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Waterbodies Thermal Energy Based Systems Interactions with Marine Environment - A Review

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Waterbodies Thermal Energy Based Systems Interactions with Marine Environment - A Review

Abstract. Waterbodies' thermal energy potential, as a green, renewable, and limitless source of energy, can be exploited in response to the growing energy demands of islands and coastal cities. Up to now, the technologies that have been developed for this purpose include seawater air-conditioning, surface water heat pump, and ocean energy thermal conversion systems or their combinations, which are presented here as Waterbodies Thermal Energy Based Systems (WTEBSs). The growth and development of these technologies raise concerns regarding their potential impacts on sustainability of the marine environment. The present work provides a comprehensive review of the available literature and state-of-the-art technologies describing potential interactions of WTEBSs throughout their life-cycle (i.e. including construction, installation, operation, and decommissioning) with the marine ecology. Modelling of seawater discharge dispersion as one of the main environmental impact concerns regarding the operation of WTEBSs is detailed and scopes for improving existing modelling tools are discussed. Potential destructive impacts of fouling and corrosion in WTEBSs are reported and deterrent recommendations are highlighted. Evidence of growth of bio-fouling inside of pipelines and associated mesh filtration baskets at abstraction pipe intakes are presented. The required permitting applications and licensing processes for installation and operation of WTEBSs by the relevant authorities are summarised. Finally, a summary of the findings from the data monitoring of water quality properties of a seawater air-conditioning pilot study performed at Brixham Laboratory, University of Plymouth, United Kingdom is reported.

Keywords: Renewable Energy; Marine Thermal Energy; Environment Impact; Dispersion; Energy resources; Biofouling; Corrosion; Seawater Air Conditioning; Ocean Thermal Energy Conversion; Surface Water Heat Pump

1 Introduction

Anthropogenic global warming is a direct consequence of activities such as burning of fossil fuels (coal, oil and gas), which causes large emission of greenhouse gases (GHG) into the atmosphere (Houghton, 2005, Meckling, 2018). Renewable energy technologies that exploit energy from sources such as solar, wind, wave, and ocean thermal energies were developed to address the environment challenges from impacts of carbon-based fuel GHG (Ullah et al., 2017, Comfort et al., 2015). The uncertainty of increasing oil prices and recent advances in efficiency of renewable energy technologies coupled with increased installation capacities have accelerated development of competitiveness of renewable energy alternatives in the global energy market (Gohar Ali et al., 2020, Arent et al., 2011). Figure 1 illustrates the growth of renewable technologies share on global electricity production over the last 35 years.

The thermal capacity of waterbodies (e.g., lakes, seas, and oceans) is by comparison an unlimited intact heat sink or source, that can help meet the high energy demands of coastal regions and islands. To illustrate the significant thermal capacity potential of waterbodies, Hunt et al. (2019) provided a comparison between energy potential in seawater with other renewable electricity generation sources for cooling purposes. Their findings highlighted the energy potential of 1 m^3/s of seawater for cooling with 10 °C temperature gradient is equivalent to either a hydropower plant with a generation head of 186 m with ten times the flow rate, a 488,000 m^2 solar power plant, or typical energy generation of 21 wind turbines.

Waterbodies Thermal Energy Based Systems (WTEBSs) harness the thermal energy of oceans, or seas, and their performance rely on the temperature of extracted seawater from waterbodies. In each waterbody region, the local water temperature is a function of water depth: surface water and deep water. In case of ocean water, surface water is warm water that extends to depths of a few hundred meters; beneath that is deep ocean water which is cold, dense, and nutrient-rich (Hunt et al., 2021, Herrera et al., 2021). Due to the latter being higher density than the former, both layers do not mix, and a transition layer called thermocline exist in between i.e., in depth of between 400 m to 1,000 m (Hunt et al., 2021, Hunt et al., 2020). Figure 2 illustrates density and temperature variation profiles with depth where the temperature variation in different latitudes of open oceans for tropical, equatorial, and middle latitudes are different. Likewise, for waterbodies regions that are separated from the deep ocean such as the Mediterranean Sea, the Sulu, Visayan, and Bohol Seas in Southwestern Philippines, the temperature profiles are distinctively different from the ones in Figure 2 (Schroeder et al., 2008, Ferrera et al., 2017). In the case of lakes, the water temperature below a depth of approximately 18 to 24 m may remain relatively constant throughout the year (Hattemer and Kavanaugh, 2005). However, this depends highly on the amount of inflow and/or outflow relative to the surface waterbody size (Mitchell and Spitler, 2013).

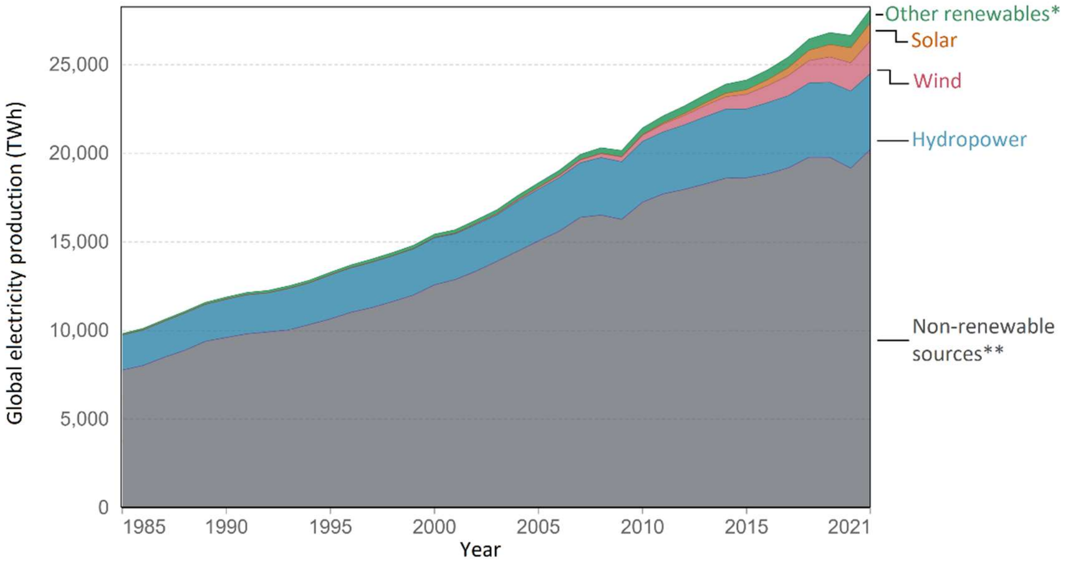

Figure 1: Global electricity production by Ritchie and Roser (2021). *'Other renewables include biomass and waste, geothermal, wave, and tidal, **'Non-renewable sources include coal, gas, oil, and nuclear.

Presently, WTEBSs is generally classed into three main categories (shown in figure 3):

- Seawater Air Conditioning (SWAC): systems that exploit water from waterbodies for heating or cooling demands using heat exchangers without heat pumps or chillers (Mitchell and Spitler, 2013). SWAC systems are onshore-based plants with intake and discharge pipelines of adequate lengths that are shore-crossing and deployed at the bottom of the seas or oceans. SWAC replaces the heaters and chillers used in conventional air-conditioning (CAC) systems. This technology aims to greatly reduce the electricity consumption cost which in ideal condition can be around 80% lower than that for CAC (Hunt et al., 2020, Makai Ocean Engineering Inc., 2015a, Soini et al., 2017). SWAC systems can be categorised into shallow and deep seawater systems according to the depth at which seawater is extracted (Hunt et al., 2019). A comprehensive list of globally deployed SWAC systems can be found in Hunt et al. (2019).
- Surface water heat pump (SWHP): systems that benefit from heat pumps or chillers to provide heating or cooling; These systems benefit from surface water as a heat source or sink (Mitchell and Spitler, 2013). Under circumstances where the direct usage of seawater cannot meet the required cooling or heating demands, SWHP can be introduced as a justified alternative. SWHP systems are onshore-based plants with an average coefficient of performance (COP) of around 4 (Mitchell and Spitler, 2013). These systems have higher efficiency compared with CAC and air source heat pump (ASHP) systems that use ambient air as a heat source/sink with an average COP of around 3 (Su et al., 2020). With the rise in energy carriers' costs, SWHP has a great potential for operational cost savings (Chua et al., 2010). A non-exhaustive list of SWHP around the world can be found in Su et al. (2020).
- Ocean Thermal Energy Conversion (OTEC): systems that generate electricity from the natural thermal gradient between warm surface and cold deep ocean waters (Pelc and Fujita, 2002). The efficiency of OTEC significantly depends on the ocean thermal gradient. Equatorial latitudes are ideal regions for OTEC systems as they provide the maximum temperature difference between surface and deep ocean water, shown in Figure 2. OTECs typically have high implementation costs and low actual efficiency of around 3% or 4%, but they are an attractive renewable energy technologies as they benefit from an unlimited source of energy (Herrera et al., 2021).

OTEC systems can either be built onshore or offshore on floating platforms (Pelc and Fujita, 2002). In the case of floating platforms, the energy can either be transported via seafloor cables or stored in the form of chemical energy (e.g. hydrogen, ammonia, or methanol) that are regularly transferred to the shore by tankers (Pelc and Fujita, 2002, Avery and Wu, 1994). Currently, there is a limited number of OTEC plants that operate worldwide, which are either mostly small-scale or pilot systems (Herrera et al., 2021, Kim and Kim, 2020).

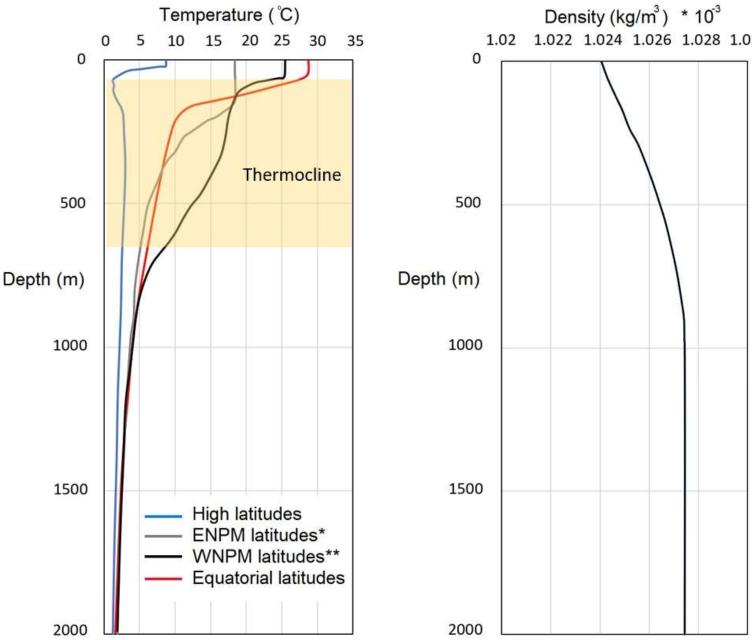

Figure 2: Typical temperature and density variations with water depth in the open ocean (Hunt et al., 2019, Talley, 2011). *ENPM is the abbreviation for Eastern North Pacific Middle, **WNPM is the abbreviation for Western North Pacific Middle.

To maximise energy utilisation efficiencies, WTEBSs can be combined. A hybrid SWHP and SWAC system presents a robust configuration that will be able to switch between two modes. The system works as a heat pump to provide cooling and heating, in case it is designed as a reversible heat pump. When the water temperature allows for it, the system can switch to the SWAC mode and utilise cool water directly for cooling purposes (Mitchell and Spitler, 2013, Ciani, 1978). Such a system has been successfully deployed in different cities around the world (Smebye et al., 2011, War, 2011). WTEBSs can also be combined with other technologies; for example, warmer seawater outlet of SWAC systems, which can be rich in nutrients, can be used for production of algae, fish, and crustaceans (Hunt et al., 2020, Von Herzen et al., 2017). In an open cycle OTEC system, which refers to systems that uses seawater as the working fluid, the desalinated water (condensate) is fresh enough for municipal or agricultural use, and the cold nutrient water can be applied to aquaculture (Pelc and Fujita, 2002, Avery and Wu, 1994). Hunt et al. (2021) proposed a combination of SWAC and reverse osmosis (RO) desalination to supply both affordable water and cooling services in a one-way district cooling system that provide several advantages compared to SWAC and RO individually, while reducing distribution costs. A combined system that employs an offshore wind-driven hydraulic pump to supply high-pressure deep seawater to a land base cooling plant (SWAC or SWHP) is proposed in many studies such as Sant and Farrugia (2013), Sant et al. (2014), Galea and Sant (2016a) , Galea and Sant (2016b), (Buhagiar and Sant, 2014). With the growth of marine renewable energy technologies, concerns regarding their impacts on the sustainability of marine environments have been raised (Comfort et al., 2015, Pelc and Fujita, 2002, Comfort and Vega, 2011, Boehlert and Gill, 2010, Gill, 2005). To address these concerns, it is critical to investigate the environmental footprint of existing WTEBSs in the effort to minimize the impacts for future applications.

The next section details the environmental impacts of WTEBSs throughout their life-cycle. In section 3, investigation on modelling of discharge dispersion as one of the main concerns regarding operation of the WTEBSs is discussed. This is followed by an investigation of the effects of biofouling and corrosion during optimal operation of WTEBSs and measures to control them in section 4. The subsequent section summarises the required permitting applications and licensing processes for installation and operation of WTEBS by the relevant authorities. In section 6, the results of an environmental impact assessment study to measure the water quality parameters near the discharge area of the pilot shallow water SWAC system at Brixham laboratory at the University of Plymouth, United Kingdom are reported. Finally, in the conclusion section, the findings of this study are summarised.

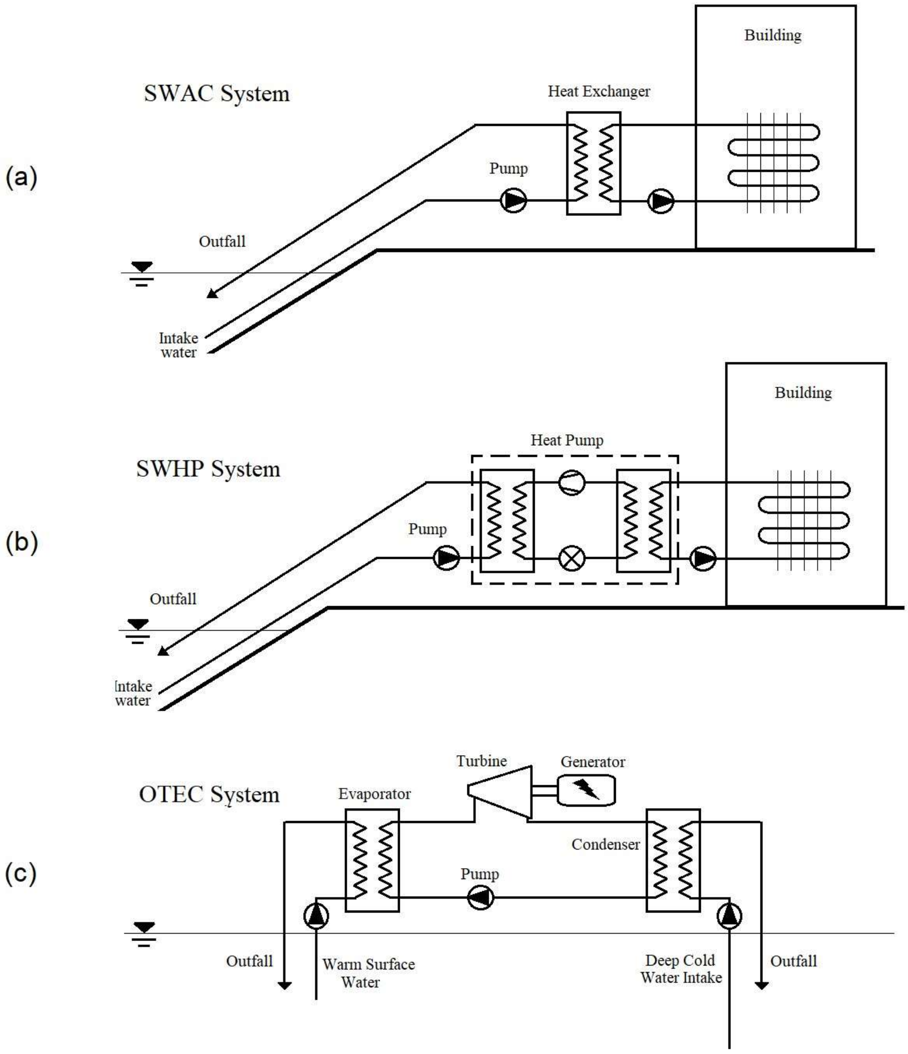

Figure 3: Simplified schematic of different types of onshore- and offshore-based WTEBSs, (a) SWAC, (b) SWHP, (c) OTEC.

2 WTEBS Environmental Impacts

Anthropogenic activities are the main reasons for major changes to marine wildlife (Gill, 2005, Ferreira et al., 2018). Terrestrial land uses and near-shore activities such as dredging, overfishing, oil and gas operations, illegal dumping of solid wastes, and other industrial processes have dramatically implicated perturbation of the marine environment (Gill, 2005, Ferreira et al., 2018, Carpenter, 2019, Barletta et al., 2016, Lima et al., 2016, Blaber et al., 2000, McLusky et al., 1992, Costa and Barletta, 2015). Recently, Halpern et al. (2019) investigated the cumulative impact of 14 stressors related to human activities at 21 different marine ecosystems globally during a 11-year period from 2003–2013. As a result, they realised that most of the ocean (59%) is experiencing increasing cumulative impact due to climate change but also from fishing, land-based pollution, and shipping. The growth in the deployment of offshore renewable energy technologies also add to the risks from their interactions with the marine environment. A list of newly emerging renewable energy technologies with a special concentration on marine energy generation is found in Wilberforce et al. (2019) and Chen et al. (2018). The life-cycle (i.e. including construction, installation, operation, decommissioning) environmental impact assessment of tidal and wave energy generation devices are reviewed and evaluated in Frid et al. (2012), Patrizi et al. (2019), Baker et al. (2020), Copping et al. (2021), Isaksson et al. (2020), Sayed et al. (2021), and Farr et al. (2021). Williamson et al. (2019) used ecological and physical measurements to show the predictability of fish school characteristics (presence, school area, and height above seabed) at a high-energy tidal site, and how this changes in the proximity of a turbine structure. Similarly, Malinka et al. (2018) studied the behaviour and movement of small cetaceans around a tidal turbine. Others, such as Seyfried et al. (2019) reviewed the potential environmental impacts of a salinity gradient energy (SGE) facility through the construction, operation, and decommission phases. The life-cycle environmental impact assessment of offshore wind turbines has been investigated in Sayed et al. (2021), Shadman et al. (2021), and Hall et al. (2020). Gill et al. (2020) studied offshore wind development effects on fish and fisheries. Tougaard et al. (2020) and Madsen et al. (2006) reviewed available measurements of underwater noise from different wind turbines during operation and reported that the underwater noise radiated from individual wind turbines is low compared to noise radiated from cargo ships. The combined noise level of a large wind farm can cause negative effects on species of fish and marine mammals. Boehlert and Gill (2010) noted that devices with subsurface moving parts, such as underwater turbines, are assumed to be the noisiest. An investigation on the underwater operational sound of a tidal stream turbine can be found in Risch et al. (2020). The potential impacts of submarine power cables during the installation, operation, and decommissioning phases on the marine environment have been studied in Taormina et al. (2018), Hutchison et al. (2020), and Scott et al. (2018).

In this section, a review of the relevant concerns and interactions of the development of WTEBSs, including different stages of construction, operation, and decommissioning, with the marine environment is provided in detail. Many of the associated effects of WTEBSs are common with other types of development in the marine environment which facilitate the impact assessments process, but potential uncertainties may arise when their impacts have not been evaluated or anticipated accurately.

2.1 Construction and Decommissioning Impacts

The construction and decommissioning phases of the development of a WTEBS are likely to cause significant positive and negative disturbances to local environmental resources and fundamental changes to the habitat, both above and below the water surface (Boehlert and Gill, 2010, Cardno Tec. Inc., 2014). Their spatial scale may have ecological impacts extending over several square kilometres, while temporal scales are both short- and long-term on marine environments (Gill, 2005, Iglesias et al., 2018). The magnitude of the impacts highly depends on the duration and intensity of the disturbance and the stability and resilience of the marine communities (Gill, 2005, Van Dalfsen et al., 2000, Lu et al., 2020, Drabsch et al., 2001). The ecological implications associated with WTEBS construction can be similar to the alterations of the benthic habitats that had been subjected to fishing or marine dredging (Gill, 2005, Hiddink et al., 2020, Blyth et al., 2004). In general, during construction, the seabed will be disturbed by installation of foundations and hard-fixed structures (such as submerged heat exchangers or pump stations), pipelines, scour-protection systems, mooring devices, and seabed-mounted power cables. Marine organisms within the footprint of these objects would be smothered or crushed (Cardno Tec. Inc., 2014). These artificial structures may have the greatest impact on benthic habitats and ecosystems (Boehlert and Gill, 2010). They also may alter the local flow which is essential to some aquatic species such as corals (Hennige et al., 2021, Georgoulas et al., 2023), lead to entrainment and deposition of sediments, and change the seabed bathymetry (Montgomery et al., 2006). Conversely, the deployment of these objects on the seabed, provides artificial reefs in benthic environments (Addis et al., 2006, Inger et al., 2009). This may stimulate the benthic ecosystem and lead to a greater biodiversity (Inger et al., 2009, Langlois et al., 2005). The construction phase may also disturb the surface and midwater with structures including spars, buoys, pipelines, and cables that may result in modifications on pelagic habitats and ecosystems (Boehlert and Gill, 2010, Langhamer et al., 2009). These effects are widely studied in the oil and gas platform industry where these structures can serve an equivalent function to artificial reefs in benthic environments (Addis et al., 2006, Inger et al., 2009). The presence of these objects may have positive effects on attraction of some species (e.g., krill, mysids, and fishes) and consequently additional predators in the region. The presence of the structures may modify the local water hydrodynamics which take up significant areas of the sea surface which could influence migratory surface dwellers (Boehlert and Gill, 2010).

In the rest of this section, the interaction of WTEBS construction with the marine environment is detailed separately for offshore and onshore systems, followed by the potential decommissioning impact of these systems.

2.1.1 Onshore WTEBS construction

The construction of seawater pipeline systems for onshore WTEBS presents the main interaction with the ocean environment. The pipelines are mounted on the seabed, up to a few kilometres long, to reach cold deep seawater. These systems may contain submersible pumps or submerged-coils (heat exchangers) in seawater/lake heat pump systems (Wu et al., 2020, Liu et al., 2019, Sarbu and Sebarchievici, 2014, Zheng et al., 2015). The pipelines are mainly made of high-density polyethylene (HDPE) material due advantages it offers, such as strength, durability, flexibility, insulation, resistance to high pressure, cost-effectiveness, and slight negative buoyancy, compared to alternative materials (Hunt et al., 2020, Nguyen et al., 2021, Miller et al., 2012). For pipelines that are exposed to storms, tsunamis, seismic activities, and other environmental concerns, the most challenging aspect of the development of the pipelines is at the coastal transition zone (sea/shore interface) aspect (War, 2011). In most cases, to reduce the risk of damage or incident, the pipelines are either trenched or tunnelled under the shoreline, from a point before the shoreline to a point in the seabed, a few meters deep (War, 2011, Cardno Tec. Inc., 2014, Lewis et al., 1989). Among these two techniques, tunnelling such as microtunnelling and horizontal directional drilling (HDD) is preferable as they are more environmentally friendly (War, 2011, Cardno Tec. Inc., 2014, Camp et al., 2019, da Silva et al., 2013, Swartz, 2020), while trenching comes with the removal of sediments and direct loss of marine habitats (Gill, 2005). The latter also increases the local water turbidity level as a result of suspended particles. This may increase the risk of spreading any contaminants from the suspended particles, and lead to a temporary reduction of the available oxygen which may smother the neighbouring habitats of sedentary species (Gill, 2005). In general, the benefits of trench-less technologies compared to open-cut trenching are their minimum impact on existing infrastructure, longer pipeline lifetime, minimum efforts to reinstate the site following pipe installation, and independence from weather conditions and waves during the construction phase (Hennig and zur Linde, 2011). Nevertheless, trench-less technologies do risk potential leakages of drilling mud through the sediment into the water column during micro-tunnelling. This can be eliminated by grouting the void between the micro-tunnel and the pipes (Cardno Tec. Inc., 2014). In the development of seawater pipelines using trench-less technologies, pipeline construction impacts would be mainly related to the excavation of a breaking point, where sediments would be removed, and bathymetry temporarily changed at the pit (Cardno Tec. Inc., 2014). The breakout point (receiving pit) is where buried pipes and seabed surface-mounted pipes are connected (Figure 4). The temporary impacts of the constructions such as elevated levels of suspended sediments in water adjacent to the excavation area can be minimized by the installation of sheet piles around the pit to isolate it from the surrounding water (Cardno Tec. Inc., 2014). The long-term impacts on ocean currents are negligible as the breakout pit would be back-filled and caped with concrete similar to the original bathymetry. Apart from the coastal transition zone, the rest of the pipelines are mounted on the seabed surface which can be installed in a controlled submergence process. Detailed discussion regarding the installation of the HDPE deep seawater pipelines is found in War (2011), Cardno Tec. Inc. (2014) , Makai Ocean Engineering Inc. (2015c).

A possible long-term impact of the mounted pipelines would be associated with the scouring and sediment transportation beneath the pipes. This can be minimized with sufficient clearance between the pipes and the seabed (Cardno Tec. Inc., 2014). Some recent studies in numerical modelling of scouring can be found in the works of Bordbar et al. (2021), and Bordbar et al. (2022a), and Bordbar et al. (2022b). Nevertheless, close monitoring will be essential, and whenever required a scour counter-measure method has to be considered (Elahee and Jugoo, 2013).

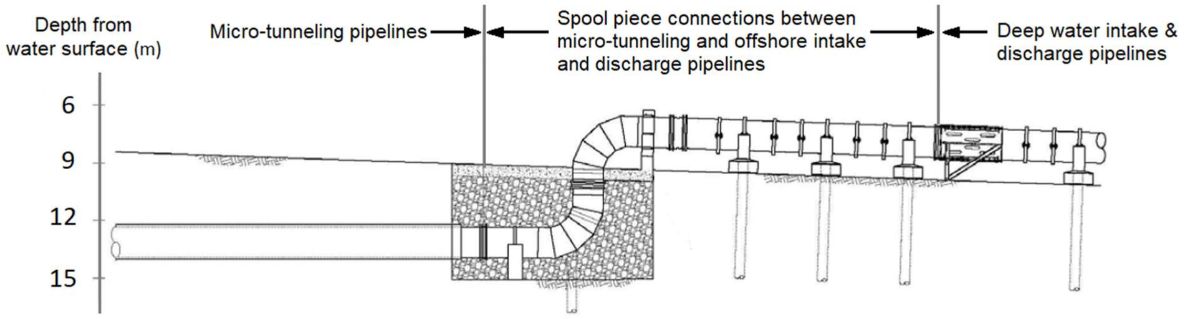

Figure 4: Details of breakout point (receiving pit) where buried pipes and seabed surface-mounted pipes are connected (Cardno Tec. Inc., 2014).

To minimize the environmental impact of pipeline installation for a future onshore WTEBS application (e.g. a SWAC system), DeProfundis and DORIS Engineering have introduced an innovative intake pipe self-burying system. The system limits the impact on the underwater environment and reduces installation costs (Doris engineering, 2022). Simply put, the system includes injecting water into the sand located under the pipe placed on the ground to thin the sand so that the pipe sinks in under its own weight. The method benefits from a new concept called "flexible pipe" which will contribute to the cost reduction of conventional onshore WTEBSs by reducing material and installation costs (Oc´eanide, 2022).

2.1.2 Offshore WTEBS construction

Offshore WTEBS, i.e. including platforms, intakes and outfall pipelines, and mooring systems, can affect both benthic and pelagic ecosystems. The main environmental impact in pelagic zones during the installation of the system is likely to be related to the seismic surveys at the start of the project, shipping movements, construction noise, and potential chemical pollution associated with marine vessel operations. Brandt et al. (2009) reported that marine mammals temporarily avoid an area where construction is underway. The effect disappears immediately after the cessation of noisy activities. If no anti-fouling is used, the presence of the offshore WTEBS structures will provide settlement habitats for a variety of organisms (Itano and Holland, 2000, Dempster and Taquet, 2004). As discussed earlier in Section 2.1, for a large-scale platform the potential impacts of the local water flow modification and the large area of the occupied ocean surface on migratory surface dwellers and pelagic ecosystem need to be considered.

McHale (1979) reported the development process of a cold-water pipeline associated with a 50 *KW* mini-OTEC plant at Kona, Hawaii. In OTEC systems, cold-water pipelines may serve as a combined cold-water pipe and mooring line (McHale, 1979, Zhang et al., 2018, Magesh, 2010). The impact on the benthic zone is likely related to the installation of mooring systems and power cables. The installation of these devices may locally disturb the ecosystem and temporary increase the turbidity of the water, however, biota density is limited in that depth, i.e., infra to 1000 *m* depth (Devault and Péné-Annette, 2017).

Water pipelines of floating WTEBS can be made of HDPE, or Fiber-Reinforced Plastic (FRP). For large-scale OTEC floating plants with 4 to 10 *m* diameter intake pipelines, FRP material is often employed as the use of HDPE is not available for pipelines with diameters larger than 2.5 *m* (Stoev et al., 2018). HDPE is not a biodegradable material and at the end of its life, it should be responsibly recycled, whereas FRP pipe material is non-corrosive (Vahidi et al., 2016, Sözen et al., 2022).

2.1.3 WTEBS decommissioning

The associated environmental impacts of decommissioning for a site are often assumed to be similar to those when the site is constructed (Boehlert and Gill, 2010, Gill, 2005). The removal of existing underwater structures will cause sudden alterations in the heterogeneity of the benthic inhabitant by removing a component of the ecosystem (Kaiser and Jennings, 2002). This may disturb the local food web and also changes habitat availability (Gill, 2005).

2.2 Operation Impacts

A WTEBS intakes/discharges large volumes of ocean water. For example, an OTEC plant typically needs around 5 m^3/s of cold deep seawater, and an equal intake of warm surface water per 1 MW capacity; therefore a commercial OTEC system with 100 MW capacity needs a massive volume of 500 m^3/s of cold and warm intake water for operation (Avery and Wu, 1994). The system mixes the water and discharges it into the ambient environment with different characteristics. Considering the lifetime of a plant (25-30 years), the operation of WTEBS may change the water characteristics in near-, intermediate- and far-fields, and consequently significantly affects marine ecosystem (Pelc and Fujita, 2002). Furthermore, concentrated deployment of large-scale WTEBSs can accumulate and intensify the impacts (Comfort and Vega, 2011). While environmental impacts associated with processing seawater are the main focus, impacts from other factors such as power cable electromagnetic fields, acoustic effects of the WTEBS machinery and pipelines, and leakage of chemicals from the system will be of importance during the operation of the system. In the pertinent literature, most of the knowledge regarding the environmental impacts of WTEBS comes from studies that investigated the pre-impact condition at a future

WTEBS site. Among them, Comfort et al. (2015), Cardno Tec. Inc. (2014), Ciani (1978), and Comfort and Vega (2011) studied the coastal area of Hawaii for SWAC and OTEC projects.

A detailed review of the features of the marine environment that may change with the operation of a WTEBS and the potential impacts of these changes on marine life are presented in this section. Figure 5 illustrates part of these impacts for an offshore and onshore WTEBS.

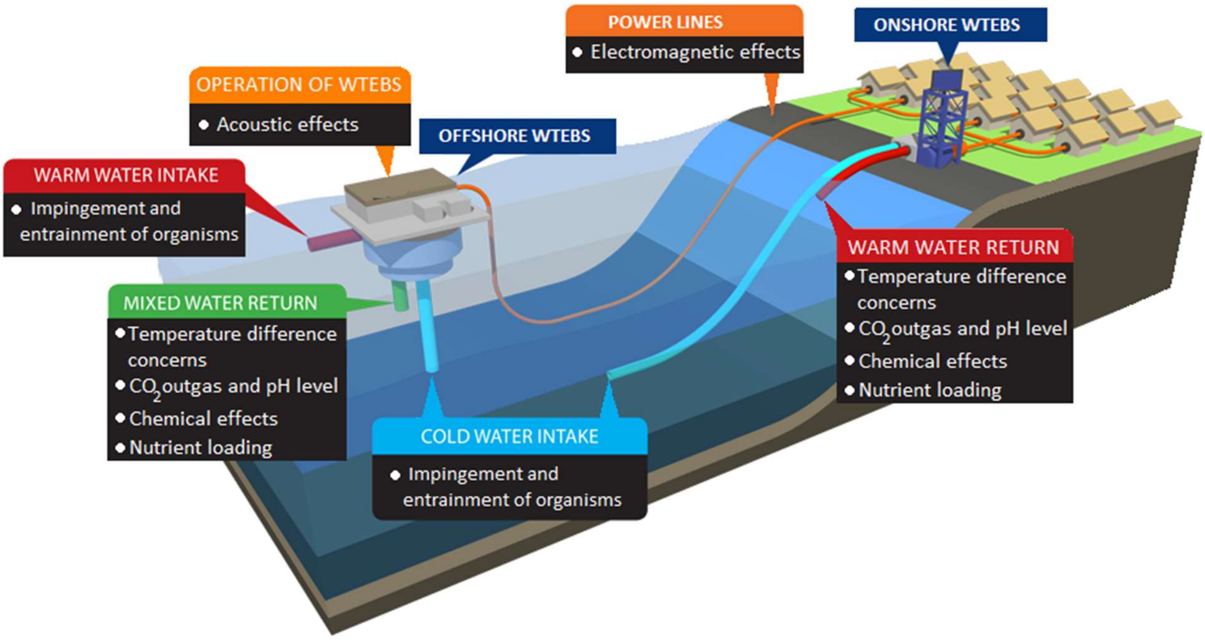

Figure 5: Operational impacts of offshore and onshore WTEBSs in marine environment.

2.2.1 Impingement and entrainment of organisms

The inlet pipelines may intake marine organisms, especially those with low mobility and are smaller than the mesh of the inlet pipe screen, into the system during operation (Avery and Wu, 1994). These organisms will be impinged to the internal walls of the system and will encounter rapid environmental changes, such as temperature, dissolved oxygen, turbidity, and light levels of the water, which to a great extent reduce their chance of survival (Avery and Wu, 1994, Elahee and Jugoo, 2013, Cunningham et al., 2010). This phenomenon has been studied in coastal nuclear power plants and is similar related for WTEBSs (Avery and Wu, 1994, Barnthouse et al., 2019, Chae et al., 2008). Due to higher concentration of marine life in shallow waters, this is an important factor in systems that intake surface water and needs to be assessed for systems that intake cold deep seawater depending on existing ecology (Comfort and Vega, 2011, Elahee and Jugoo, 2013, Myers et al., 1986, Deevey and Brooks, 1971). Here, the intake pipelines are designed to preserve and maintain a low approach velocity to minimize the risk of marine organisms being sucked into the system (Cunningham et al., 2010). Nonetheless, plankton, small nekton, and most tuna larvae are often at risk of entrainment into surface water intakes (Myers et al., 1986, Boehlert and Mundy, 1994). This risk decreases significantly for larger organisms due to their swimming capabilities (Comfort and Vega, 2011). In addition, pipeline vibrations during system operation may generate signals for marine mammals and fishes to avoid approaching the pipelines (Comfort and Vega, 2011). The discharge outfall can also be an attractive destination for marine organisms as it may be rich in nutrients; this increases the probability of impingement and injury to marine organisms (Elahee and Jugoo, 2013). Using the discharged water for secondary purposes can influence the water discharge quality which needs further monitoring and observations.

2.2.2 Chemical effects

In systems with closed-cycle operations, the working fluid is normally ammonia, or R-134a, and is isolated from the water being abstracted. As ammonia is highly toxic to fish, and concerns growing regarding the impact of R-134a on marine life, many studies have focused on use of these chemicals (Emani et al., 2017, Zhao et al., 2020, Jung and Hwang, 2014). A study of a selection of working fluids in terms of toxicity, environmental performance, and flammability can be found in Jung and Hwang (2014). Leakage or spill of working fluid may endanger the local marine population if the working fluid concentration in water exceeds toxicity levels (e.g. the United State environmental protection agency considers the concentration of ammonia higher that 0.4 *mg/Litre (ppm)* toxic to fish (both freshwater and marine)). To give an indicator of dangerous amount of working fluid leakage, consider that a seawater flow rate through a 40 *MW* OTEC plant is around 2×10^7 m^3/day (Avery and Wu, 1994). To exceed the environmental protection agency's limit, an ammonia leakage of around over 8×10^3 *kg/day* into the seawater flow is required. This could only occur if there were serious malfunctions such as a major breakdown, a collision with an ocean-going vessel, an unpredicted climate condition, terrorism, or causes from human errors. It is to be noted that a workflow leakage due to a malfunction of the system should be avoided at all costs (Owens and Trimble, 1981). Apart from the working fluid, during normal operation, the potential of leakage from devices that use a hydraulic fluid needs to be considered along with the evaluation of toxicity impact of heavy metal concentrations from heat exchangers (Fast et al., 1990). Chemicals used for controlling bio-fouling and corrosion, such as chlorine or protective coating materials can accumulate in the tissues of organisms and be passed up in the food chain (Avery and Wu, 1994, Elahee and Jugoo, 2013). Pre-treatment before disposal of chemicals and/or mechanical control of fouling should be implemented (Elahee and Jugoo, 2013).

2.2.3 Nutrient loading

For WTEBSs that intake deep cold ocean water, the untreated plume will have different physical and chemical properties (e.g. temperature, density, salinity, dissolved gases, nutrient level, and pH level) than the surrounding ocean water where it is discharged (Comfort et al., 2015, Comfort and Vega, 2011, Boehlert and Gill, 2010). The density difference between the discharge outfall and the ambient water will cause the plume to sink or rise to an equilibrium depth and produce an artificial nutrient-enriched zone (Comfort and Vega, 2011). If the plume equilibrium occurs in the photic zone, it may induce phytoplankton and algal blooms and subsequently, affect changes in the pelagic food web ecosystem and habitat (Comfort et al., 2015, Boehlert and Gill, 2010, Devault and Péné-Annette, 2017, Harrison, 1987, Richardson and Schoeman, 2004, Lilley et al., 2012). In coastal areas, this may interrupt economical activities such as shore-based businesses, the fishing industry, and recreational tourism (Boehlert and Gill, 2010, Elahee and Jugoo, 2013). To minimise the environmental impact of WTEBS plumes, it is crucial to ensure that the nutrient-rich plume does not mix with surface waters and remains beneath the most biologically productive depths (below the 1% light level) (War, 2011, Comfort and Vega, 2011, Farr et al., 2021); different water depths between 90 to 200 *m* have been recommended for this purpose (Comfort and Vega, 2011, Farr et al., 2021, Lilley et al., 2012, Deprofundis, 2016). The recommended depth depends on detailed local conditions, environmental regulations, and diffuser dispersion modelling employed for each case (Makai Ocean Engineering Inc., 2015b). Nutrient enhancement for WTEBS that intake water from shallow sea or lakes also needs to be investigated, as many lakes and shallow sea areas show vertical stratification of water during warm seasons (Boehrer and Schultze, 2008, Hickman et al., 2012).

The rich-nutrient discharge of WTEBS can also serve secondary utilisation for energy production, cooling, desalination, aquaculture, and agriculture (War, 2011, Samuel et al., 2013, Elsafty and Saeid, 2009). Nevertheless, the environmental impact of the effluent from the secondary utilization system into the ocean needs to be assessed. A comprehensive review of experimental and numerical modelling of effluent dispersion is provided in Section 3.

2.2.4 Temperature concerns

If WTEBS water discharge is not returned to isothermal depths, there will be a risk of a slight change in water temperature. This thermal effect may have severe consequences on marine life, as thermal changes can lead to reductions in the hatching success of eggs, inhibition of larvae development, and increase in death among coral and fishes (Pelc and Fujita, 2002, Elahee and Jugoo, 2013, Lilley et al., 2015). However, Avery and Wu (1994) reported results from several theoretical and experimental studies (e.g. Adams et al. (1979)) concluded that climatic alterations due to operation of OTEC systems are negligible, or extremely localised. In fact, over the long term, the large volume of discharge plume has the potential to alter the marine ecosystem in regions near the discharge outlet (Boehlert and Gill, 2010, Harrison, 1987). The impacts in the far-field region can only be noticeable in the case of deployment of a very large number of OTEC plants (Avery and Wu, 1994).

2.2.5 CO_2 outgas and *pH* level

Seawater has many different gases dissolved in it, including nitrogen, oxygen, and carbon dioxide. The intake water into WTEBSs are subjected to changes in temperature and pressure which lead to changes in the solubility of dissolved gas. For systems that intake deep sea ocean water, it can result in dissolved CO_2 outgas (Elahee and Jugoo, 2013). While this amount will depend on the volume of water being pumped, Avery and Wu (1994) pointed out that such an amount would be smaller than emissions from a fossil-fuel-fired plant. Conversely, CO_2 and other carbon compounds (e.g. carbonate and bicarbonate) play an important role in the pH level of ocean water (Webb, 2021). Changes in the concentration of CO_2 levels in water may increase concerns regarding the acidification effect of the artificially upwelled water (Boehlert and Gill, 2010, Feely et al., 2008, Griffith et al., 2011). The change in the pH level of the seawater can disturb the marine ecosystem, biodiversity, and marine food web (Griffith et al., 2011).

2.2.6 Acoustic effects

Acoustics play an important role in underwater ecosystems and are essential in animal communication, reproduction, orientation, and prey and predator sensing (Boehlert and Gill, 2010). Anthropogenic underwater noise will likely add to the normal background acoustic environment (Boehlert and Gill, 2010). The possible impacts of artificial noise on fish, marine mammals, and crab and lobster larvae have been indicated in Montgomery et al. (2006), Hastings and Popper (2005), and Southall et al. (2008). The generated noise associated with the operation of WTEBS can be of concern, as the plants operate permanently over a long period of 25-30 years (Rucker and Friedl, 1985). The operational acoustic noises from onshore WTEBS in the marine environment are caused mainly from the vibration of pipelines, however, there are no evidence of such an impact being studied in the literature. For offshore systems, cold water pipelines, water pumps, and noise associated with devices in a typical WTEBS plant (such as pumps associated with the transport of working fluid) are the main contributor of noise (Rucker and Friedl, 1985, Janota and Thompson, 1983). Ducatel et al. (2013) conducted a preliminary study to predict the potential acoustic impact of an OTEC plant due to onboard machinery and noted the potential impacts of the system on marine mammals at short distances, less than 200 *m*.

2.2.7 Electromagnetic effects

The generated electricity by offshore-based OTEC systems may be transmitted to shore using a network of cables that are mounted on the seabed. Transmission of the produced electricity through these cables will emit low-frequency electromagnetic fields (EMF) (Boehlert and Gill, 2010). A number of marine organisms use electroreception as a fundamental sensory mode for mate finding, feeding, and navigation (Boehlert and Gill, 2010, Hutchison et al., 2020, Öhman et al., 2007, Kirschvink, 1997, Whitehead and Collin, 2004). it is likely that EMF from power cables will have a direct effect on these animals. Scott et al. (2018) indicated that EMF from sub-sea power cables affect edible crabs both behaviourally and physiologically. Westerberg and Lagenfelt (2008) reported a significant change in eels migration swimming speed around the sub-sea power cables. Other growing concerns regarding mounted or buried power cables include an increase in temperature of the adjacent water, sedimentation, and impacts on benthic ecosystems due to electricity transmission (Boehlert and Gill, 2010). Further investigation is recommended for better understanding of the impact of sub-sea power cables on marine organisms.

3 Modelling of Discharge Dispersion

Discharge dispersion modelling of WTEBSs can assist addressing concerns regarding their impacts on the sustainability of marine environments and provide opportunities for achieving maximum effluent mixing efficiency and understanding of the mixing behaviour of plume jets.

The application of modelling of discharge dispersion is not confined to WTEBSs as the topic is also of interest in other growing technologies such as desalination plants, thermal power plants, and aquafarming that discharge a considerable amount of wastewater directly back to waterbodies. Desalination brine, a by-product from desalination plants, comprises high concentrations of dissolved substances and suspended solids as well as possible waste heat (Jiang et al., 2014). Thermal power plants of coastal cities discharge enormous quantities of waste heat into seas and lakes (Pryputniewicz and Bowley, 1975), while aquafarming effluent is typically enriched in suspended organic solids, carbon, nitrogen, and phosphorus (Zeng et al., 2013), which may have a detrimental impact on many species living around the discharge location.

In general, wastewater discharges from industrial processes are categorized into two major groups based on their density discrepancy with the ambient water bodies (Kheirkhah Gildeh et al., 2014). If the effluent has a higher density than the ambient water, the plume of outfall discharge tends to sink, which is known as a negatively buoyant jet plume. Conversely, if the effluent has a lower density than the ambient water the effluent jet plume, this then rises, which is termed a buoyant plume (Bleninger et al., 2010). Nevertheless, the mixing behaviour of the discharged effluents can show a great diversity of flow patterns, depending on the geometric and dynamic characteristics of the environment and discharge flow (Shao and Law, 2010, Jirka and Domeker, 1991).

In the pertinent literature, the study of submerged jet flows haves been extensively covered. Experimental investigations on the characteristics of inclined brine dense jets, such as maximum jet height rise and concentration field, into stagnant environment can be found in Roberts et al. (1997), Cipollina et al. (2005), and Lai and Lee (2012). These studies reported that dense jets with 60° inclined angle produce the longest trajectory for entrainment and thus the highest dilution. Jiang et al. (2014) and Shao and Law (2010) studied the effects of stationary shallow water with mixing of 30° and 45° inclined dense jets. It was realized that the surface constraint may lengthen jet-spreading distances and reduce surface dilution. They also recommended that the terminal rise related to 60° inclined dense jet is rather high and therefore the angle may be too large to provide efficient mixing in shallow waters.

Pryputniewicz and Bowley (1975) investigated turbulence buoyant jets that are vertically discharged into a large body of stagnant non-stratified water. The temperature characteristics of a hot rising plume as a function of discharge Froude number and discharge depth were illustrated. The impacts of horizontal buoyant jets discharged into stationary environment and the effect of bed proximity or so known as the Coanda effect, were detailed in Sharp (1975), Sharp et al. (1977), and Sobey et al. (1988). Coanda effect occurs when the jet discharge is placed close to the bed boundary, the discharge will then cling to and proceed along the boundary (Shao and Law, 2010). This improves the mixing efficiency of buoyant flows, while for saline dense jets, it may cause negative effects on benthic communities around the impacted area (Shao and Law, 2010). Huai et al. (2010), Kheirkhah Gildeh et al. (2014), Kheirkhah Gildeh et al. (2015) carried out numerical modelling of turbulent buoyant jets in stationary ambient water. These studies applied Reynolds-Average Navier-Stokes (RANS) combined with different turbulence closure models. Their findings showed that realizable $k-\epsilon$ and Launder, Reece, and Rodi (*LRR)* turbulence models were the most reliable and accurate in modelling Coanda effect, buoyant and non-buoyant jet in stagnant environments.

Abessi et al. (2012) conducted a series of experimental tests for negatively buoyant effluents discharged through a protruding surface channel into unstratified stagnant water. The results show that the influence of free-surface on the entrainment and mixing of the flows is small. Abessi and Roberts (2014) carried out comprehensive laboratory experiments on multiport diffusers for negatively buoyant effluents into stationary water. Their results recommended that to prevent reduction in entrainment, it is essential to consider sufficient spacing between the designed ports. Ardalan and Vafaei (2018) developed a classification chart for thermal-saline inclined single-port jet, as a result of an extensive set of laboratory experiments for thermal-saline effluent with three different discharge angles of 30°, 45°, and 60° in stagnant water environments. This is subsequently followed on in Ardalan and Vafaei (2019) where they carried out numerical and experimental studies of negatively buoyant jet discharged with 45° inclined angle in a stationary water; simulations were conducted using a RANS model with realizable $k-\epsilon$ model and the outcome showed good consistency with the results of physical modelling. Rodríguez-Ocampo et al. (2020) implemented an OpenFOAM-based solver that can be applied in modelling thermal discharge into water bodies. The solver was suitable for simulating three fluid phases with different densities and temperatures, i.e., two miscible liquids and air, and was validated against an experiment of a multiphase dam-break. However, the model did not consider buoyancy effects. More recently, a study of submerged thermal-saline jet discharge into a stagnant environment using the LES turbulence model was carried out by Azadi and Firoozabadi (2022). The results illustrated that the flow patterns only depend on the density ratio, which is the thermal flux to salinity flux ratio. The main drawback of the above group of studies was that they have not considered the marine environment conditions including wave and current flow.

Investigations on the characteristics of jets into non-stationary environments have also been widely carried out; notably, Roberts and Toms (1987) conducted a series of experiments on the characteristics of vertical and inclined dense jets with different angles discharged into a uniform crossflow of various velocities and directions. As a result, they discovered that inclined jets are generally preferable to vertical jets. When a submerged discharge outlet is located where currents may flow in all directions, then vertical jets may be the preferable choice instead of inclined jets (Ahmad and Baddour, 2012). Mossa (2004) conducted laboratory experiments for turbulent nonbuoyant jets that are vertically discharged into two different environments, one with stagnant ambient water and a second with regular waves; they observed higher entrainment velocities in the latter case. An experimental study on the behaviour of horizontal non-buoyant jet located at the mid-depth of a shallow water wave environment was investigated by Ryu et al. (2005). The results revealed that the influence of wave amplitude on jet diffusion is substantial. Zhen et al. (2007) numerically simulated seawater temperature field to monitor the environmental impacts of hot effluent discharged from a seawater-source heat pump in Dalian, using a two-dimensional convection-diffusion equation model; the water temperature elevation impacts on the marine ecosystem were found to be negligible. Yu et al. (2009) established a two-dimensional hydrodynamic model to predict and optimise the thermal plume from a Rizhao power plant discharge on Rizhao sea. Chen et al. (2012) conducted numerical modelling of a buoyant and non-buoyant round jet discharge into wave environments using Large Eddy Simulation (LES) where the buoyancy effect was considered using the Boussinesq assumption. The results were validated against the experimental data in Chen et al. (2009). As an outcome, they realised that under the buoyancy force the wave effect on jet entrainment and mixing is considerably weakened.

Other related work, such as Pat Grandelli et al. (2012) developed and validated a three-dimensional time-dependent model for predicting biological and physical impacts of OTEC. The model simulated negatively buoyant discharge flows by a dynamically coupled Lagrangian jet-plume entrainment model in the near-field, and by dynamic oceanic circulation and turbulence in the far-field for the water surrounding O’ahu in Hawai’i, USA (Figure 6). The model is used to define the effect of nutrient-rich and low-oxygen deep sea water on increased productivity of phytoplankton. Similarly, Kim and Kim (2014) developed a primitive three-dimensional model to predict and minimise the mixing behaviour of thermal discharges of an OTEC system in coastal water of Kosrae, Micronesia. They declared that the model was capable of reproducing the plume behaviour. More recently, the effect of free-surface waves in temperature distribution in thermal boundary layer region close to the seabed was analytically modelled in Michele et al. (2021) and Michele et al. (2023). The study suggested a need for expanding existing models that neglect the effects of free-surface wave field.

There are also some well-known commercial models that have been widely used for predicting the effluent discharges in waterbodies. In this group, Lee et al. (2003) implemented a Lagrangian interactive virtual reality model (JETLAG/VISJET) based on the project-area entrainment hypothesis and a heuristic theory to treat the shear to vortex entrainment transition. Frick (2004) developed the VISUAL PLUMES (VP) model which is a platform for mixing zone modelling. Jirka and Domeker (1991) introduced an integral model for turbulent buoyant jets in unbounded stratified flow which was coded into a Fortran program COREJET/CORMIX. Palomar et al. (2012) carried out a detailed analysis of these commercial models (i.e. JETLAG, COREJET, and VP) and realized insensitivity of these models in predicting the influence of crossflow direction on jet behaviour.

Most recently, Xu et al. (2016) investigated a nonbuoyant vertical round jet in a wave-current coexisting and current-only environments, both numerically using LES, and experimentally. They observed the effluent clouds phenomenon in the wave-current coexisting which leads to considerable increment of jet spread and dilution. Xu et al. (2017) investigated the impact of regular waves on three dimensional scalar structures of a vertical jet in the wave-following-current environment using numerical modelling of submerged non-buoyant vertical round jets.

Followed on, Xu et al. (2018) developed a set of semi-empirical equations to quantify the wave effect on the initial dilution of wastewater discharge based on numerical modelling of non-buoyant jet discharges in wave-following-current environments. This is extended in Xu et al. (2019) where they conducted several experimental tests about submerged multiport diffuser effluent discharges in a wavy cross-flow environment. It was discovered that the wave-to-current velocity ratio is a very important parameter in describing effluent discharge dilution. Comparably, Fang et al. (2019) implemented an integral model for predicting the characteristic behaviour of a buoyant jet in wavy crossflow environments.

A set of laboratory tests in modelling of submerged negatively buoyant outfall under typical conditions in the Mediterranean Sea was carried out by Ferrari et al. (2018). The results revealed that the strongest waves tested in the study tend to decrease dilution, while the weakest waves tend to improve it. Anghan et al. (2022) reviewed the literature of the jet in the wave environment and identified the various mean and turbulence quantities of the jet in the regular and random waves environment. They concluded that the behaviour of the jet can be predicted based on the ratio of the jet inlet velocity to the wave orbital velocity.

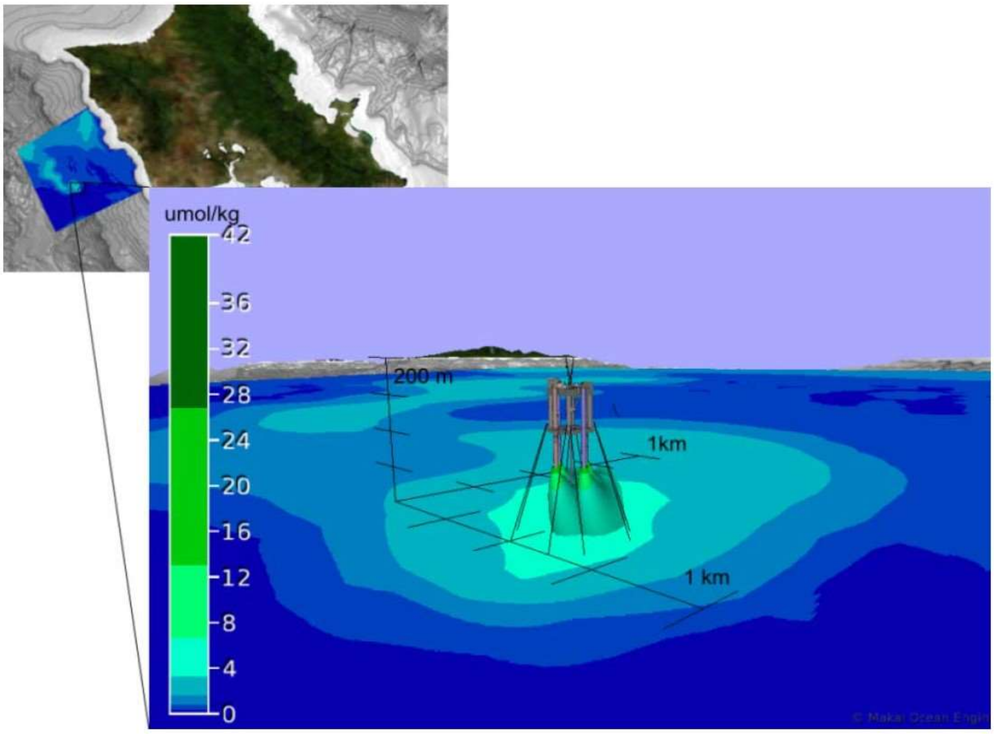

Figure 6: Simulated plume and nitrate (nutrient) concentration of discharge dispersion of a 100 *MW* OTEC plant with four 70 *m* depth and 1 *m/s* mixed discharges by Pat Grandelli et al. (2012).

As listed above, many numerical and experimental studies have been conducted to study effluent dispersion, however the literature lacks a comprehensive and sophisticated computational fluid dynamics (CFD) model to simulate the hydro-thermal behaviour of discharged effluents into waterbodies under combined wave-current conditions. Table 1 presents the advantages and disadvantages of the numerical investigations on effluent dispersion into non-stationary environments. Recent advances in development of numerical tools in simulation of hydrodynamics of wave and currents in mesh-based approach (such as, Higuera et al. (2013) that developed a realistic wave generator and active wave absorber for the Navier-stokes equation and Larsen and Fuhrman (2018) that implemented a new turbulence model capable of predicting accurate pre- and post-breaking surface elevations, as well as turbulence and undertow velocity profiles of surface waves) and mesh-less approach (such as, Ni et al. (2018), Ni et al. (2020) that implemented a numerical wave-current flume based on Smoothed Particle Hydrodynamics (SPH)) provide an opportunity to bridge this knowledge gap.

Table1: Advantages and disadvantages of numerical models in simulation of submerged jet into non-stationary environment.

Submerged jet into non-stationary flow modelling	Advantages	Disadvantages
Zhen et al. (2007) and Yu et al. (2009)	- Simulate seawater temperature field in two-dimensional field	- Can only be applied in shallow water - Lack of validation - No buoyancy effect modelling
Chen et al. (2012)	- Consider buoyancy effect - Validated against laboratory data	- Only valid under wave condition with no current flow
Pat Grandelli et al. (2012)	- Large-scale modelling - Consider buoyancy effect - Capable of biological modelling	- Only designed for OTEC systems - Not applicable for shallow water modelling
Kim and Kim (2014)	- Can reproduce the plume behaviour in coastal water	- Lack of validation
JETLAG/VISJET (Lee et al., 2003), VISUAL PLUMES (Frick, 2004), COREJET/CORMIX (Jirka and Domeker, 1991)	- Capable of fast prediction of mixing zone characteristics - Platform for mixing zone modelling	- Insensitive in predicting the influence of crossflow direction on jet behaviour - Commercial software
Xu et al. (2016), Xu et al. (2017), Xu et al. (2018), and Xu et al. (2019)	- Simulate jet flow under different combination of current and wave conditions	- No temperature distribution modelling
Fang et al. (2019)	- Predict the characteristic behaviour of a buoyant jet in a wavy crossflow environment	- No temperature distribution modelling

4 Biofouling and Corrosion

Exposed surfaces of systems that use seawater as the main processing fluid can be affected by the physiochemical properties of seawater such as fouling and corrosion (Abidin et al., 2021). Fouling occurs as a result of the deposition of dissolved and particulate matter in the water on surfaces that are in contact with it (Abd El

Aleem et al., 1998). The undesired growth and accumulation of foulant on surfaces in contact with water can potentially affect the system’s efficiency, while damaging equipment in the process (Abidin et al., 2021). Uncontrolled growth of fouling can have damaging consequences to WTEBSs (Abidin et al., 2021), marine vessels (Magin et al., 2010), rigs (Gormley et al., 2018), marine aquaculture (Fitridge et al., 2012), and other infrastructure that is submerged in the sea. Crystalline fouling, organic fouling, particle and colloidal fouling, and microbiological fouling are categorised as the most important types of fouling (Flemming, 1997, Al-Juboori and Yusaf, 2012). Among them, controlling biofouling (microbiological fouling) is the most complicated one (Flemming, 1997, Al-Juboori and Yusaf, 2012).

Marine biofouling is the unwanted growth of marine micro- and macro-organisms like bacteria, algae, sponges, barnacles, mussels, Balanus etc. (Mahto and Pal, 2020). The growth and accumulation process of biofouling on the exposed surfaces are detailed in Abidin et al. (2021), Flemming (1997), Al-Juboori and Yusaf (2012), Maddah and Chogle (2017), and Mitchell and Benson (1980).

Figure 7 illustrates the growth of fouling inside of the pipeline and on the mesh filter basket of the intake pipeline of a pilot SWAC system at Brixham laboratory, University of Plymouth, United Kingdom. The SWAC system has been out of service for many years, while the intake pipeline, with an internal diameter of 5.08 *cm*, has been used regularly for filling seawater tanks (i.e. abstracting seawater for 1 or 2 hours per day) for other activities in the laboratory. The pipelines were installed in the early 1980s and no anti-fouling treatment has been carried out since, whereas the pipeline and the mesh filter basket have been pressure washed once in 2007. When retrieved, a growth of fouling with a thickness of 2-3 *mm* is observed in the pipelines, while the mesh filter basket is covered with micro- and macro-organisms (e.g.bacteria, algae, barnacles, Balanus)(Euroswac, 2021).

Bott (2011) classified the parameters that can influence biofouling growth into three main categories of chemical, physical, and biological, as listed in table 2.

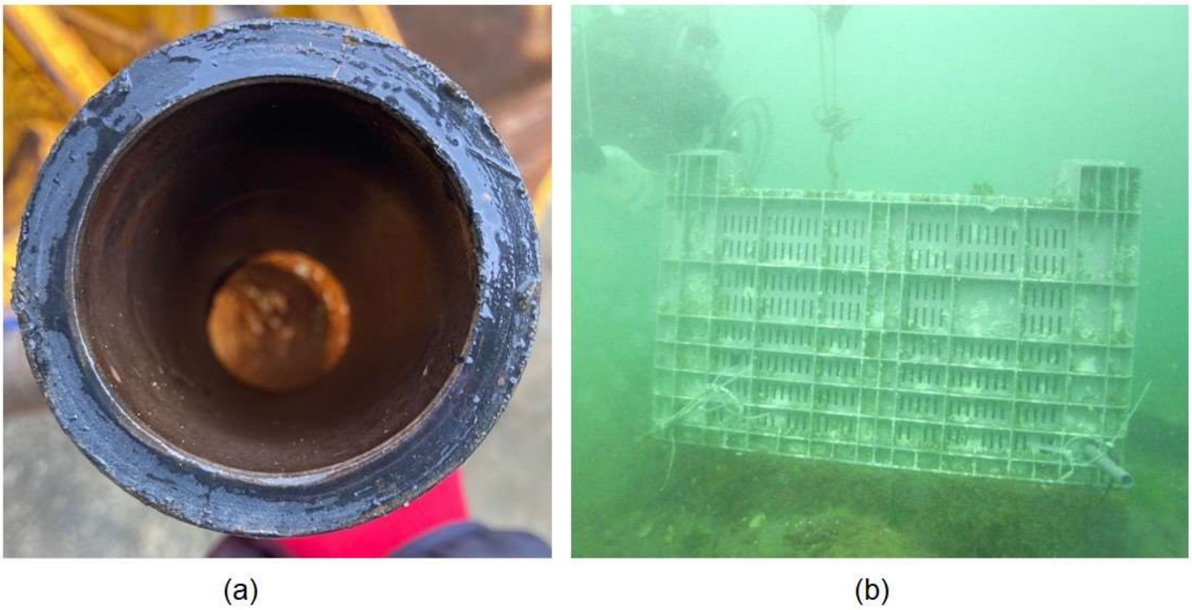

Figure 7: (a) Biofouling at the intake pipeline, and (b) The mesh filter basket at the inlet of the intake pipeline of the pilot SWAC system at Brixham laboratory, Brixham, UK (Euroswac, 2021).

Table 2: Chemical, physical, and biological parameters that affect biofouling growth (Bott, 2011).

Chemical	Physical	Biological
Substrate type	Temperature	Microorganism type
Substrate concentration	Fluid shear stress	Culture type
pH	Heat flux	Suspended cell concentration
Inorganic ions	Surface composition	Antagonist organism
Dissolved oxygen	Surface texture	
Microbial inhibitors	Fluid residence time	

Untreated fouling can lead to increases in the thermal resistance as well as required pumping power (Mitchell and Spitler, 2013). Abidin et al. (2021) and Jenkins (1978) affirmed that in the design of OTEC systems, biofouling is an inevitable condition that cannot be avoided. They highlighted the impacts of flow velocity and temperature of the seawater intake as two main parameters on the control of biofouling growth. The relationship between flow velocity and biofouling growth is complicated to correlate due to its dual impacts. The rapid velocity of the water can provide sufficient oxygen and nutrient that favours the growth of macrofoulants, but it can also prevent biofouling growth if the water shear rate surpasses the shear rate of biofouling settlement (Flemming, 1997, Jenkins, 1978). Panchal and Knudsen (1998) pointed out that seawater temperature in the range of between 20°C to 50°C is desirable for microorganisms growth which explains why high-temperature surface seawater exposed to continuous sunlight accommodates the growth of biofouling (Mitchell and Spitler, 2013). Likewise, higher potential for biofouling is anticipated at shallow water-based onshore facilities in comparison to offshore ones owing to the high concentration of organisms in seawater adjacent to the shoreline (Avery and Wu, 1994). Seasonal seawater temperature changes also influence the potential for biofouling growth, for example, low range of temperature changes in tropical area, provides a steady condition for biofouling development (Affandy et al., 2019).

One of the most common techniques employed to kill organisms in WTEBSs is through the use of biocides (Makhlouf and Botello, 2018) namely via oxidising and non-oxidising types. Oxidising biocides, such as chlorine, peracetic acid, bromine, and sodium bromide, attack microorganisms by disrupting nutrients from passing across the microorganism cell walls (Makhlouf and Botello, 2018, Ilhan-Sungur et al., 2015). On the other hand, non-oxidising biocides, such as 1,2-benzisothiazolin-3on and 5-chloro-2-methyl-4-isothiazolin-3-on, interfere with reproduction, respiration process, and harms the microorganism cell walls (Makhlouf and Botello, 2018, Ilhan-Sungur et al., 2015). These biocides can target more specific biochemical pathways, thus reducing the potential of unwanted side-effects, but the targeted bio-foulants may become resilient to them. In open systems, due to environmental concerns of chemical discharge, only using direct injection of the oxidising agent, such as sodium hypochlorite (chlorine) is allowed (Mitchell and Spitler, 2013).

Anti-fouling coating is another usual practice in marine and maritime industries to prevent biofouling. Until recently, tributyltin (TBT) was an active biocide ingredient in many paints that were very successful in reducing biofouling (Chambers et al., 2006). However, its use has been prohibited as it was found harmful to marine organisms. Its replacements include use of metallic species, such as copper and zinc and many other alternatives are highlighted in detail in Chambers et al. (2006).

Abidin et al. (2021) elaborated on a list of common and potential techniques of biofouling assessment for OTEC systems including microscopic optical, spectroscopic, physical assessment, electrical, biological and chemical detecting techniques. This list can be generalized and adapted for biofouling assessments for all other types of WTEBSs. Makai Ocean Eegineering Inc. (2014b) stated as a result of long-term testing of heat exchangers that fouling is not a serious problem with WTEBSs that intake cold deep seawater in the range between 3°C to 8°C. However, for warm water systems, e.g., OTEC systems that intake warm surface water with temperatures above 25°C, biofouling is unavoidable. In addition, other system components such as strainers, pumps, holding tanks and pipeline fittings are among the equipment that are most at risk of being exposed to potential biofouling (Abidin et al., 2021). Berger and Berger (1986) recommended that injection of chlorine at a concentration between 50 to 70 *ppb* for 1 hour per day (24-h average of 2-3 *ppb*) can completely prevent fouling in systems and can be used as a continuous and non-destructive method of prevention.

Apart from biofouling, corrosion can also impact WTEBSs' performance due to seawater interactions with system components and structures. Corrosion is defined as the process of destruction of material under the chemical or electrochemical action of the surrounding environment (Pourbaix, 2012). An essential key to improving the marine structures' optimum service life against corrosion is understanding the type of marine environment, materials used, appropriate design, and corrosion control measures (Shifler, 2005).

An important controlling factor in structures made using metals and alloys is the formation of a passive film that reduces ionic transport of reactive species (Shifler, 2005). In seawater, the dissolved oxygen and chloride ions lead the formation and repair or breaking down of passive films (Shifler, 2005). Environment parameters such as atmospheric salt concentration, temperature, oxygen concentration, salinity, and flow-related corrosion parameters (e.g. Erosion-corrosion (Shifler, 1999) and cavitation (Hoyt and Furuya, 1985)) need to be considered. The presence of bio-films also increase the corrosion rate of a structure or operate as a passive deterrent (Shifler, 2005, Videla and Characklis, 1992). Proper design, including a selection of compatible materials from both corrosion and mechanical aspects, optimizing geometries and joining processes that minimize corrosion, and utilization of corrosion control measures, is the most effective way to reduce corrosion costs (Shifler, 2005). Typically, corrosion can be controlled by using coatings that act as either ionic filters, oxygen diffusion barriers or cathodic protection that can be very cost-effective solutions (Shifler, 2005, Diler et al., 2020). For WTEBSs that need to have pipelines across large depths, it is advised that polyethylene is an excellent choice of material as the pipelines will not corrode or contaminate the water (Elsafty and Saeid, 2009). In heat exchanger systems, corrosion due to the salty seawater can be eliminated using either titanium or aluminium heat exchangers; titanium is proposed as a low-risk solution for a condenser, especially when employed in cold seawater (Elsafty and Saeid, 2009, Van Ryzin and Leraand, 1991, Makai Ocean Eegineering Inc., 2014a).

5 Permits and Licensing

Brixham Laboratory at the University of Plymouth has its pilot shallow-water-based SWAC system installed but was not used since 1990s. Through a project, EUROSWAC (Euroswac, 2021), the facility was reawakened and modified to enable the SWAC performance to be monitored. This required obtaining the necessary permitting and licensing approvals from the relevant authorities in the United Kingdom, such as the Marine Management Organization and Environmental Agency. All necessary permits in order to construct and operate have been identified and simplified in a licensing flow-chart presented in Figure 8. This flow chart is applicable and valid for the installation and operation of other types of WTEBSs in English waters.

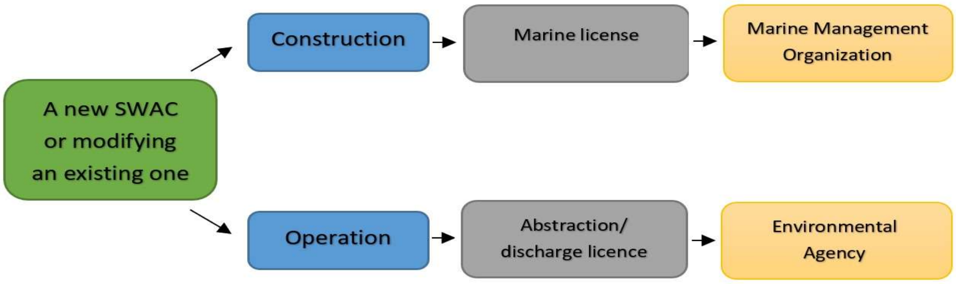

Figure 8: Permits and licensing flow chart for installation and operation a SWAC facility in the United Kingdom.

In the United Kingdom, the Marine Management Organization defines the deployment of new WTEBSs, or any extension of an existing system including installation of new pipelines, fits into the construction, dredging and deposit category. The Marine Management Organization offers an assistance tool to guide and check if the construction of a new WTEBS or any extension of an existing system, requires a marine licence. Generally, the tool provides three options regarding the marine licence:

- Exemption: In certain circumstances, the need for a marine licence can be removed. Note that an exemption is not applicable for the construction of new underwater structures for example non-oil and gas pipeline.
- Self-service marine licence: This licence covers a number of activities that can be considered low-risk activities. This excludes deploying a new WTEBS or any extension of an existing systems.
- Standard marine licence: If a proposed activity does not meet the exemption or self-service marine license criteria. Application for a standard marine licence is required for both new WTEBS and extension of existing systems that includes the installation of new pipelines which needs to be authorised by the Marine Management Organization.

Abstraction and discharge licences are controlled by the Environmental Agency. A full licence is required when the abstraction volume flow rate surpasses 20 m^3 a day. For heat exchangers and discharge to surface water, the Environmental Agency requires that the maximum temperature of water at the borders of the mixing zone should not exceed 23 °*C*. Furthermore, it requires that the maximum temperature rise outside of the mixing zone should not be higher than 3 °*C*. The size of a mixing zone is not directly defined or constrained by the Environmental Agency and is on a case-by-case basis, dependent on local geography and environment data. This is not true for Scotland where the mixing zone is defined as 100 *m* from the centre outward in every direction. If biofouling control measures are taken, their effects on the discharge water need to be considered.

In the case of the WTEBSs, the Environmental Agency offers an option for submitting a single application for abstraction and discharge licenses as long as the following conditions are met:

- Discharge volume is lower than 1000 m^3 a day.
- Temperature regulations outlined for discharge to surface water as mentioned above are met.
- No polluting chemicals present in the discharge.
- Discharge is to the same water body as the abstracted water, but not close to 200 *m* to another heated discharge.
- Discharge is not at any water body containing protected species or within 100 *m* from a local wildlife site.
- Discharge must not be a watercourse point where salmon spawn.

Interested readers are referred to government websites for more details on each application. A similar flow chart can be designed to summarise the required permits and licensing process for other territorial waters.

6 Environmental Impact Assessment of Pilot SWAC System at Brixham Laboratory

In the EUROSWAC project, an environmental impact assessment study was performed to measure water quality parameters near the discharge area of the pilot shallow-water SWAC system at Brixham laboratory (Euroswac, 2021). For this purpose, two buoys which were equipped with sensors for monitoring water properties (such as water temperature, pH, oxygen concentration, dissolved oxygen saturation, conductivity, total dissolved solids, and the oxidation-reduction potential) were deployed for nine months at different positions between 5 *m* and 40 *m* from the discharge outlet situated near the shoreline; the SWAC system had a discharge flow rate of around 150 *litre⁄min*. During this period, the pilot system was run for short periods of up to 8 hours to assess changes in water quality, but no significant and detectable change in water properties were observed.

To ensure that the discharge complies with mixing zone requirements outlined in the temperature regulations by the Environmental Agency, a numerical model was developed in OpenFOAM®. The model solves governing hydro-thermal equations to predict temperature distribution field created by submerged hot seawater discharge into cold seawater while accounting for marine environment influences from dynamic interactions of currents and waves, where it is verified against a wide range of analytical and experimental data. Different scenarios were modelled using data from the pilot SWAC discharge with varying ambient environmental conditions. Figure 9 shows the set conditions and geometry specified for the computational domain in one of the simulated cases based on experimental discharge data captured at the bay in Brixham. Simulations were conducted with the aid of high-performance computing due to heavy computational requirements to predict the dynamic thermo-fluid current-wave interactions in three-dimensions. A snapshot of the results in Figure 9 depicts the iso-surface of mean seawater temperature discharge at T = 16 ℃ when the flow is fully developed which means that the seawater temperature rise beyond that area is less than 1 ℃. It clearly delineated the operational condition of the SWAC at Brixham produces a small impact footprint of detectable temperature alterations only between 4 to 5 meters in size from the discharge point, which conform to the abovementioned temperature regulations for abstraction and discharge licenses of WTEBSs.

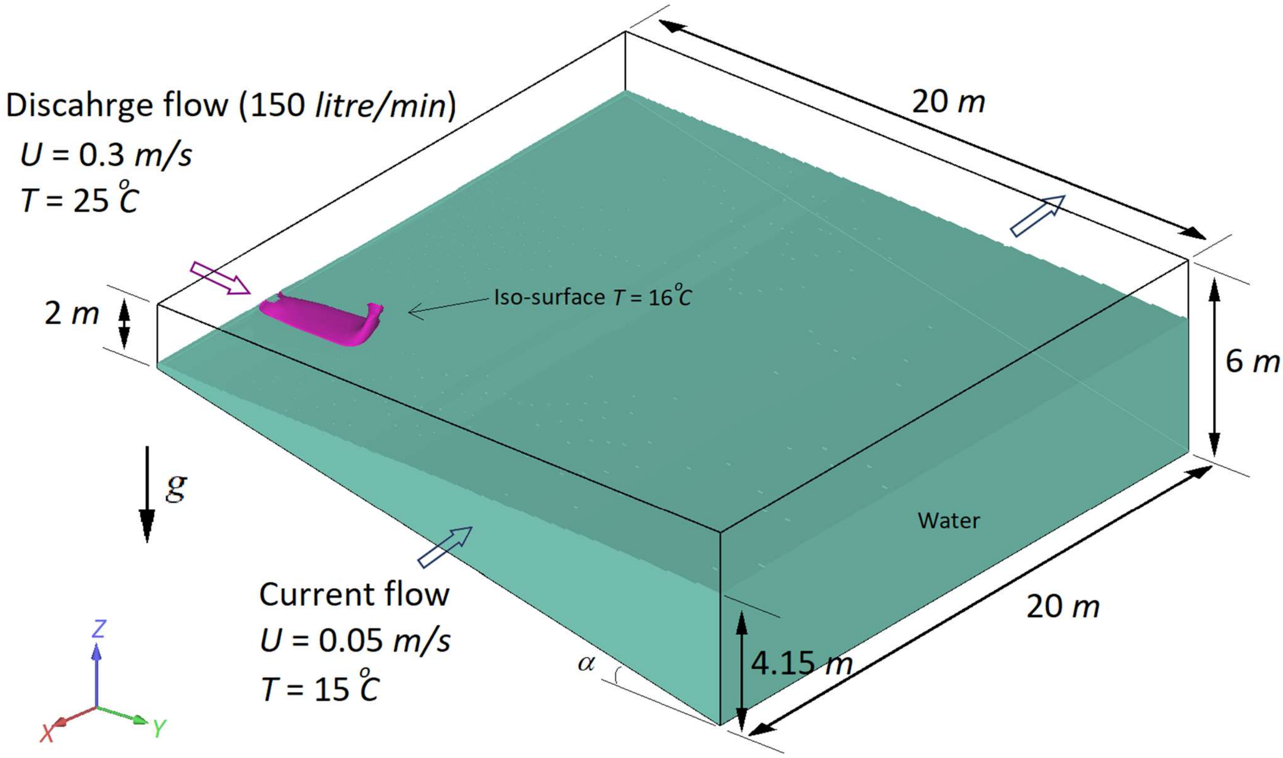

Figure 9: Numerical modelling setup and iso-surface of mean seawater temperature discharge for T = 16 °C for a fully developed flow condition.

7 Conclusion

The growth and development of WTEBSs raise concerns regarding their impacts on sustainability and degradation of marine environments. The present paper provides a full review of previous studies and state-of-the-art in different aspects of WTEBSs' interactions with marine environments. The study highlighted the relevant concerns on the development of WTEBSs including different stages of construction, operation, and decommissioning based on other types of development in the marine environment such as coastal power plants or other marine-based renewable technologies. The construction and decommissioning phases of a WTEBS, including installation of foundations and hard-fixed structures (such as submerged heat exchangers or pump stations), pipelines, scour-protection systems, mooring devices, and seabed mounted power cables are likely to cause significant positive and negative disturbances to local environmental resources and fundamental changes to the benthic habitat. Innovative new solutions such as those proposed by DeProfundis and DORIS Engineering in self-burying and flexible pipe technologies can assist with minimising the environmental impact and costs of pipeline installations. Operation-wise, WTEBS continuously affects the marine environment throughout its lifetime of between 25 to 30 years. A comprehensive review of the environmental impact associated with discharge of processed seawater, power cable electromagnetic fields, acoustic effects from the WTEBS machinery and pipelines, and leakage of chemicals from the system on benthic and pelagic ecosystems was presented. As discharge dispersion is one of the main environmental concerns, related experimental works in the area are reported, following by numerical tool employed to predict their effects. The lack of a comprehensive and sophisticated computational fluid dynamics (CFD) model to simulate the hydro-thermal behaviour of discharged effluents into waterbodies under combined wave-current conditions was discovered, and scopes for improving the existing models to bridge the knowledge gaps were discussed.

The potential destructive impacts of fouling and corrosion in WTEBSs were subsequently presented, followed by an example observed at Brixham laboratory. Deterrent recommendations, such as using HDPE pipelines, materials for heat exchangers, appropriate designing and assessment, as well as injection of chlorine as a continuous and non-destructive method were highlighted.

Required permitting applications and licensing processes for installation and operation of new WTEBS or modification of an existing one by the relevant authorities in the United Kingdom are summarized. Current regulations may subject to changes in future with growth in the development, deployment, and adoption of WTEBSs.

Most of the information regarding the environmental impacts of WTEBS came from studies that investigated pre-impact conditions at a potential WTEBS site, or those that have been adapted from other marine technologies environmental impacts. Actual monitoring of the environmental impact of WTEBSs during operation is therefore lacking and thus necessary. The finding from data monitoring of water quality properties for short term operation of a pilot SWAC system at Brixham Laboratory at the University of Plymouth in the United Kingdom was discussed. It was found that no large detectable changes in water quality are measured, with seawater mixing zone temperature variation being very localised that complies within limits to what is allowable by permits and licensing regulations. However, further studies would be required if demand for abstraction and therefore discharge of seawater flow rate increases, and if the SWAC system was to operate continuously to assess long term effects.

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