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Using an Agent based model (ABM) to predict fish interactions with a Tidal Stream Turbine

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Using an Agent based model (ABM) to predict fish interactions with a Tidal Stream Turbine

A dissertation in partial fulfilment of the requirements for the degree of Master of Science by Research (MScRes) in Ocean Sciences.

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Declaration

This thesis is being submitted in partial fulfilment of the requirements for the degree of Master of Science by Research (MScRes) in Ocean Sciences, under supervisory agreement of its submission.

I hereby declare that this thesis is the results of my own investigations, except where otherwise stated. All other sources are acknowledged by bibliographic references.

This work has not previously been accepted in substance for any degree and is not being concurrently submitted in candidature for any degree unless, as agreed by the University, for approved dual awards.

Signed (Candidate)

25.05.2023 ________________

Date Signature

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R.G.

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Abstract

There is a concern regarding how changes in local hydrodynamics as the result of tidal stream turbine (TST) arrays may affect foraging opportunities for piscivorous marine mammals and seabirds. The 3D behaviour and distribution of forage fish determines its availability to predators and understanding how TST alter school characteristics helps estimate impacts on foraging opportunities. However, previous methodologies used to study impacts of changing hydrodynamics on fish and top-predator populations around TST have struggled to comprehensively quantify school characteristics across the water column due to the turbulent nature of tidal stream environments and the high flows experienced there. To overcome challenges, and provide insights into potential changes in foraging opportunities, this study applies an agent-based model (ABM) approach to a high-fidelity simulated TST wake, estimating responses of forage fish to installations. The results here indicate that the schooling behaviour of fish has the biggest influence on individual responses to a TST. I also show that the presence of a TST has little effect on the behaviour and density of schooling fish within a tidal-stream environment. Yet, we also showed that a tidal-energy device still provides top-predators with a foraging hotspot that contains fish aggregations which consistent in both space and time. We therefore demonstrate the potential to simulate how fish and top-predators interact with a tidal turbine structure at a fine-scale, which can (once validated) be applied to understanding scaling concerns and providing a more accurate assessment of risks for legislators and planners.

Introduction

Globally, governments are setting ambitious targets to increase renewable energy utilisation, such as wind and solar (e.g. REN21, 2019). The UK has an estimated tidal-stream energy resource of 95 TWh per year (Crown Estate, 2012). There are currently 14 sites in the UK and British Channel Islands that are under-development, from these sites, approximately 10.8MW is currently operational (Coles et al., 2021), with the majority of sites constrained around Orkney and the North of Scotland (Wolf et al., 2021). Although recently developments have been made, there is still a scarcity of operational turbines. Development of devices is still in it's infancy due to problems of energy device interaction with the environment (Wolf et al., 2021), including both environmental interactions and the challenge of constructing devices that can withstand the harsh environment. Moreover, introduction of new Marine Renewable Energy (MRE) devices has been constrained due to the legal requirement to reduce environmental impacts, coupled with high-uncertainty with regards to assessing impacts (Wolf et al., 2021). The impact of tidal-stream device developments has potentially drastic changes to an ecosystem, which would pose far-reaching negative ecological affects, as tidal dissipation is responsible for the transport of nutrients, pollutants and individual organisms, through an ecosystem. Due to the scale of these changes, both spatially and temporally over the operating lifetime of a device, it can be hard to accurately assess their ecological consequences (Neill et al., 2018; Waters and Aggidis, 2016). Difficulties in permitting of new installations has been cited as a major inhibitor to MRE development (Neill et al., 2018). Better assessment of the ecological consequences of MRE devices would allow an increase in growth of this industry and allow the UK to meet the ambitious renewable energy targets that it has set itself.

Unlike other forms of marine renewable energy, tidal turbines have moving components that sit within the water column, posing novel risks to local wildlife populations (e.g., Grecian et al., 2010). Seabirds and marine mammals (hereafter referred to as top-predators) and fish populations have been identified as the most at-risk species from negative interactions with turbine structures (e.g., Benjamins et al., 2015). The three main interactions considered when consenting of Marine renewable energy installations (MREI) are: death, disturbance and displacement (Wolf et al., 2021). Death is considered in this process as the collision risk of individual animals with a MREI, and the estimated survival rate from a collision. Collision risk has been assessed through both field and laboratory experiments (e.g. Yoshida et al., 2021; Williamson et al., 2019; Shen et al., 2016; Onoufriou et al., 2021) and although not definitive, the studies indicate that collision risk for fish (Fraser et al., 2018; Yoshida et al., 2021; Shen et al., 2016; Williamson et al., 2019; Hammar et al., 2013; Broadhurst et al., 2014) and marine mammals is low (Sparling et al., 2018; Joy et al., 2018; Hastie et al., 2019b; Onoufriou et al., 2021; Gillespie et al., 2021), yet information on seabirds remains largely qualitative and appears to be species-dependant (Robbins et al., 2014). Disturbance effects of MREI perturbs to a source of stress that directly affects the behaviour of an individual. For example, noise pollution during construction and operation is an example of displacement effects – for further details see Pirotta et al., (2018). One key source of disturbance suggested are the effects of habitat modification caused by altered hydrodynamics in the wake of a MREI (Wolf et al., 2021; references within). The presence of a MREI usually will slow the main flow of a tidal stream, but with increased flow speed past the structure and increased turbulence in its wake (Wolf et al., 2021). Yet we still do not have a good understanding how this altered state may indirectly affect the foraging opportunities for both fish and top-predators. Finally, displacement effects around a MREI occur when a source of disturbance is so large that individuals may no longer frequent the affected area. Noise pollution during construction and operation of MREI has shown to be a source of disturbance for *Phoca vitulina* populations (Joy et al., 2018; Sparling et al., 2018), this being exemplified into a displacement affect when singular devices are scaled up into arrays (Onoufriou et al., 2021). Yet uncertainty remains when assessing the risk of MREI on local wildlife populations due to the limited number of studies and the possibility of site-specific habitat use. To better assess the risks posed by these developments, there is a need to understand site-specific processes and habitat use. The use of tidal stream environments by top-predators for foraging is well documented in the literature. As these are sites where prey availability for top-predators is thought to increase (e.g. Zamon, 2003). We still do not have a good understanding of how altered hydrodynamics as a result of a MREI may indirectly affect the foraging opportunities for both fish and top-predators. The 3D behaviour and distribution of forage fish determines its availability to predators and understanding how MREI alter school characteristics from alterations in local hydrodynamics helps estimate impacts on foraging opportunities, which may have important implications for the reproductive and survival rates of local wildlife populations. However, we do not fully understand the scale of these risks, particularly the impacts as single installations are scaled-up into arrays, preventing growth of the industry. Understanding impacts of habitat modification from altered hydrodynamics could assist environmentally sustainable arrays.

The schooling behaviour of fish is an important consideration here as it plays a key role in the behaviour of an individual. At least 10,000 species of fish have been reported to exhibit schooling behaviour during at least one stage during their life-cycle (Shaw, 1978). There has been much discussion regarding the definition of a school but is generally accepted that a fish school refers to a group of individuals that perform co-ordinated swimming with those around themselves (Pitcher, 2001). Whilst the size of fish schools is obviously extremely dependant on the species of fish, geographical location is another key factor in determining school size. For example, the species of focus for this study, *Pollachius virens,* are capable of forming schools up to 170,000 strong (Misund, 1993), whilst other studies have reported P.virens schools containing between 4 to 483 individuals (Pitcher and Partridge, 1979; Partridge et al., 1980; Misund et al., 1992). Fish schools, although once considered as compact dense units with homogenously behaving individuals (Shaw, 1978), have more recently been observed as dynamic structures in which fish position and behaviour changes consistently (Partridge et al., 1980; Partridge, 1981; Misund, 1993). Due to this, packing density of schools varies greatly from one section of a school to another (Partridge et al., 1980), with the ability of schools to form sub-groups that may act independently from one another (Misund, 1993). Whilst schooling, fish will try to maintain an approximate distance of about one body length to surrounding individuals (Misund, 1993), laboratory experiments on numerous species of fish have shown that fish will aim to swim nearby individuals of a similar size (Pitcher et al., 1985; Pitcher et al., 1986; Theodorakis, 1989; Ranta and Lindström, 1990; Ranta et al., 1992a; Ranta et al., 1992b; Krause, 1994; Krause and Ruxton, 2002). It is hypothesised that this occurs for various reasons including that individuals swimming next to a conspecific of a different size to themselves have a greater risk of being spotted (Krause and Ruxton, 2002), to improve the hydrodynamic advantages of swimming in a school (Breder, 1965; Belayaev and Zuev, 1969), and to minimise the risk of cannibalism (Larsson, 2001). This cohesive behaviour of schooling fish is achieved through the use of visual and acoustic cues as well as through use of their lateral line system (Partridge et al., 1980; Partrdge, 1981). Schooling behaviour is often induced during 'risky' situations (Pavlov and Kasumyan, 2000) and it has been hypothesised that schooling occurs largely as an anti-predatory response (Krause et al., 2000; Pavlov and Kasumyan, 2000; Turesson and Bronmark, 2007), with solitary fish showing an increased risk of predation (Magurran et al., 1985; Godin et al., 1988), but has also been discussed in terms of increased mating and foraging benefits (Partridge and Pitcher, 1979; Pavlov and Kasumyan, 2000; Svendsen et al., 2003) Schooling fish have a reduced encounter rate with predators (Turesson and Bronmark, 2007), are able to recognise danger at a larger distance than solitary fish (Turesson and Bronmark, 2007) and may even benefit from an increased reaction time (Webb, 1980). The confusion effect (Welty, 1934), whereby predators find it hard to focus on one individual due to the dynamic behaviour of fish in schools, is more pronounced in larger schools (Milinski, 1979). Fish also benefit from an increased swimming efficiency due to the hydrodynamics of schooling fish (Breder, 1965; Belayaev and Zuev, 1969), with the resulting hydrodynamics confusing the lateral line system and electro-sensory capacity of a would-be predator (Larson, 2009). Increased feeding intensity has also been reported in schooling fish (Pitcher et al., 1986), although food competition is likely to be greater in larger schools (Pitcher, 1986; Pitcher and Parrish, 1993). Larger schools may be more detectable by predators, with laboratory experiments showing that predators will often attack a larger school of fish (Krause and Godin, 1995; Krause et al., 1998). Although successful predation upon fish schools usually relies on disturbance of the school structure through behavioural adaptations or from environmental factors such as the presence of strong flows (Pavlov and Kasumyan, 2000), targeting areas or times where prey is more easily accessed or localised density is increased (e.g. Waggitt et al., 2018; Couto, 2022).

At tidal stream sites, localised density of fish is thought to increase due to the tidal-coupling hypothesis (Zamon, 2003), whereby energetic sites that create predictable variability in zooplankton distribution, abundance or availability, attract small fish, thereby attracting piscivorous predators. However, we know that top-predators are not necessarily attracted to locations with the highest prey density. It is a combination of factors that influence prey availability; including depth, schooling density and schooling behaviour (i.e., Lieber et al., 2021; 2019; Waggitt et al., 2018; Boyd et al., 2016; 2015; Chimienti et al., 2014). The presence of a MREI slows the main flow of a tidal stream, but with increased flow speed and turbulence around a tidal turbine structure (TTS) and in its wake (Wolf et al., 2021). The physical alteration of flow properties as a result of a TTS has the potential to have significant impacts on forage fish behaviour. Studies have found mixed responses of fish behaviour in response to TTS. In some cases, this is a positive effect, where studies have reported an aggregation effect to a TTS, as well as changes in schooling behaviour that may enhance prey availability or top-predator foraging efficiency (Fraser et al., 2018; Williamson et al, 2019;

Williamson et al., 2021). Whereas other studies have reported significant negative effects of these devices, stimulating far (Viehman and Zydlewski, 2015) and near field (Bevelheimer et al., 2017; Shen et al., 2016; Viehman and Zydlewski, 2015) avoidance of turbine structures by fish schools, which may have negative consequences for top-predator foraging behaviour and local wildlife dynamics. Other evidence of TTS altering forage fish behaviour include; vertical evasion behaviour around the rotator swept height; shift of fish behaviour to tidal phase dependence (Fraser et al., 2018). Discrepancies have been reported between studies, specifically in terms of the aggregating effect (e.g. Bevelheimer et al., 2017; Fraser et al., 2018), whereby some studies reported an increase in abundance of fish in the presence of MREI due to favourable hydrodynamic conditions whereas others reported a decline due to far-field avoidance, indicate a degree of site and species specificity in responses (Williamson et al., 2021). Thus, gaining a better understanding of prey responses to turbines, and changes in hydrodynamics around turbines, would allow us to assess how individual installations, as well as arrays, could alter prey availability, allowing us to gain a better understanding of the indirect impacts of MREI on top-predators.

Recording prey behaviour in tidal-stream environments and around TTS is key if we are to understand the impacts of turbines on prey availability. Field studies have utilised a number of conventional techniques to record fish presence around MREI. Early studies used video-monitoring to record fish interactions with these devices (e.g. Hammer et al., 2013; Broadhurst et al., 2014). However, video-monitoring techniques are confined to periods of good visibility (Chamberlain and Ioannou, 2019; Leahy et al., 2011; Abou-Seedo et al., 1990), whilst also being unable to encompass the diurnal cycle of day and night without the aid of artificial lighting, which may influence fish behaviour (Yoshida et al., 2021). Video monitoring techniques also mean that the effect of decreased visibility, either from turbidity or from the day-night cycle, on the avoidance capabilities of fish cannot be measured. Acoustic monitoring has also been used in field studies to assess the impact of MREI on fish behaviour. However, the high ambient noise present in TSE leads to difficulties in tracking individual fish movements (Fraser et al., 2017) and tracking movements of schools and individuals can be difficult over the entire water column, whilst also providing a relatively narrow sampling window both upstream and downstream of a MREI (Viehman and Zydlewski, 2015). Moreover, acoustic monitoring techniques also prove to be an expensive method of assessing fish behaviour. There is a need to predict the behaviour of fish in the presence of arrays rather than singular installations, as there have currently been no studies published on this. A predictive method, that utilizes empirical information on prey responses to altered hydrodynamics, which reduces the need for challenging, complicated and expensive data collection would prove useful in this rapidly growing field, providing a useful tool in the risk and environmental impact assessment processes.

An agent-based modelling (ABM) approach may address these needs. Agent, or individual-based modelling; a term which is interchangeable in the literature; describes a modelling technique in which Lagrangian points are advected, and tracked, through a programmed environment. The discretised Lagrangian points within an ABM represent individual organisms, which are programmed with a set of individually varying behavioural traits that dictate how an agent will respond and interact with the environment into which it is programmed, as well as with other individuals (McLane et al., 2011). The ability for individual organisms to respond to an environments' dynamism, as well as to other individuals within the model, means that ABM have been a particularly useful tool in ecology (DeAngelis and Mooij, 2005). For reviews on past applications of ABM, see either DeAngelis and Mooij (2005) or McLane et al. (2011). Recently, ABMs have been used to assess the levels of stress exerted on fish populations by underwater noise from construction of a hypothetical marine renewable energy installation (Benson et al., 2016). A more recent study from Benson et al. (2021) highlighted the predictive capabilities of an ABM methodology, where field trap data of juvenile eel populations were systematically compared with the model outputs to determine key behaviours in the selective tidal stream transport of agents (eels). The parametrised model was then applied to a second case study to predict entrainment rates in a power plant's cooling water intake and outfall, showing that an ABM can accurately predict juvenile eel movements when migrating upstream, which can potentially be used to mitigate anthropogenic impacts against this species in the future. The author also stated that this methodology could be applied to various other fish species in further studies.

This study develops an ABM for fish schools around tidal turbines, which represents a potentially useful approach for assessing the impacts of MREI or arrays on prey availability to top-predators. Agents within the model are programmed into a dynamic environment, allowing individual responses from changing physical properties of the water column to be measured. By using an ABM approach, this study will (1) assess the effect of a MREI on the hydrodynamic regime of a tidal stream environment and how alterations in hydrodynamics will affect 3D fish behaviour and distribution – within the bounds of model parameter uncertainties (i.e. a "so-called sensitivity test"). (2) Highlight areas where the behaviour of the fish may increase top-predator foraging efficiency and thus (3) predict how foraging opportunities for top-predators may be affected in the presence of a MREI. Specifically, we will model 3D fish distribution and schooling capabilities of fish in the wake of a MREI and how these change over 24 hours, encompassing the semi-diurnal tidal phase and the day-night cycle. Once the ABM has been validated, it can be re-parametrised under a multitude of different scenarios, including looking at impacts of arrays rather than a singular device; how impacts may change depending on turbine design; the method can also be applied to sites earmarked for TT installation or for use on currently deployed tidal turbines to gain an insight into their ecological impact. Providing a more accurate assessment of risk for legislators and planners that can be applied on a site-by-site basis.

Methods

2.1 Model Setup

2.1.1 Basics

HydroBoids is an ABM developed by Thomas Benson (Benson et al., 2016; Benson et al., 2021) that aims to simulate 3D movement of individuals, or agents. This is achieved by a set of simple behavioural traits that dictate how an individual, or agent will move throughout the 3D environment into which it is programmed. Agents move throughout the model using a correlated random walk algorithm (Codling et al., 2008; Willis, 2011), aiming to mimic how fish swim through their environment. Details regarding how the displacement of each agent is tracked, can be found in Benson et al. (2016) and Benson et al. (2021).

2.1.2 Environment

The model environment was programmed using TELEMAC 3D hydrodynamic software with a domain of 2300m x 1400m. The water depth of the model ranged from 42.4m to37.6 m above the seabed. This is based on a site nearby the Apapa shipwreck on the Anglesey coastline with a flattened bathymetry. Although this can be altered in future studies to reflect a specific site of interest. The TELEMAC model consisted of a vertical mesh has been discretised using 11 planes using 2 sigma grid separated at a fixed elevation of -16m. The bottom 7 planes are sigma grid up to -16m, and the top 4 planes are sigma grid above that. This ensures the structure does not get affected by the tidal surface elevation which causes the sigma plane spacing to vary through the tide.

Maximum horizontal current speed was 4.0 ms^{-1}, with typical values around the turbine ranging from 0.0 ms^{-1} to 2.6 ms^{-1}. This coincides with typical 1st generation tidal-stream energy technology that require peak flows exceeding 2.5 ms^{-1}, in a water depth between 25 and 50m (Iyer et al., 2013). The model domain included a TTS, a bottom-mounted monopile, of dimensions 5m diameter and 20m tall, having dimensions of a typical 1st generation tidal-stream-energy project (Lewis et al., 2015). Due to constraints on time in this project, the model domain does not include a rotor blade, however more complex structures could be accommodated into the model environment in the future.

2.1.3 Fish

At time=0, agents are placed into the model as discretised points in 3D space. Fish are originally initialised in 11 separate schools distributed randomly throughout the model environment. School size ranges from 4 to 483 individuals, with densities ranging from 2.82 individuals per m^3 to 93.80 individuals per m^3, taken from the literature to provide realistic schooling densities of fish in tidal streams from the outset of the model run (Pitcher and Partridge, 1979; Misund, 1993). The initial heading of the agents is randomly selected between 0 and 360 and the elevation angles are initialised to zero. Agent heading, position and speed was recorded from 1200s, and after this every 60s. Agents were recorded starting during flood tide just after slack-water. Allowing the agents to react to their initialisation settings before data is recorded for post-processing and analyses. Fish characteristics for the model inputs were selected based on observations in the literature about the behaviour of saithe, *Pollachius virens.* P.virens is a common species in Northern Scotland, where the majority of UK-based tidal stream developments occur (Wolf et al., 2021). Saithe is also a common prey species for top-predators, such as for European shags, *Phalacrocorax aristotelis,* in Norway (Barrett et al., 1990; Barrett, 1991) as well as contributing a significant portion to the diet of common guillemots, *Uria aalge*, razorbills, *Alca torda,* and black guillemots, Cepphus grylle (Anker-Nilssen, 2010).

2.2 Analysis

2.2.1 School Metrics

Schools of fish were identified using distance-based clustering from Marcon (2022). Where schools of fish were defined as a minimum of 3 individuals, within 6m of one another. This was taken from Williamson et al. (2017) in which they aggregated targets within 6m of each other to define a school. This value was then tested against visual inspection of schools to determine its accuracy in selecting all the individuals that are schooling within the model. Schools of fish were distinguished from one another in the XY plain, a method which was verified through visual inspection. Following the identification of schools, several measurements of school distribution and behaviour were extracted from ABM outputs: (1) The schooling density of fish represented by the number of fish in a school (n) divided by the volume (m^3) of the school. (2) The depth of schools represented by their height above the seabed (m) (hab). (3) Number of schools present. Density, as well as other schooling characteristics, such as, school depth, n and number of schools were calculated every 60 minutes for post-processing speed, providing 24 measurements of schooling characteristics throughout the model period.

2.2.2 Sensitivity Testing

Sensitivity testing of the model was performed to identify the effect of changing the model inputs, outlined in Table 1, on the model outputs mentioned above. A baseline scenario was created with parameters outlined in table 1. From this, some parameters were modified, as per table 2, to create scenarios with different input parameters to assess their effect on various response variables. The response variables focused on during sensitivity testing were schooling density of fish (nm^3), mean height above seabed (hab) of all fish within the model (m) and the number of fish within close association of the turbine structure (N). To assess N, a 100x50x20m area was created with the turbine in the centre and the number of individuals entering this area was recorded every 60s, hab was also recorded every 60s. Scenarios were compared to one another using a one-way ANOVA to find significant differences ($p < 0.05$) in (1) schooling density of fish (2) mean height above the seabed of all fish within the model (3) number of fish within close association with the turbine structures, as well as assessing if changing these parameters had any effect on the temporal pattern of this relationship. Then, a moving means analysis was conducted with a sampling window of 3 time-steps for all the different parameters to assess how the various model outputs changed over time, in comparison to other scenarios. All analysis assumed a Gaussian distribution. Results were then compared to the literature, to find the values of parameters which produced the most realistic results (table 3). For analyses, each of the response variables (density, N, and hab) was averaged every 60 minutes to improve computing efficiency. During sensitivity testing only, the mean values for each schooling metric was compared between different scenarios. Values for tidal height were obtained from the TELEMAC results file.

2.2.3 Final model runs

Input parameters obtained from the sensitivity testing of the model were used in the final model runs of the study. These were used to parametrise the model to compare the behaviour and distribution of fish when the turbine structure was removed from the model environment (table 3). A moving means analysis, with a sampling window of 3, which averages the response variables between each sampling window (60mins), was conducted to assess the effect of a tidal turbine structure on the distribution and behaviour of fish over a 24hr period.

Table 1: The baseline ABM parameters and their description. Values at this stage were selected randomly unless otherwise specified.

Parameter	Description	Value
Schooling Probability	Schooling behaviour of fish in the model was programmed as a likelihood of schooling, and given a value between 1 and 0. If this equals 1, fish all fish within the model will attempt to school at every time step, effectively meaning that the agents will only focusing on schooling and will not, for example, avoid areas of strong flow. If set at 0.25 for example, an agent will undertake schooling behaviour at every	0.10

	4th time step, or 25% of the fish will undertake schooling behaviour at every time step, on average.	
Response Probability	Response probability is a parameter that controls the response behaviour of individuals to changing velocity magnitudes of the model. Like schooling probability, it is a value between 0 and 1. It determines how likely an individual is to respond to a changing Velocity Magnitude at each time step.	0.10
Navigation Probability	Navigation probability controls how quickly fish will return to their normal swimming behaviour after being stimulated by stimulus. Including schooling behaviour and changing velocity magnitudes. Also a value between 0 and 1, the higher the value, the quicker fish will return to normal swimming behaviour after being stimulated.	0.05
Velocity Magnitude Threshold (VMT)	This sets a threshold above which agents will begin to swim against the flow or find refuge. Below this, agents will swim freely throughout the model environment. Although their speed is limited to imitate energy saving behaviour and prevent fish from swimming their maximum speed when flow speed is reduced, which was deemed to be biologically inappropriate.	0.5 (ms^{-1}) from (Liao, 2007; references within), where they highlighted changes in behaviour above this value during laboratory experiments.
Swim Speed	Agents are assigned swimming speeds randomly selected from a normal distribution of speeds with maximum and minimum	0.51-1.50 ms^{-1} from He and Wardles calculation (1988), body lengths taken from Lorentsen et al., (2004) of 0 to

	values obtained from He and Wardle's (1988) calculation relating body length to swim speed of saithe.	year 3 age classes of *Pollachius virens.* These were selected as these age groups are the most important items of prey for top-predators. Values were extended beyond this for sensitivity testing to assess how faster swimming fish may be affected.
School Separation	Fish are also assigned a maximum and minimum value of school separation within the model. Below the minimum, fish will steer away from one another to avoid crowding, and above this fish will effectively be invisible to one another.	Min: 0.25m ; Max: 6m Max taken from Williamson et al. (2017) where they aggregated schools of fish within 6m of each other. Minimum value taken from average packing densities of fish from Pitcher and Partridge (1979).

Table 2. Details of scenarios with modified input parameters. For each scenario, the parameters not being modified will have values from the baseline scenario, detailed in table 1.

Parameter being modified	Values
Schooling Probability	0.00; 0.01 ; 0.025 ; 0.05 ; 0.125 ; 0.25 ; 0.50 ; 0.75
Response Probability	0.00; 0.01 ; 0.025 ; 0.05 ; 0.125 ; 0.25 ; 0.50 ; 0.75
Navigation Probability	0.00; 0.01 ; 0.025 ; 0.05 ; 0.125 ; 0.25 ; 0.50 ; 0.75
Velocity Magnitude Threshold (ms^{-1}) (VMT)	0.5; 1.0 ; 1.5 ; 2.0
Swim Speed (ms^{-1})	0.50-0.80; 0.81-1.00; 1.01-1.50; 1.50-2.50; 2.50–3.50

3: Results

3.1 Sensitivity Testing

3.1.1 Schooling Probability

Schooling densities of fish were significantly higher when the probability of schooling was higher ($F_{(7,183)}$=52.02, p <0.01) (figure 1). This relationship also appeared exponential and substantial, with mean densities of > 50 individuals per nm^{-3} at 50 and 75% schooling probability compared to <15 individuals per nm^{-3} at < 50% schooling probability. Figure 2 (note the differing y-axis range between plots) shows that there are inconsistent patterns in changing density over time between different scenarios. Generally, however, a trend in higher schooling densities around high tide, can be observed, with scenarios that have similar schooling probabilities, showing similar general trends. The magnitude in changing densities is also not consistent, when schooling probability is greater than 0.50, we see a significant increase in the observed schooling densities of fish.

Height above seabed was significantly higher when the probability of schooling was higher ($F_{(7,11360)}$=1958.32, p <0.01) (figure 3), significantly different results occurred when schooling probability was less than 0.05 compared to when it was greater than 0.50. Thus, schooling probability will have an impact on the ABM result if the parameter is unknown. Looking at figure 4, we can see that in scenarios with a higher schooling probability, fish generally swam higher in the water column, indicating that fish that are able to form denser schools, will swim closer to the surface. We also see a general pattern of fish rising in the water column around high slack tide, before sinking again during peak flows.

There are significantly fewer fish that come into close association with the turbine when schooling probability is higher ($F_{(7,11360)}$=26.37, p<0.01) (figure 5). When schooling probability is higher, and there a fewer but more dense schools, there are less individuals that come into close association with the turbine than when schooling probability is lower and there are more, but more sparsely populated schools. Looking at figure 6, we see 3 distinct periods when there is a significant spike in N in multiple scenarios. This is associated with reduced current speeds around slack-water. Outside of these periods however, N remains consistently low.

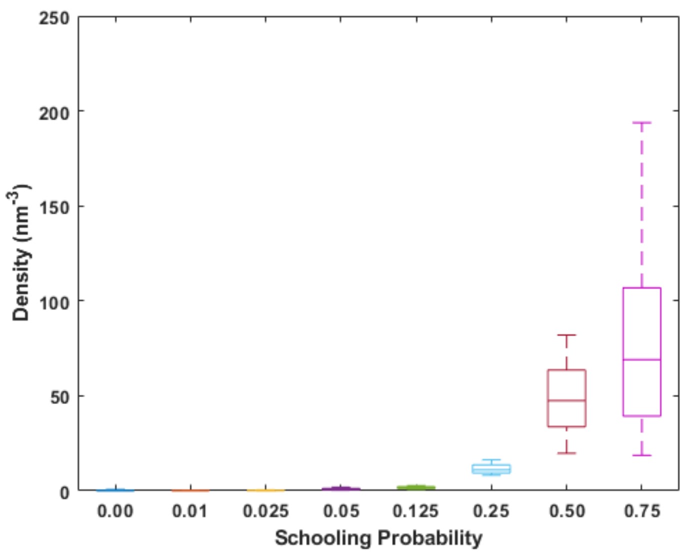

Figure 1. Boxplot showing how densities (nm^3) of schooling fish was affected by changing schooling probability within the model.

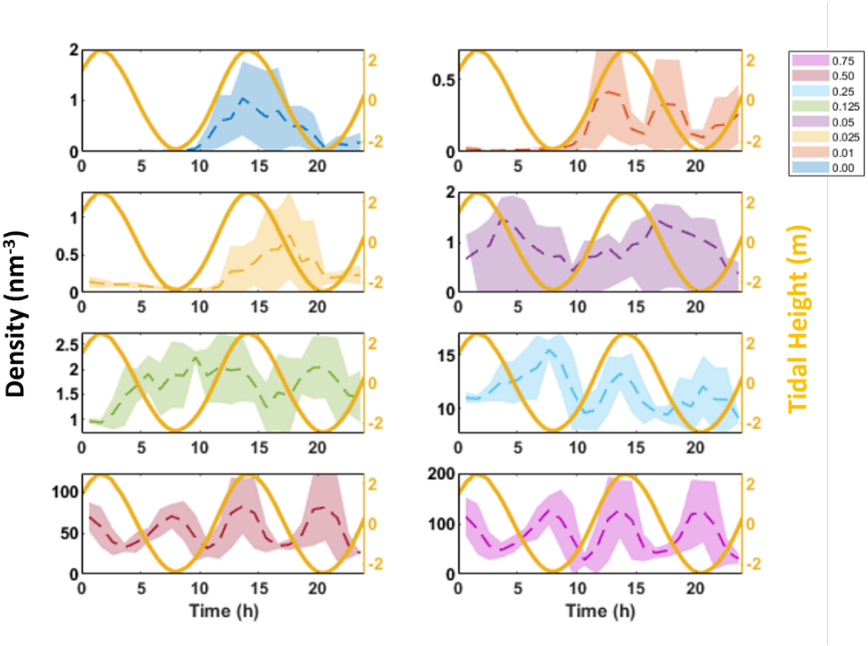

Figure 2. Effect of schooling probability as a function of time on density (nm^3) of all fish. Schooling probabilities shown on figure legend. Tidal height (m) shown on y_2 axis.

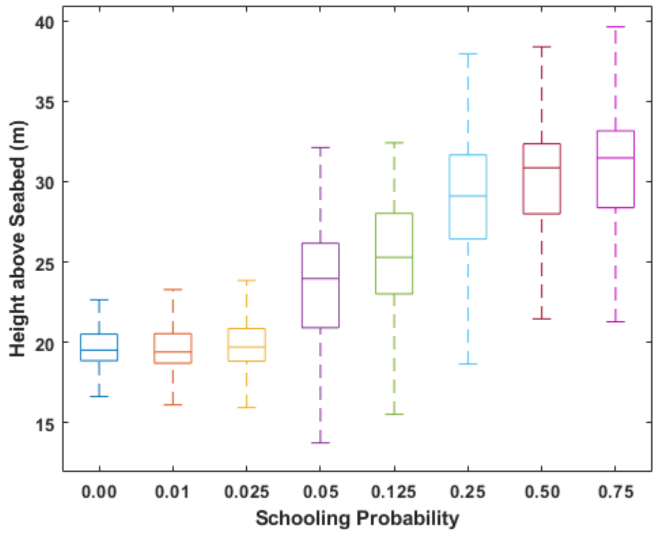

Figure 3. Effect of schooling probability on height above seabed (m) of all the fish within the model.

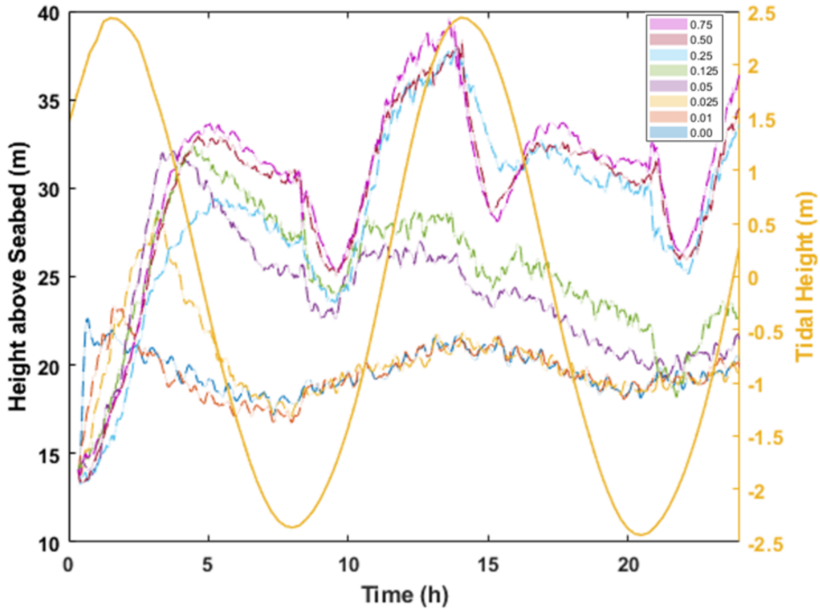

Figure 4. Effect of schooling probability as a function of time on hab (m) of all fish. Schooling probabilities shown on figure legend. Tidal height shown on y_2 axis.

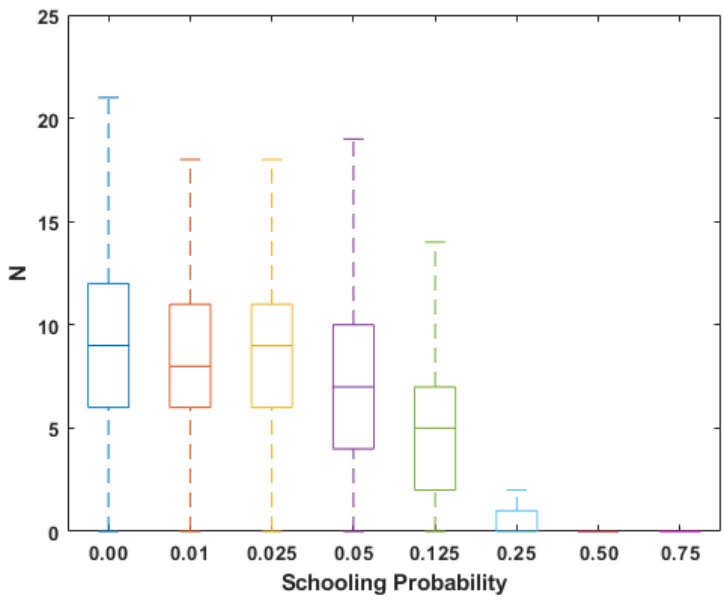

Figure 5. Effect of schooling probability on the number of individuals (N) within close association with the turbine.

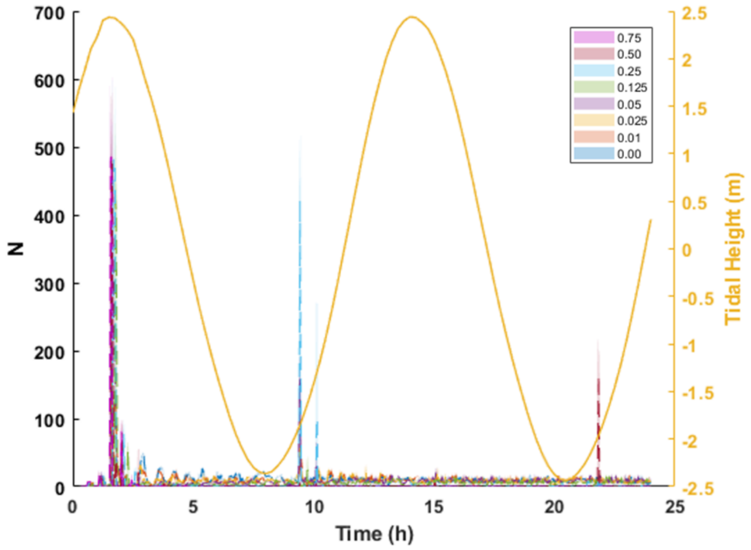

Figure 6. Effect of time and schooling probability on the number of individuals (N) within close association with the turbine. Schooling probabilities shown in figure legend. Tidal height (m) shown on y_2 axis.

3.1.2 Navigation Probability

There is a significant difference in schooling densities of fish as a result of changing Navigation probability ($F_{(7,184)}$=8.33, p>0.01). However, the significant difference only occurs when Navigation Probability is < 0.01 (figure 7). In general, a peak in density is experienced in the last low tide throughout all scenarios (figure 8).

Navigation Probability only had a significant effect on the hab of fish within the model when scenarios had a probability greater than 0 ($F_{(7, 11360)}$=974.42, p<0.01), otherwise, there was no significant difference between scenarios (figure 9). Excluding the scenario where p =0.00, fish showed a general rise in the water column around slack tides, before sinking again during peak flows (figure 10).

When Navigation probability was switched-on in the model (>0.00), there was a significant difference in N in the model ($F_{(7,11360)}$=92.58, p<0.01). Otherwise, changing Navigation probability had no effect on N (figure 11). N remained low consistently during the model period, except when p = 0.00 and during times when flow speed would be reduced (at high tide and just after low tide).

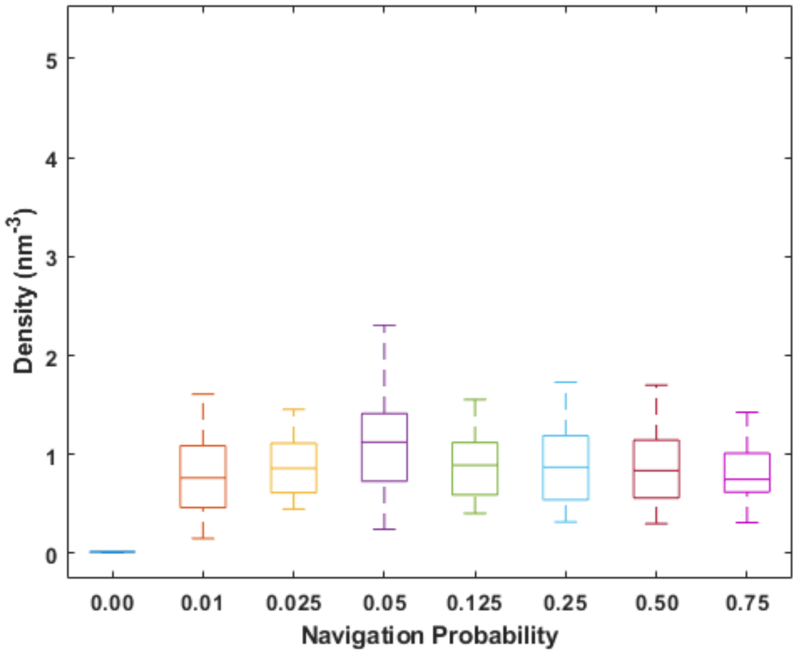

Figure 7. Effect of changing Navigation probability on the density (nm^3) of schooling fish.

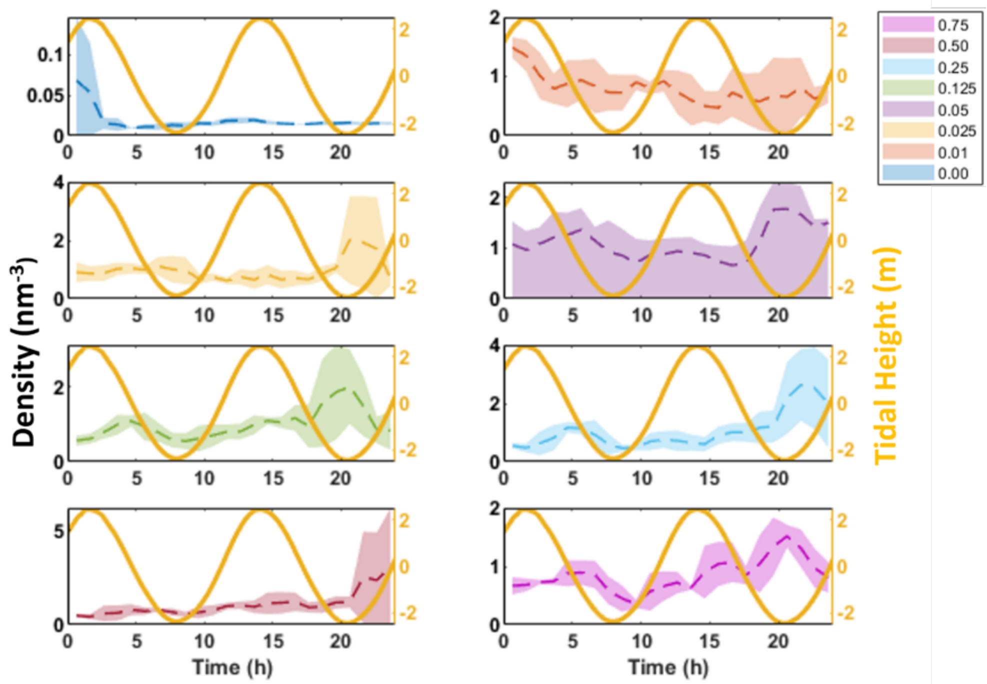

Figure 8. Effect of Navigation Probability on the density (nm^3) of schools as a function of time. Legend showing the parametrised Navigation Probability. Tidal height (m) shown on y_2 axis.

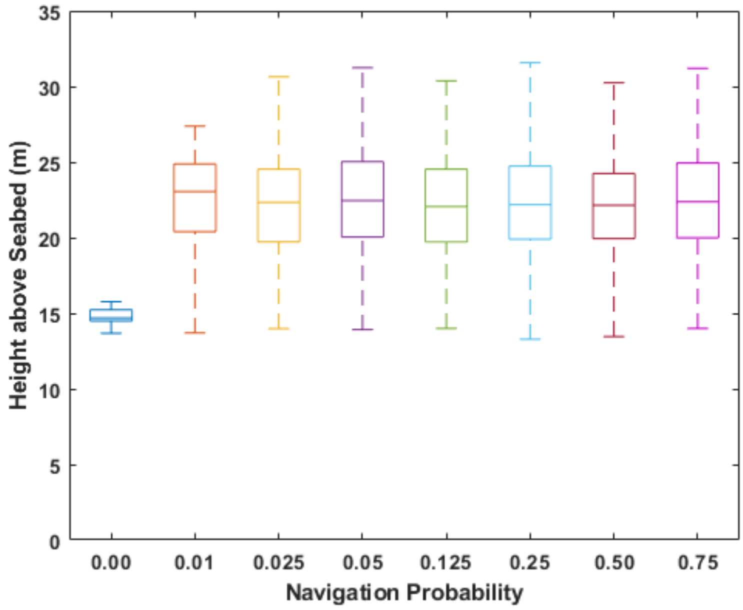

Figure 9. Effect of Navigation Probability on the height above seabed (m) of all fish within the model.

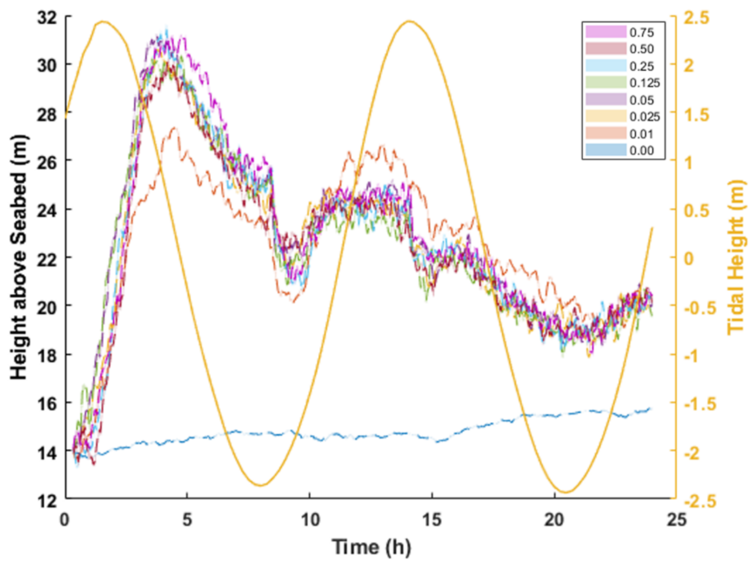

Figure 10. Effect of Navigation Probability on the height above seabed (m) of all fish within the model as a function of time. Legend showing the parametrised Navigation Probability. Tidal height (m) shown on y_2 axis.

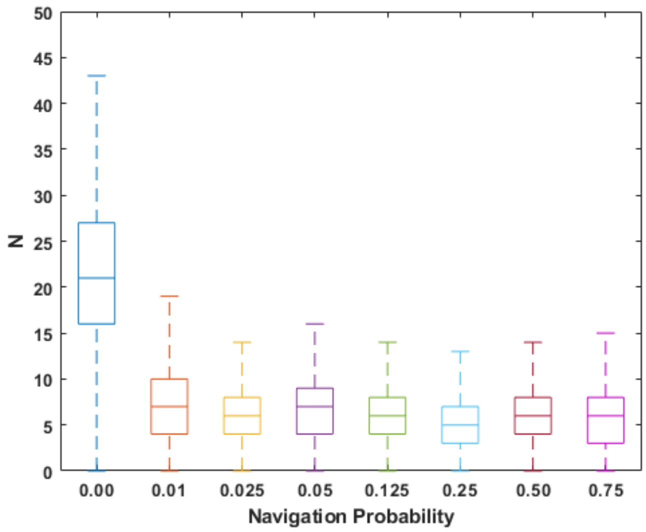

Figure 11. Effect of navigation probability on the number of individuals (N) within close association with the turbine structure.

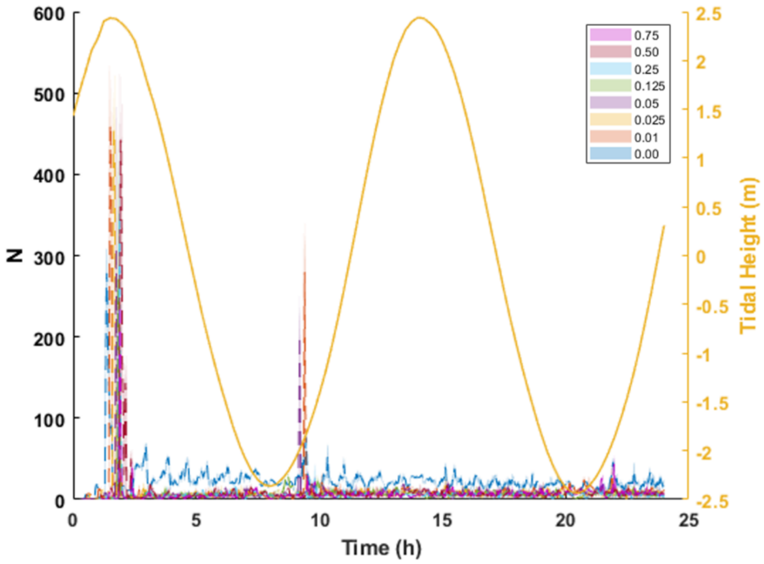

Figure 12. Effect of Navigation probability on the number of individuals (N) in close association with the turbine as a function of time. Legend showing the parametrised Navigation Probability. Tidal height (m) shown on y_2 axis.

3.1.3 Response Probability

Changing response probability has a significant effect on the schooling density of fish ($F_{(7,184)}=13.26$, $p<0.01$) but there is no obvious trend in the magnitude of this effect between scenarios (figure 13). Peaks in density occurred around low tide in all the scenarios, with some also experiencing peaks in density around high tide; corresponding to times when flow speed is reduced (figure 14).

Response probability had a significant effect on hab of all fish in the model ($F_{(7, 11360)}$=584.28, p<0.01). This effect was more profound at lower schooling densities (Response Probability < 0.05), with there being no significant effect of response probability on hab when Response probability is greater than 0.05 (figure 15). Hab increases throughout all scenarios around high tide, before decreasing again in correspondence to increasing flow speeds (figure 16). Although the hab is different between scenarios, the difference is negligible and could be a result of variation within the model, rather than a meaningful trend.

Response probability has a significant effect on the number of fish within close association with the turbine structure ($F_{(7,11360)}$=8.72, p<0.01), scenarios of a higher Response Probability generally showing an increase in N (figure 17). There was an increase in N occurring around high tide in all scenarios but remained consistently low between these periods (figure 18).

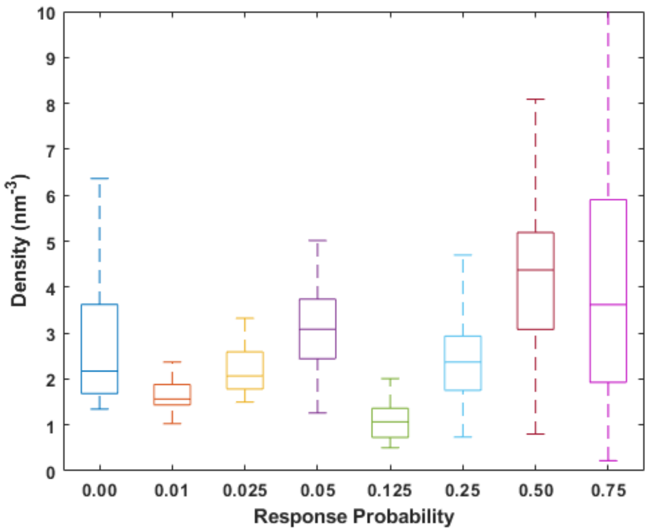

Figure 13. Effect of Response Probability on the average schooling density of fish (nm^3).

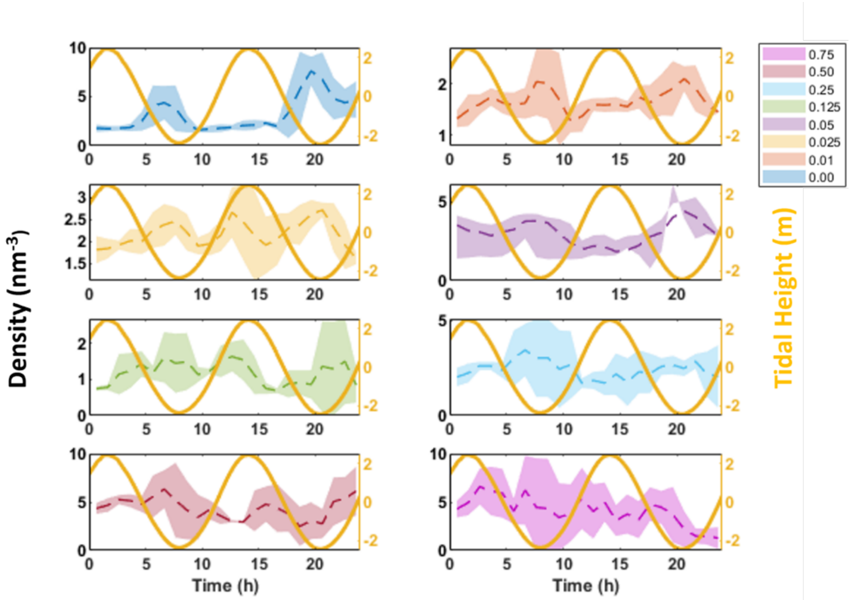

Figure 14. Effect of changing Response Probability on the temporal patterns of schooling density. Legend showing the parametrised Response Probability. Tidal height (m) shown on y_2 axis.

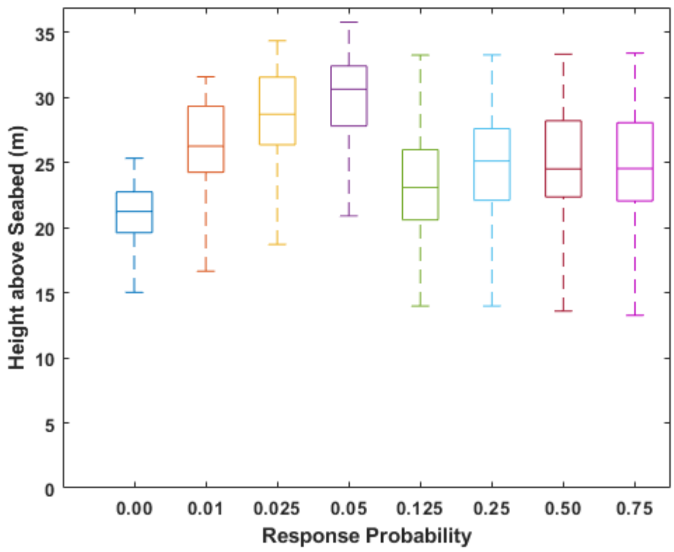

Figure 15. Effect of response probability on the mean height above seabed (m) of all fish within the model.

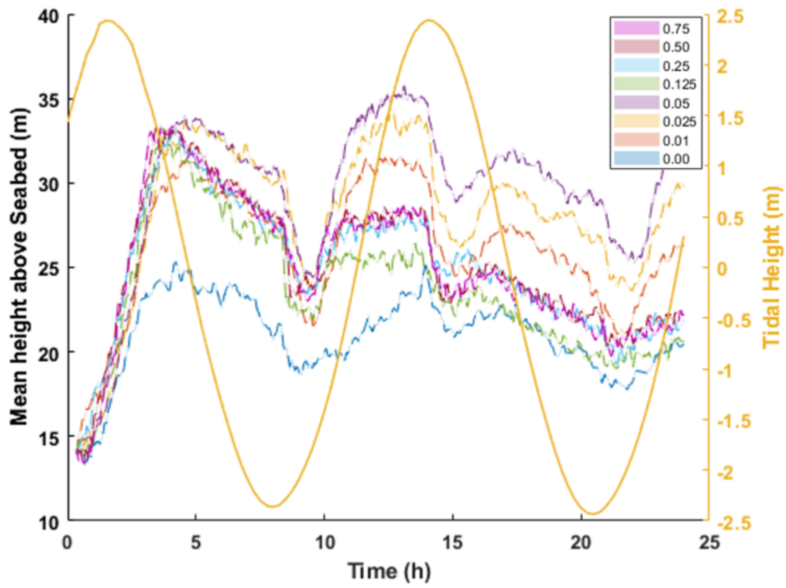

Figure 16. Effect of changing Response probability on the relationship between mean height above seabed (m) and time. Legend showing the parametrised Response Probability. Tidal height (m) shown on y_2 axis.

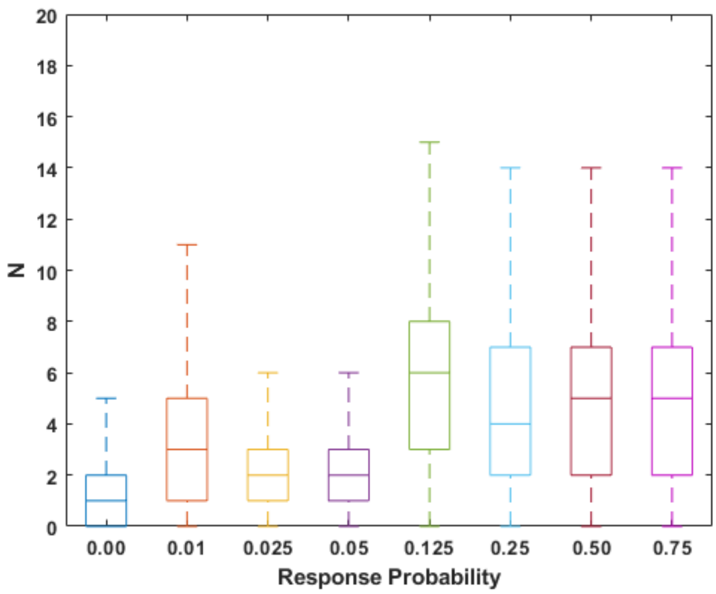

Figure 17. Effect of changing response probability on the number of individuals (N) within close association to the turbine.

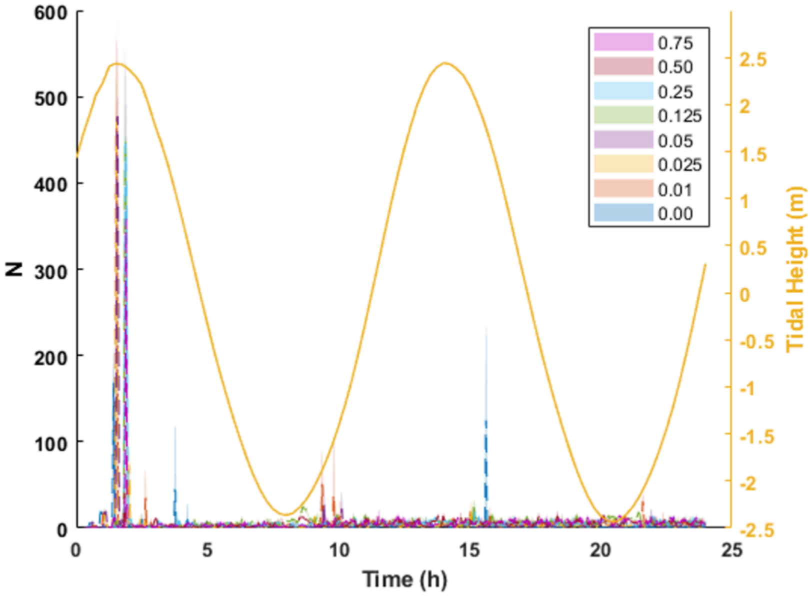

Figure 18. Effect of changing Response probability on the relationship between number of fish (N) in close association with the turbine and time. Legend showing the parametrised Response Probability. Tidal height (m) shown on y_2 axis.

3.1.4 Velocity Magnitude Threshold

Changing VMT had no effect on the schooling densities of fish (figure 19) ($F_{(3,92)}=2.48$, $p=0.066$). There was no clear relationship between density and time between different scenarios, although peak abundances were recorded when approaching low tide in three of the scenarios (figure 20).

Changing VMT had a significant effect on mean hab of all fish within the model ($F_{(3,5680)}=1046.89$, $p<0.01$). When VMT ≥ 1.50 ms^{-1}, mean hab was significantly lower than when VMT≤ 1.00 ms^{-1} (figure 21). All scenarios show the same trend of rising in the water column around high tide, as well as rising in the water column during the first flood tide, when flow speed is at its highest (figure 22).

VMT had no effect on N (figure 23) ($F_{(3, 5680)}=1.12$, $p=0.34$). All scenarios show a general trend of reduced activity around the turbine structure during times of peak flows (figure 24), spikes in abundance occurred around these structures usually occurring only briefly around periods of reduced flow speeds.

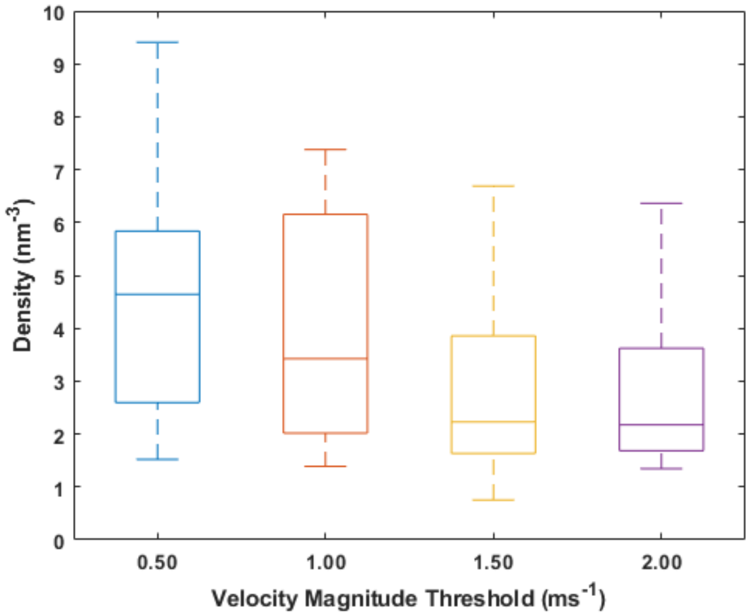

Figure 19. Effect of changing Velocity Magnitude threshold (ms^{-1}) on schooling density (nm^{-3}) of fish.

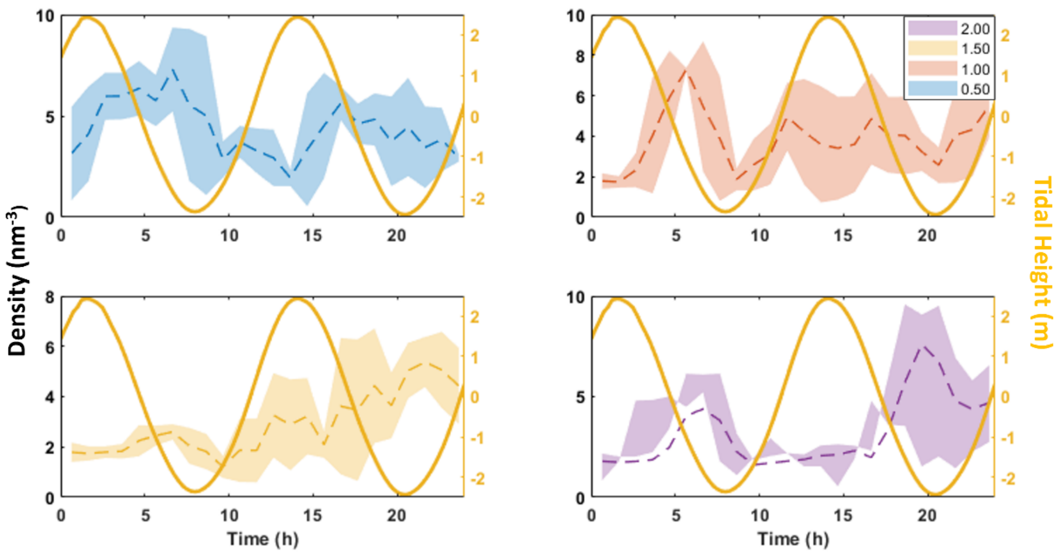

Figure 20. The effect of changing VMT (ms^{-1}) on the density(nm^3) of schooling fish over time. Legend showing VMT (ms^{-1}). Tidal height (m) shown on y_2 axis.

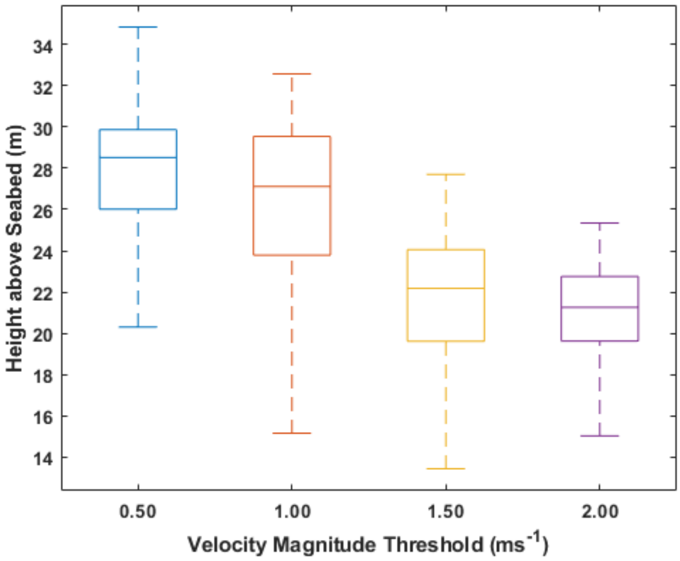

Figure 21. The effect of changing VMT (ms^{-1}) on the mean height above seabed (m) of all fish within the model.

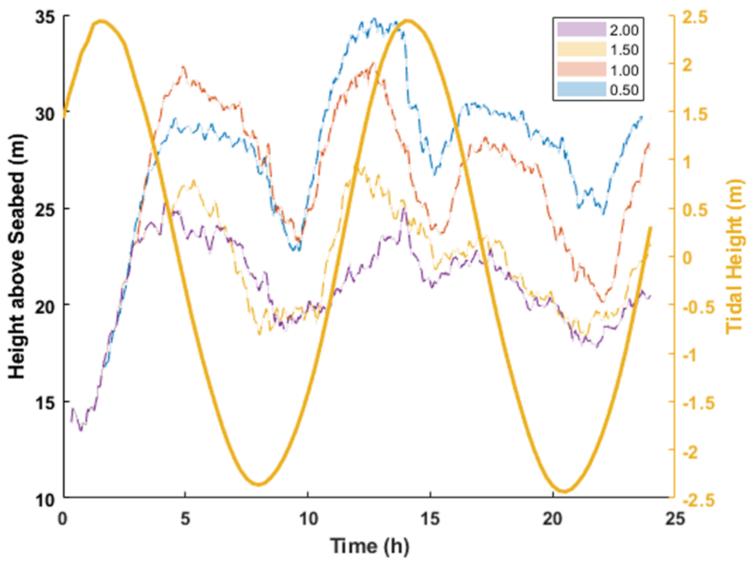

Figure 22. Effect of changing VMT (ms^{-1}) on the relationship between mean height above seabed (m) and time. Legend showing VMT (ms^{-1}). Tidal height (m) shown on y_2 axis.

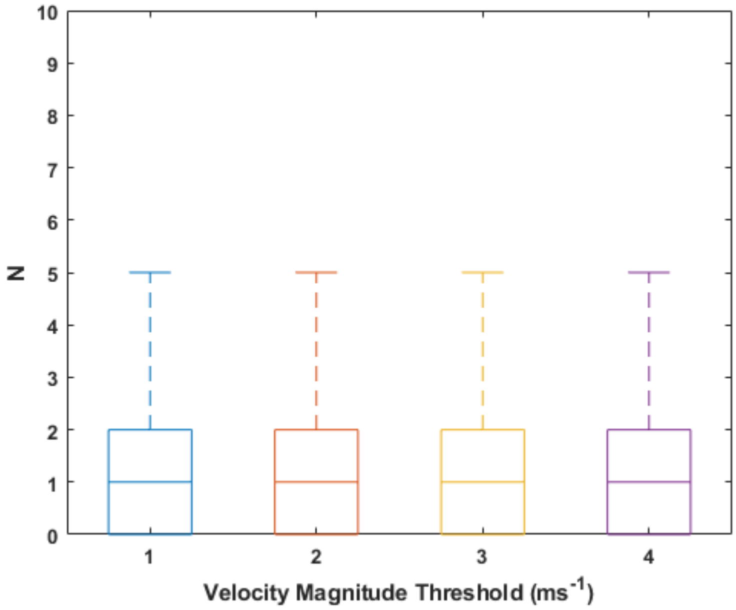

Figure 23. The effect of changing VMT on the number of fish (N) within close association with the turbine.

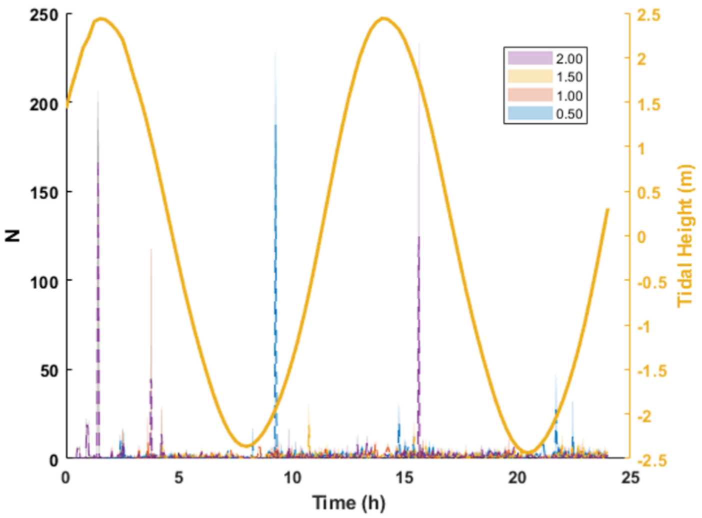

Figure 24. The effect of changing VMT (ms^{-1}) on the number of individuals (N) within close association of the turbine. Legend showing VMT (ms^{-1}).

3.1.5 Swim Speed

There was no significant difference in density when swim speed was altered between scenarios (figure 25) ($F_{(4, 52)}$=0.91, p = 0.46). There were no consistent patterns observed between density over time between scenarios where fish had different swim speeds (figure 26).

Swim speed had no significant effect on the mean height above the seabed of all fish within the model (figure 27) ($F_{(4, 7100)}$=1.51, p=0.20). Hab changed little throughout the model run and between scenarios, but a small degree of rising in the water column was observed throughout the scenarios during high tide (figure 28).

Scenarios in which fish were programmed with faster swimming speeds, showed a significant decrease in the number of fish within close association to the turbine structure (figure 29) ($F_{(4, 7100)}$=112.91, p<0.01). From figure 30 we see that there is no clear difference in trends between scenarios over time. A peak at the first high tide is observed, but this could be due to the direction of the current and where the fish are released in the model.

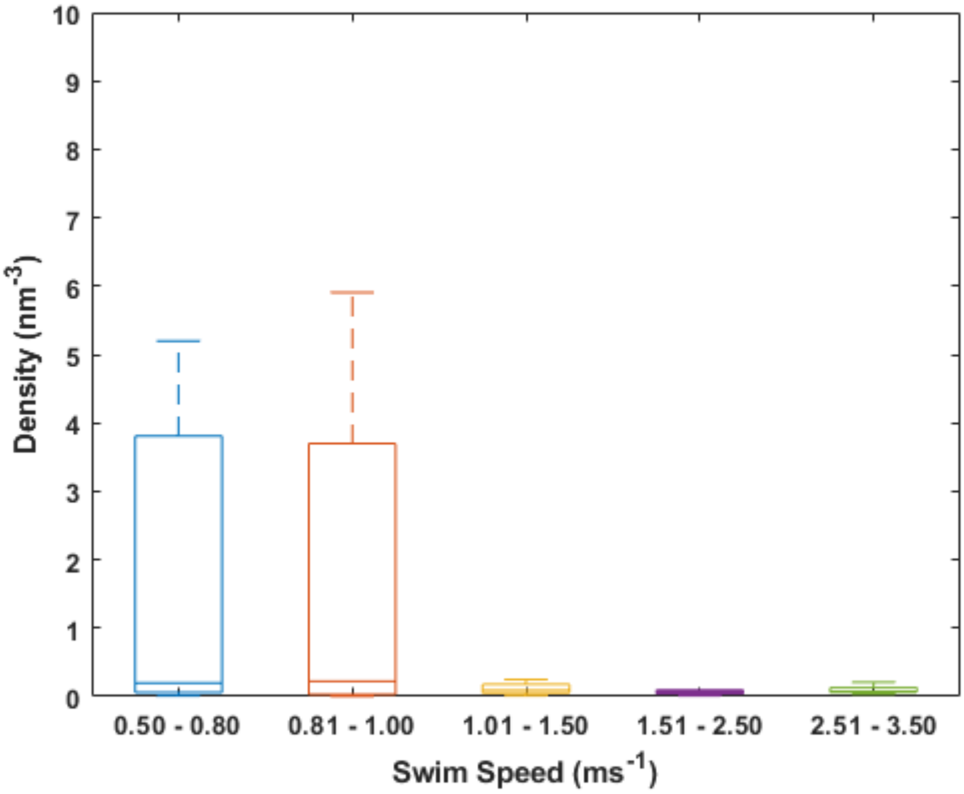

Figure 25. The effect of increasing swim speed (ms^{-1}) on schooling density (nm^3).

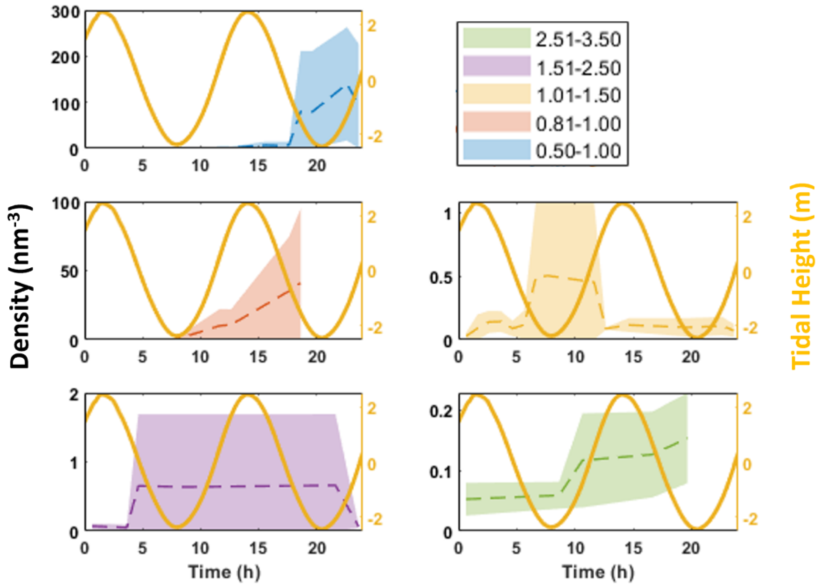

Figure 26. The effect of changing swimming speed (ms^{-1}) on schooling density (nm^{3}) over time. Swim Speed (ms^{-1}) shown in figure legend. Tidal height (m) shown on y_2 axis.

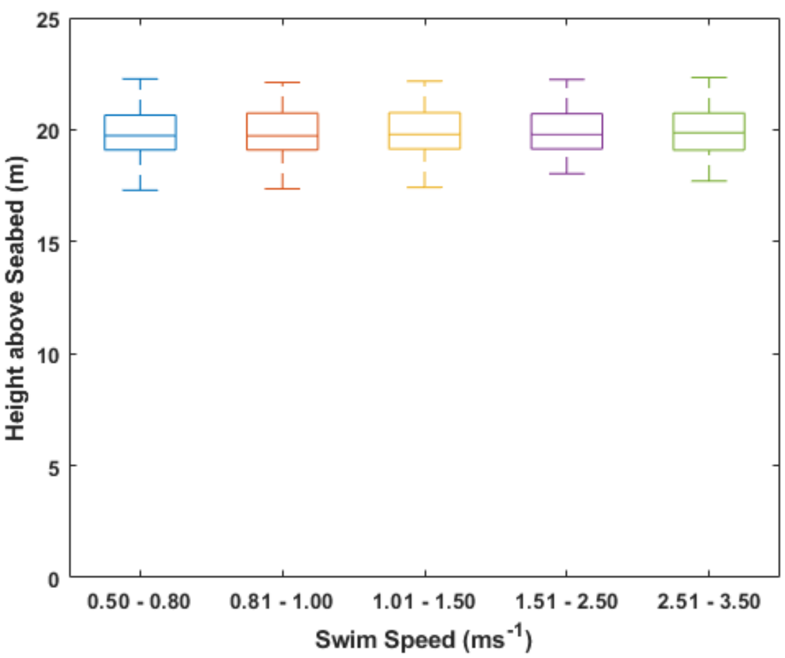

Figure 27. The effect of swim speed (ms^{-1}) on mean height above seabed (m) of all fish within the model.

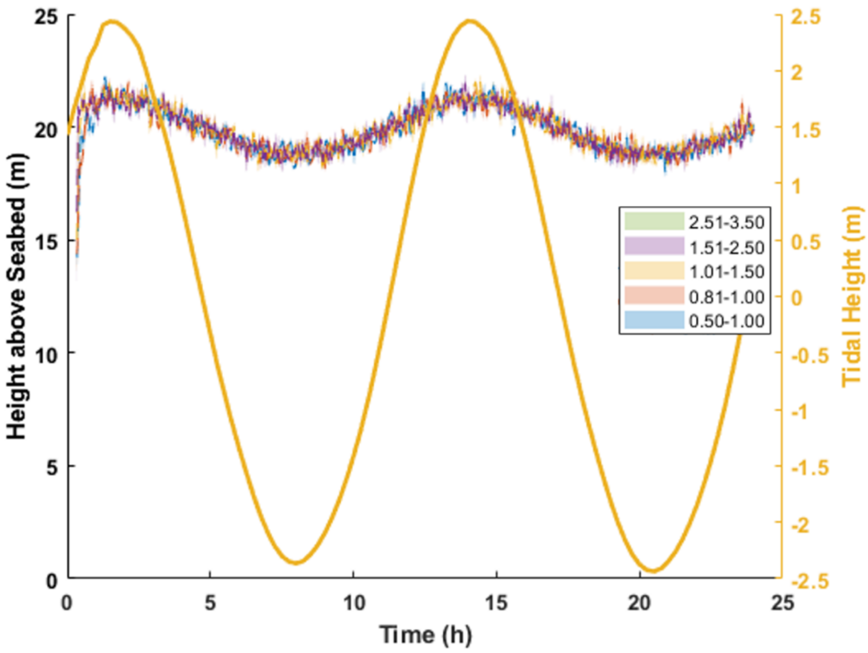

Figure 28. The effect of changing swim speed (ms^{-1}) on the relationship between mean height above seabed (m) and time. Swim speed (ms^{-1}) shown on figure legend. Tidal height (m) shown on y_2 axis.

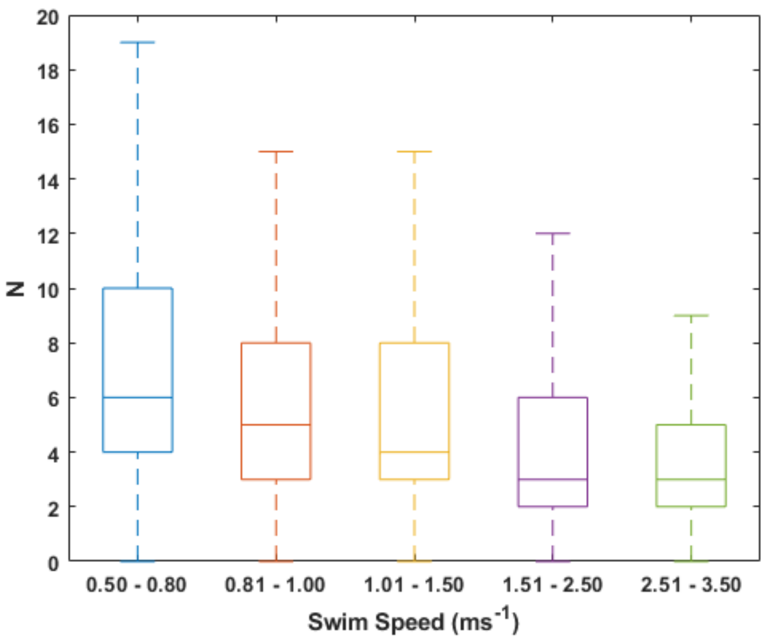

Figure 29. The effect of changing swim speed (ms^{-1}) on the number of fish (N) within close association with the turbine.

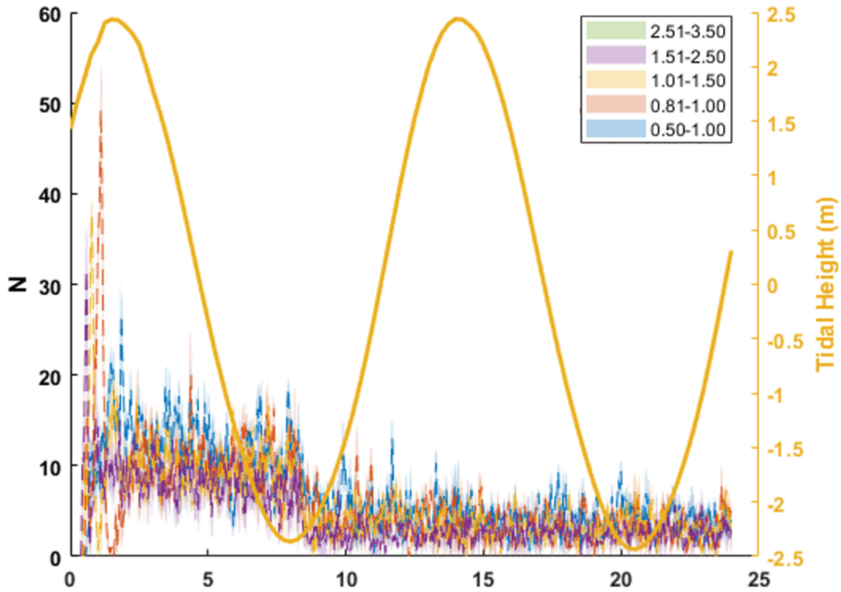

Figure 30. The effect of changing swim speed (ms^{-1}) on the temporal relationship of the number of fish (N) that come into close proximity to the turbine structure. Swim speed (ms^{-1}) shown on figure legend. Tidal height (m) shown on y_2 axis.

Table 3. Input parameter values obtained from sensitivity testing of the model to be used in assessing the effect of Tidal Turbine Structure (TTS) on fish behaviour and distribution (i.e., parameters to be used during the final model runs).

Parameter	Value	Description
Schooling Probability	0.18	This is a value where avoidance behaviour of the turbine structure will still occur, but not to a level where it may become biologically inappropriate, like that of a higher Schooling Probability. This value is also consistent with Pitcher and Partridge (1979), where they reported an average schooling density of 31 nm^3, with a schooling probability between 0.125 and 0.25, producing densities most similar to that (figure 1).

Navigation Probability	0.05	This value produces behaviours consistent with the literature (increased densities during slack water, hab varying with flow speeds and a decrease in N as flow speed increases).
Response Probability	0.125	This value produces station-holding behaviour in the wake of the turbine, a behaviour which is consistent with the literature, yet a value higher than this appeared to have little effect on the response variable, and would not be weighted higher than schooling probability.
Velocity Magnitude Threshold (VMT)	1.0 ms^{-1}	This value is consistent with the literature in that fish will experience behavioural changes to velocity magnitudes greater than 1 ms^{-1} (Liao, 2007; references within). A higher VMT constricted the ability of fish to form denser schools as fish tend to spend more time swimming freely in the model. This was deemed to be unrealistic.
Swim Speed	0.5-1.5 ms^{-1}	Retrieved from Pitcher and Partridge (1979) and Håvard (2004), this is the average swimming speed of year class 0-2 Saithe. Selected as Saithe are common in tidal stream environments and year class 0-2 as these are important food sources for top-predators.

3.2 Final Model Runs

The final model runs of this study were ran using the parameters outlined in table 3 to assess the impact that a TTS had on fish behaviour within the model environment.

There was no significant difference between schooling densities when the turbine was removed from the model environment (figure 31) ($F_{(1,46)}$=0.01, p = 0.91). In both scenarios, there was little fluctuation in schooling density until the last ebb tide of the model run (figure 34a), when schooling density increased in both scenarios corresponding to a reduction in flow speed.

The presence of a turbine structure had no effect on the schooling height of fish (figure 32) ($F_{(1, 13506)}$=0.44, p =0.51). The vertical distribution of fish in the water column relating to tidal stage was inconsistent, first rising when current speed started to increase, before quickly sinking again, before once again rising in the water column as a result of increased current speed (figure 34b). This relationship was unaffected by the presence of a turbine structure (figure 34b).

Significantly fewer fish entered the area around the turbine when a turbine structure was present (figure 33) ($F_{(1, 2840)}$=60.39, p<0.01). More fish entered this area during high flows when the turbine was not present compared to when it was (figure 34c). The peak in N at the start of the model run corresponds to the fish being released and passing through the area.

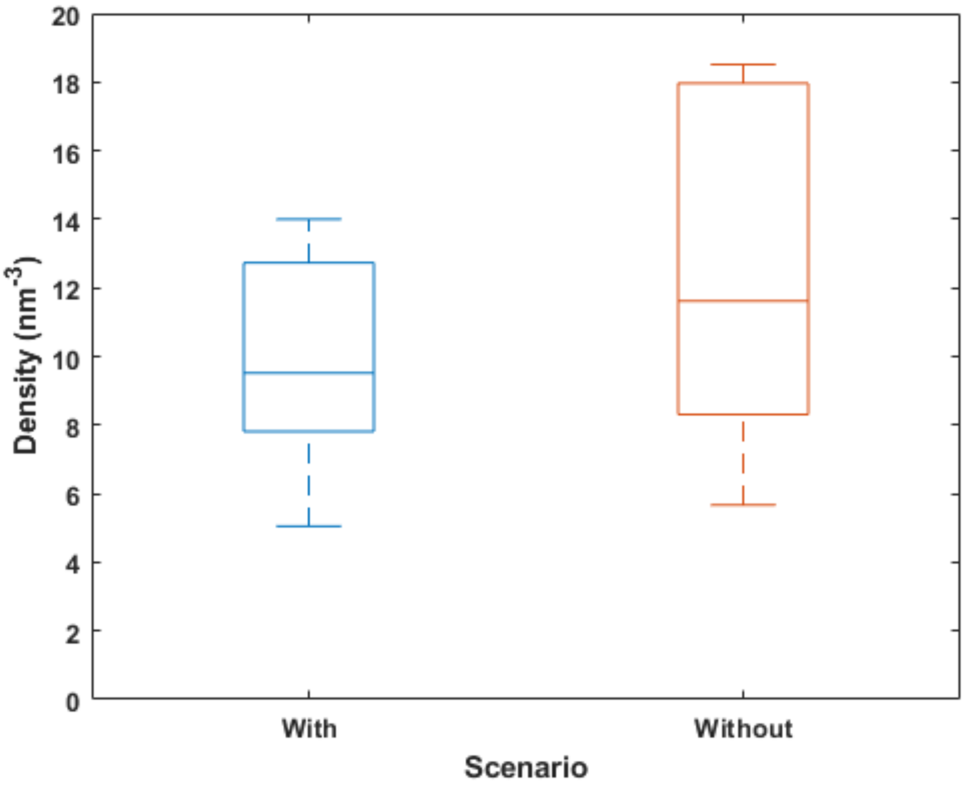

Figure 31. Difference in schooling density (nm^3) when a turbine is present in the model environment.

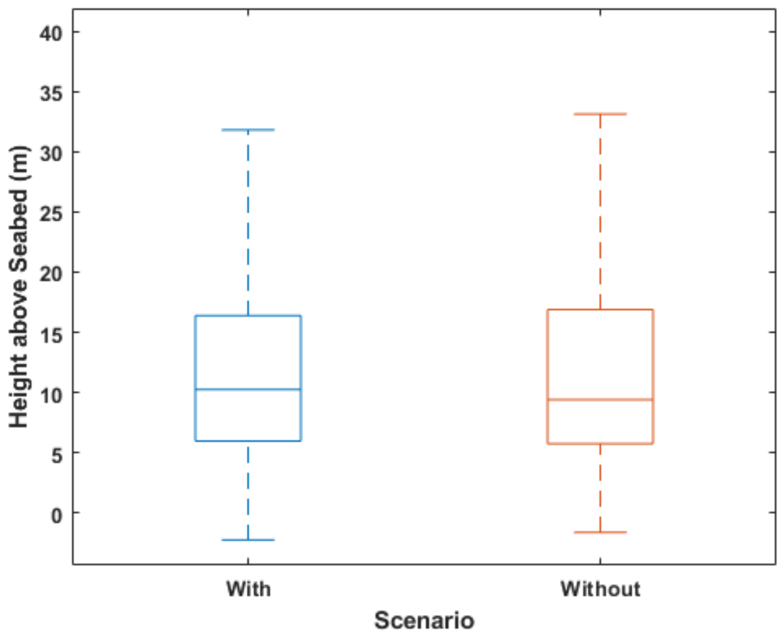

Figure 32. Effect of the turbine structure on the mean height above seabed (m) of all fish within the model.

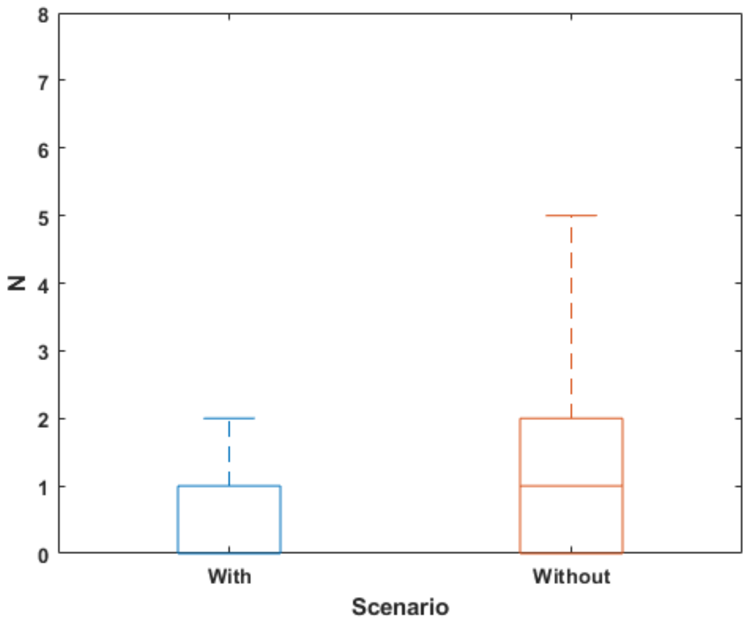

Figure 33. Effect of the turbine structure on the number of fish (N) within close association with the turbine structure.

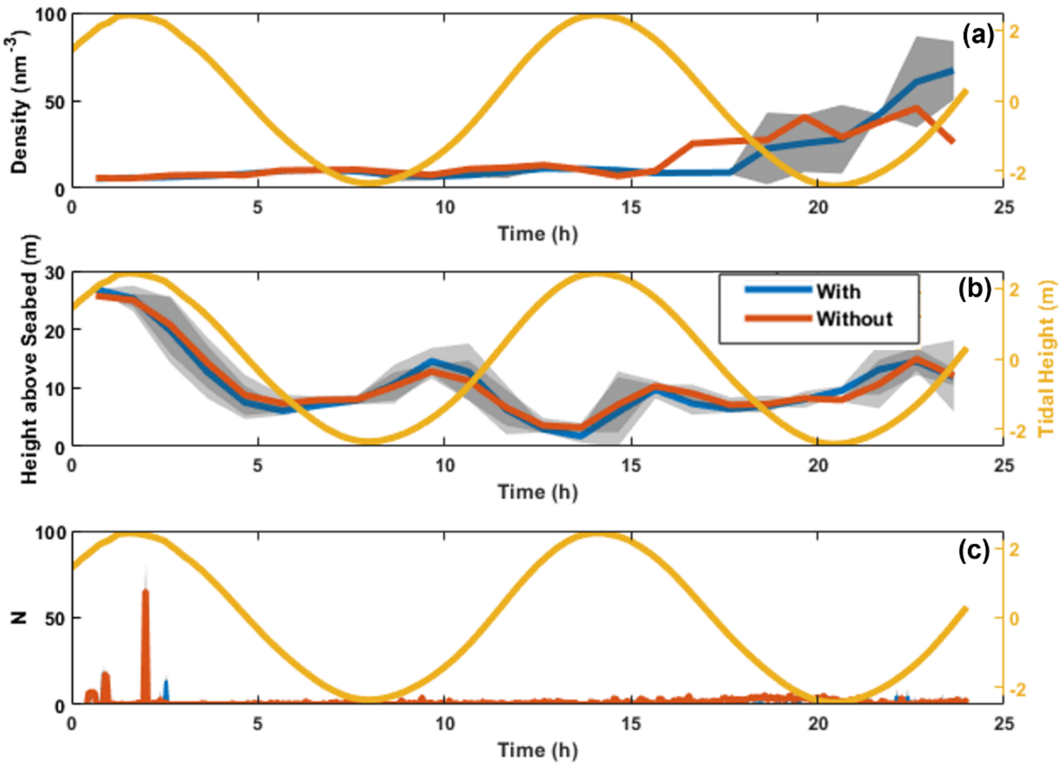

Figure 34. The effect of a turbine structure on fish (a) Density (nm^{-3}) (b) mean height above the seabed (m) (c) number of fish (N) within close association with the turbine structure. Figure legend shows when the environment was modelled with and without a TTS. Tidal height (m) shown on y_2 axis.

Discussion

This study developed an ABM to (1) assess the effect of a MREI on the hydrodynamic regime of a tidal stream environment and how alterations in hydrodynamics will affect 3D fish behaviour and distribution; (2) highlight areas where the behaviour of the fish may increase top-predator foraging efficiency and thus (3) predict how foraging opportunities for top-predators may be affected in the presence of a MREI. I used an ABM to predict 3D fish distribution and schooling capabilities of fish in the wake of a MREI and how these changed over a 24-hour model run. It is important to note that although the results presented here are useful in understanding how forage fish and top-predators may behave around a TTS, the results presented here are the output of model simulations rather than from empirical data collection and should be treated as such.

During sensitivity testing of the model, we found that schooling probability had the greatest influence on behaviour, as such, the schooling capabilities of fish will likely have an important impact on the observed behaviour of fish around a TTS. I also found that the presence of a TTS did not have a significant effect on the schooling behaviour and distribution of fish. Yet, the presence of a TTS did not deter fish from entering the nearby area. Thus, it is likely that these areas still will provide an important foraging hotspot for top-predators. These results, plus more, will be discussed further below.

4.1 Sensitivity Testing

All school metrics were significantly changed by altering the schooling probability, indicating that the schooling tendencies are of major influence on observed distributions and abundance. The strong influence of different schooling tendencies on fish behaviour is something that has not previously been observed in the literature. It highlights the importance of ground-truthing the modelled data with observed behaviour from field-work, so that the programmed schooling probabilities give rise to realistic schooling behaviours as this will have the greatest impact on the end results of the ABM. Hence, achieving an accurate schooling probability is important for the future predictive capabilities of this ABM.

The other model parameters largely had little to no effect on the abundance and distribution of fish within the model. However, having these other parameters allowed some key empirical behaviours that have been recorded in the literature to be present within the model, such as an increase in response probability leading to an increase in station holding behaviour (Fraser et al., 2018; Williamson et al., 2019; Williamson et al., 2021). The results of the sensitivity testing showed that more information on the vertical distribution of fish and the schooling densities of fish in a highly energetic tidal environment is needed to better parametrise the model. This would aid in the model's predictive capabilities and validate the model for use in further sites and applications in the future.

4.2 Effect of a Tidal Turbine Structure (TTS) on abundance and distribution

Both when the TTS was present and when it was absent, fish showed an increase in schooling density corresponding to a decrease in flow speeds, although previous studies have reported the largest schools occurring during periods of peak flow speeds (Fraser et al., 2018; Williamson et al., 2019). This could be as a result of the model programming, whereby reduced flow speeds allow fish to form denser schools, when in a natural environment they have been shown to disperse under these conditions. This highlights the need for focused data collection on fish school characteristics that can be used to parametrise the model.

Fish vertical distribution was inconsistent but generally showed a trend of rising in the water column during peak flow velocities. This is consistent to what has been observed previously in tidal-stream environments (Embling et al., 2012; Williamson et al., 2017; Williamson et al., 2019) and highlights the predictability of aggregations in energetic environments. The two scenarios showed consistent patterns of density and vertical distribution over time (figure 34), highlighting the consistency in behaviour of aggregations. This represents positive foraging opportunities for top-predators and shows that the introduction of a TTS had no effect on the aggregations of fish, either spatially or temporally, this indicates that a TTS doesn't affect the increased foraging opportunities that arise in tidal stream environments (e.g. Benjamins et al., 2015). However, this also suggests that top-predators will still be drawn to this area when a TTS is present. The depth that fish swim at during the model run overlaps with the location of the TTS. Moreover, upon visual analysis on the model outputs, when fish were seen entering the area behind the turbine-they will usually first move past the turbine structure, being advected by the current. Initially, this indicates that the TTS poses a significant risk to fish populations. However, although fish still move into close proximity of the TTS, it is likely that this will occur during slack tide, when the rotor blades of a turbine would be stationary, with occurrences of fish passing close to the turbine during peak flows, being rare. What's more, previous studies have identified the low collision risk posed by these structures to fish (Hammar et al., 2013; Broadhurst et al., 2014; Shen et al., 2016; Fraser et al., 2018; Williamson et al., 2019; Yoshida et al., 2021). Indicating that the risks from tidal turbine installations to fish, in relation to collisions and changes in distribution and behaviour, are minimal. Furthermore, there was a decrease in N when a turbine structure was introduced into the model environment, especially during peak flows (figure 34c). This indicates that fish show near-field avoidance of the turbine structure, an observation that is consistent with evidence from the literature (Williamson et al., 2017; Fraser et al., 2018). However, there is evidence that MREI have an attraction effect for fish in tidal stream environments (Fraser et al., 2018; Williamson et al., 2019; Williamson et al., 2021). Although this was not observed from the model outputs in this study. If an attraction effect were due to favourable current speeds in the wake of a TTS, it is likely that this would have been observed from the model outputs. However, as it was not, it suggests that an attraction effect is not due to current speed alone and may be because fish are simply attracted to large objects in their environment. Although, as other studies have reported a near-field avoidance effect (Viehman and Zydlewski, 2015; Shen et al., 2016; Bevelheimer et al., 2017), more information is needed on the fine-scale behaviour of fish in the wake of a MREI in order to parametrise realistic behaviour onto the agents.

Conclusions and Recommendations

The aim of this study was to assess the effect that TTS may have on fish populations as a result of changing hydrodynamics in the structure's wake. This study showed that the presence of a turbine structure showed to has no clear effect on forage fish behaviour and density, potentially indicating that these structures have no effect on prey availability and therefore foraging efficiency for top-predators. However, the results suggest that the area around these installations still provides an environment in which fish aggregations are predictable and dependable in space and time, in accordance with the literature (e.g. Zamon, 2003; Fraser et al., 2018; Williamson et al., 2019; Williamson et al., 2021). Thus, it is likely that top-predators would still be attracted to these sites. Although our results indicate that the most energetically efficient times for foraging would be when tidal current speeds are low, minimising collision risk. Moreover, as top-predators will likely frequent these sites during installation and maintenance of these structures, care should be taken to minimise the impacts to the surrounding wildlife.

ABM provides a methodology to assess the potential impacts of tidal stream turbines on fish without the need for expensive and challenging field studies. Parametrisation of the model was informed from physiological studies of behaviour, rather than from empirical evidence. Yet, behaviours seen in the literature have been observed here, highlighting the potential usefulness of this method. However, focussed data collection, regarding schooling behaviours of fish, is needed to improve parametrisation and model performance. In this way, an ABM approach can be used to guide the types of empirical data collection that are needed to help further the field. The method could then be applied to current MREI sites or sites that have been earmarked for future installations, since TELEMAC models are widespread in development sites, this method would be able to utilise those existing models to make predictions regarding potential impacts. The impact of scaling, from singular turbines to arrays, is also poorly understood. But the predictive capabilities of an ABM method would allow this to be measured in the future, once better information on fish behaviour has been collected to parametrise the model.

During sensitivity testing, schooling probability had the greatest effect on fish behaviour, indicating that different species of fish, with different schooling tendencies, will likely have varying behaviours in a tidal stream environment and different responses to a MREI. As such, this particular parameter must have careful consideration when it is being parametrised for the ABM. There should also be a good understanding of the dominant or important species present at a given location as differences in parametrisation of behaviour can have a significant effect on the model outputs. The model should also be updated to include behaviours such as attraction or avoidance behaviour of a turbine structure. However, better fine-scale observations of fish around a TTS are needed to inform the parametrisation of these behaviours.

Future work should focus on the collection of fish behaviour data to parametrise the model, using the data from the model runs to predict areas where encounters for top-predators would be greatest. By then using field-work observations to ground-truth the results and potentially make improvements to the model, the method could then be applied to multiple sites in the future.

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