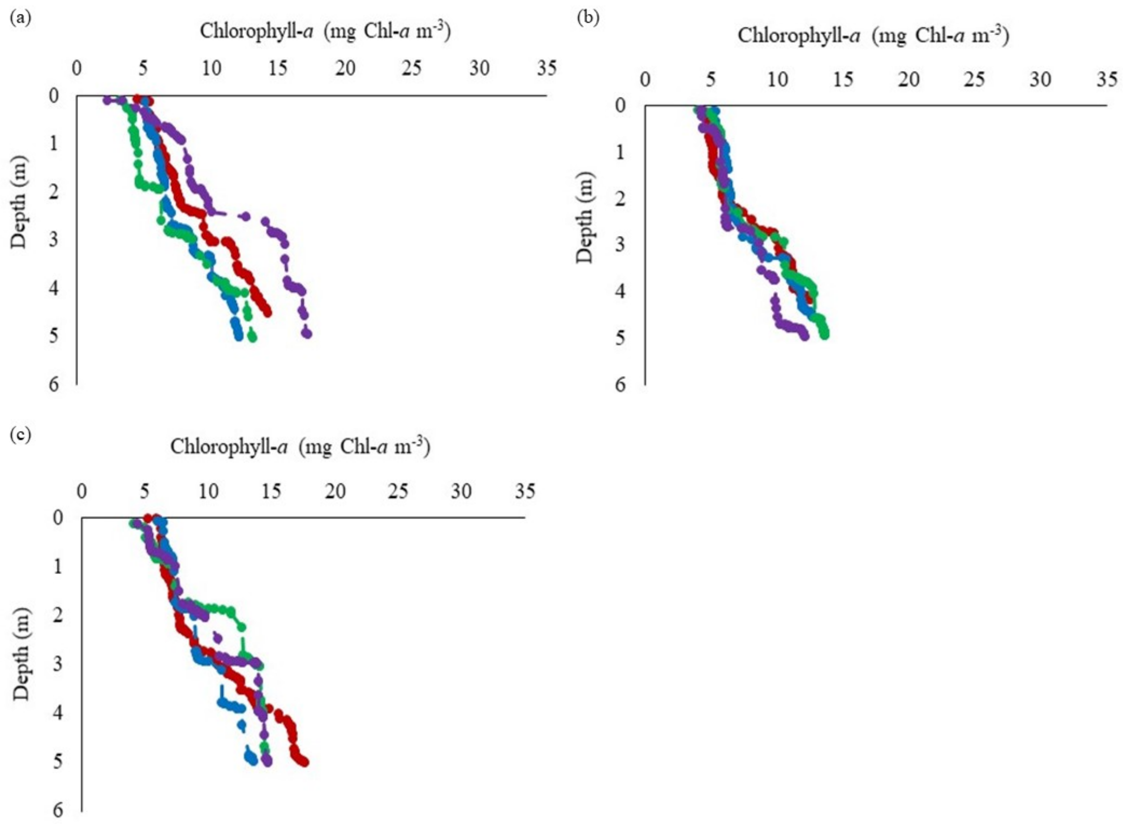

Figure 5. Depth profiles of chlorophyll-a in Swansea Bay. Stations 1, 6, 9, 11 are shown in 5a; stations 2, 3, 5, 7 are shown in 5b and stations 4, 8, 10, 12 are shown in 5c. These are for survey 1 only. Profiles for survey 2 are not shown as the trends were similar to those for survey 1.

3.2. Model predictions under the no lagoon, ebb-only and two-way operational mode scenarios

Table 4 shows the water volume in the lagoon region without the lagoon and with the lagoon under the ebb-only and the two-way operational mode scenarios based on the 0-D modelling. The mean water volume within the lagoon region was found to be higher with the lagoon under the operational mode scenarios compared to that without the lagoon. However, the volume for the two-way operational mode was more comparable to that for the no lagoon scenario (Table 4). As mentioned in the Section 2.3, the mean water volume was used in the risk assessment model in order to calculate the DIN, DIP and phytoplankton chlorophyll biomass concentrations under the no lagoon scenario as well as the ebb-only and two-way operational modes.

Table 4. Predicted water volume within the lagoon region for the no lagoon, ebb-only and two-way operational mode scenarios.

	No lagoon	Ebb-only	Two-way
Minimum annual lagoon volume ($\times 10^6$ m^3)	31.7	39.1	35.9
Maximum annual lagoon volume ($\times 10^6$ m^3)	126.0	119.0	111.0
Mean annual lagoon volume ($\times 10^6$ m^3)	68.4	78.9	70.7

The DIN and DIP enrichment predicted by the risk assessment model under the no lagoon scenario is shown in Table 5. The reliability of the model predictions was assessed by comparing the predictions under the no lagoon scenario with measured concentrations in the bay. The predicted maximum phytoplankton chlorophyll biomass concentration (i.e. 14.6 mg Chl-a m^{-3} (DIN); 12.6mg Chl-a m^{-3}(DIP)) were within the range measured in the bay (i.e. 2.4 – 14.3 mg Chl-a m^{-3}). However, the predicted DIN concentration (i.e. 12.0 µM) was lower than those measured in the bay (i.e. 13.4 – 20.2 uM). As for the predicted DIP concentration (i.e. 0.6 uM), it was within the range previously reported in the bay (0.1 – 0.7 µM, Humphreys et al., 1980). The predicted maximum primary production rates were 0.43 gC m^{-2} day^{-1} for DIN and 0.37 gC m^{-2} day^{-1} for DIP (Table 6). The predicted maximum rates were comparable to those calculated on the basis of the difference in chlorophyll concentrations measured in surface waters in the two surveys (i.e. 0.13 – 0.36 gC m^{-2} day^{-1} assuming a carbon: chlorophyll ratio of 40) and those previously reported in the bay (0.34 – 0.41 gC m^{-2} day^{-1}, Joint, 1980). The comparison demonstrates that the model performed reasonably well in predicting DIN, DIP, phytoplankton chlorophyll biomass concentrations and primary production rate in the bay. The model has also performed well in other estuarine environments, further validating the model predictions (Tett et al., 2003; Painting et al., 2007; Zhang et al., 2019).

Nutrients and phytoplankton biomass are key indicators used to assess the eutrophic status of a waterbody. Under the Oslo Paris Convention for the Protection of the Marine Environment of the North-East Atlantic (also known as OSPAR Convention) strategy to combat eutrophication, water columns with DIN concentrations > 15 µM and DIP concentrations > 0.8 µM are considered eutrophic (OSPAR, 2003). Painting et al. (2007) based on Nixon et al

(1995) recommended DIN, DIP and phytoplankton chlorophyll biomass thresholds for trophic status in UK estuarine and coastal waters (Table 5). The predicted DIN, DIP and phytoplankton chlorophyll biomass concentrations in the water column were lower than the eutrophic thresholds defined by OSPAR (2003) as well as Painting et al. (2007) but were within the thresholds for mesotrophic status (Table 5). This indicates that the bay cannot be considered eutrophic under the no lagoon scenario, despite been nutrient enriched. However, it is mesotrophic and this moderate response to nutrient enrichment in the bay is further supported by the lack of oxygen depletion evidenced by the measured DO concentrations in the bay > $6 mgl^{-1}$ (Table 5). That said, the rate of light controlled phytoplankton growth under the no lagoon scenario was predicted to be greater than the sum of the exchange rate of water and the rate of loss of phytoplankton (Table 6). This demonstrates the favourable conditions for phytoplankton growth in the bay, which is unsurprising given the turbidity levels and water residence time in the bay.

Under the ebb-only and two-way operational mode scenarios, the predicted DIN, DIP and phytoplankton chlorophyll biomass concentrations were consistent with those under the no lagoon scenario and these concentrations did not exceed the threshold values for eutrophic status but were within the mesotrophic threshold (Table 5). These findings imply that there is unlikely to be a risk of eutrophication in the impounded water column within the lagoon following its operation under ebb-only and two-way modes, with no change to the moderate response to nutrient enrichment predicted under the no lagoon scenario. While the rate of light controlled phytoplankton growth was significantly greater than the sum of the water exchange rate and the rate of loss of phytoplankton under the ebb-only scenario indicating favourable conditions for phytoplankton growth, the same cannot be said for the two-way scenario (Table 6). This suggests that the conditions under the ebb-only scenario are more favourable for growth in comparison to those under the two-way scenario. This reflects the higher current velocities within the lagoon due to advective accelerations and turbulent wakes under the two-way operational mode compared to the ebb-only (Angeloudis et al., 2016). High velocity currents enhance sediment resuspension which in turn increases turbidity and reduces sunlight availability within the lagoon, creating less favourable conditions for phytoplankton growth and suppressed primary production rates over time.

Table 5 Summary of predicted DIN, DIP and maximum phytoplankton chlorophyll biomass concentrations for the no lagoon, ebb-only and two-way operational mode scenarios. The predictions were compared with threshold values for trophic status in UK estuarine and coastal waters (Painting et al., 2007).

	DIN (uM)		DIP (uM)		Phytoplankton chlorophyll biomass (mg Chl m^{-3})	
	Predictions	Trophic status	Predictions	Trophic status	Predictions	Trophic status
No lagoon	12.0	Mesotrophic (<37 & ≥12)	0.6	Mesotrophic (<1 & ≥0.5)	14.6 (N); 12.6 (P)	Mesotrophic (<40 & ≥13)
		Eutrophic (<61 & ≥37)		Eutrophic (<2 and ≥1)		Eutrophic (<67 and ≥40)
Ebb-only	12.0	Mesotrophic (<37 & ≥12)	0.6	Mesotrophic (<1 & ≥0.5)	14.6 (N); 12.6 (P)	Mesotrophic (<40 & ≥13)
		Eutrophic (<61 & ≥37)		Eutrophic (<2 & ≥1)		Eutrophic (<67 and ≥40)
Two-way	12.1	Mesotrophic (<37 & ≥12)	0.6	Mesotrophic (<1 & ≥0.5)	14.7 (N); 12.6 (P)	Mesotrophic (<40 & ≥13)
		Eutrophic (<61 & ≥37)		Eutrophic (<2 & ≥1)		Eutrophic (<67 & ≥40)

Table 6. Summary of predicted light-controlled phytoplankton growth rate, maximum primary production rate, water exchange rate and loss due to grazing for the no lagoon, ebb-only and two-way operational mode scenarios.

	No lagoon	Ebb-only	Two-way
Relative light-controlled growth rate μ (d^{-1})	0.12	0.12	0.10
Exchange rate + loss due to grazing filter feeders	0.01	0.02	0.11
Primary production (g C m^{-2} y^{-1})	0.43 (N); 0.37 (P)	0.41 (N); 0.36 (P)	0.36 (N); 0.31 (P)

4. Conclusion

In this study, an assessment of the existing physical estuarine characteristics in Swansea Bay has shown that the bay is largely well-mixed and vertically homogenous with little evidence of stratification. The turbidity levels were found to be low in the surface waters and with depth, suggesting sunlight penetration through the water column. The high DIN concentrations observed were indicative of nutrient enrichment. The phytoplankton biomass, indicated by the chlorophyll-a concentrations, was found to be high and increased with depth due to the nutrient enrichment and sunlight availability in the bay, coupled with the water residence time in the bay. However, there were no signs of eutrophication as the phytoplankton biomass did not reach the level typical of harmful algal blooms and the bay was well oxygenated with no evidence of oxygen depletion. As a result, eutrophication is not believed to be present in the bay. In addition, the predicted DIN, DIP and phytoplankton chlorophyll biomass concentrations under the no lagoon scenario suggests that the bay is mesotrophic and thus, exhibits a moderate response to nutrient enrichment. Following the operation of a tidal lagoon under both the ebb-only and two-way operational modes, there is likely to be no net change in the status of the bay, with no risk of eutrophication. However, the conditions for phytoplankton growth are likely to be more favourable under ebb-only operational mode compared to two-way operational mode due to changes in the current velocities in the enclosed basin within the lagoon.

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