Department of Biosciences

An Assessment of Artificial Floating Islands as a Method of Habitat Creation in Marine Environments

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Submitted to Swansea University in fulfilment of requirements for the Degree of Doctor of Philosophy

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Summary

Most megacities are located adjacent to the coast due to the continuous seaward migration of human populations; a process referred to as marine urban sprawl. The subsequent hardening of the natural coastline has caused the loss and degradation of coastal habitats. In order to halt, mitigate and compensate for further losses of biodiversity, it is important that habitat restoration techniques with involve ecological engineering are considered. Artificial floating islands (AFIs) are a habitat creation method used to improve water quality and support biodiversity in aquatic environments. This study aimed to assess the installation of AFIs as a restoration tool in heavily modified coastal water bodies. That included investigating: the suitability of halophytes for transplantation into the AFI matrix; the biofouling communities that establish on the AFIs; the abundance, species richness and behaviour of fish in association with AFIs; the density and behaviour of birds in association with the AFIs; and the public perception of current environmental concerns and therefore, opinion on AFIs as an ecological engineering method. Based on the results of this study sea purslane (*Halimione portulacoides*) would be recommended for transplantation on AFIs installed in saline environments. The invertebrate community assemblages were notably controlled by the primary settlement of blue mussel (*Mytilus edulis*) and Australian tubeworm (*Ficopomatus enigmaticus*). Juvenile phase European sea bass (*Dicentrarchus labrax*) and gull (*Laridae)* spp. foraged on the benthic invertebrates that fouled the AFIs underside and European eel (*Anguilla Anguilla*) rested in the matrix. The public supported the use of AFIs in coastal environments but concerns regarding maintenance and degradation were raised. In conclusion, this study highlighted the importance of AFI size, structure, location and vegetation cover as these factors influence the species composition, degree of isolation and environmental exposure, contributing to the overall success of AFI deployments in heavily modified coastal water bodies.

Declaration and Statements

Declaration

I, Jessica Ware certify that this work had not previously been accepted in substance for any degree and is not being concurrently submitted in candidature for any degree.

Signed: [redacted]………………………………………………… (candidate)

Date: 30 September 2020……………………………………………….

Statement 1

This thesis is the result of my own investigations, except where otherwise stated. Where correction services have been used, the extent and nature of the correction is clearly marked in a footnote(s).

Other sources are acknowledged by footnotes giving explicit references and a bibliography is appended.

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Authorship declaration

Ruth Callaway from Swansea University contributed to the publication of work in Chapter 6 of this thesis "Public perception of coastal habitat loss and habitat creation using artificial floating islands in the UK". The candidate (Jessica Ware) researched and designed the survey, collected and analysed the data and wrote the article. Ruth Callaway assisted in survey design and analysis and reviewed several drafts of the article before publication submission.

We agree with the above stated "proportion of work undertaken" for the submitted peer-reviewed manuscript contributing to this thesis:

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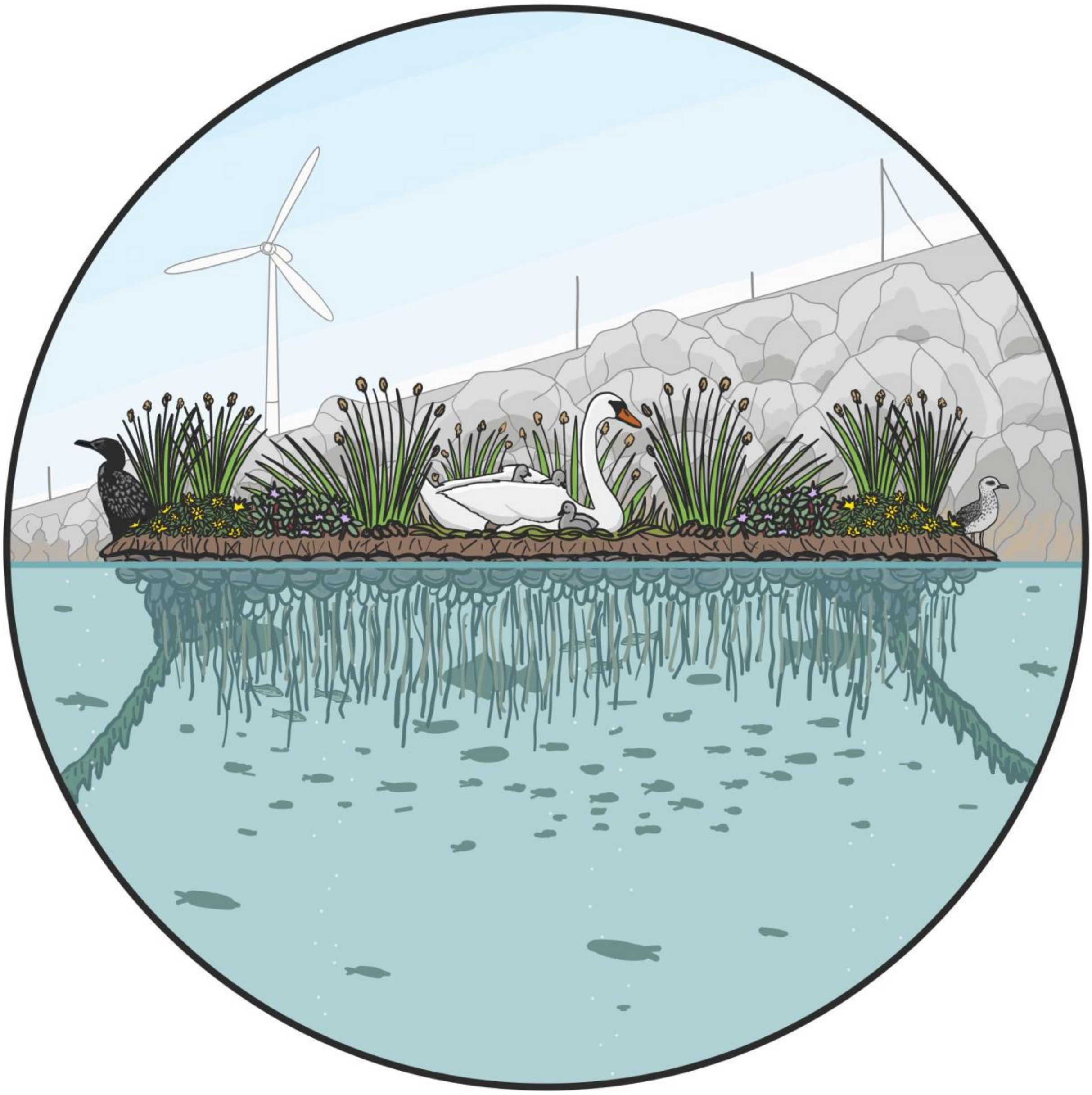
Abbreviations	Definitions
AFI	Artificial floating island
UK	United Kingdom
FAD	Fish aggregation device
SE	Standard error
N	Nitrogen
P	Phosphorus
R/S	Root/shoot
ANOVA	Analysis of variance
ANOSIM	Analysis of similarities
Spp.	Species (plural)
Sp.	Species (singular)
NA^{+}	Sodium
Cl^{-}	Chloride
3-D	Three dimensional
ABP	Associated British Ports

Chapter 1: General Introduction



(Illustration commissioned for thesis; Steedman, 2019)

1.1 Changing Coastal Landscapes

Coastlines account for <15 % of the terrestrial surface and support >60 % of the human population (European Environment Agency, 1999; Mercader *et al.*, 2017a). By 2025, this figure is estimated to increase by at least 15 %, resulting in approximately 1 billion more individuals occupying the coast (European Environment Agency, 2006; Beck & Airoldi, 2007; Mercader *et al.*, 2017a). This so called marine urban sprawl is due to rapid population increase (Small & Nicholls, 2003), availability of resources via marine trade (Parrish, 1989), access to transport links, recreational facilities attracting tourism (Hall, 2001) and the aesthetic benefits of living by the coast (Neumann *et al.*, 2015). Infrastructure associated with offshore aquaculture (Ogburn, 2007), renewable energy technologies (Asif & Muneer, 2007) and oil and gas exploration (Cordes *et al.*, 2016) are also on the rise, as global energy demand increases and the availability of arable land declines (Dafforn *et al.*, 2015a). The resultant shift of human populations towards the coast has potential environmental benefits for inland resources already strained by urbanisation (Browne & Chapman, 2011; Wang & Wang, 2015). However, this strain is now placed on coastal landscapes due to the development of marinas, seawalls and barrages that all facilitate increased urbanisation, industry and tourism activities (Bellan & Bellan-Santini, 2001; Holloway & Connell, 2002; Perkol-Finkel *et al.*, 2006a).

As the climate continues to warm, large-scale glacial melt has resulted in a global rise in sea level of approximately 10 – 25 cm during this century (Hall & Fagre, 2003), contributing to an increased rate of shoreline retreat and habitat degradation (Dugan *et al.*, 2008). In order to protect communities vulnerable to flooding and erosion, sea defences such as groynes, breakwaters and riprap revetments are constructed and can form the dominant habitat feature, creating a homogenised and less complex environment in intertidal zones (Beck & Airoldi, 2007; Chapman & Blockley, 2009; Mercader *et al.*, 2017a). Structural developments on the coast are increasing at a rate of 3.7 – 28.3 % per annum (Duarte, 2014; Bishop *et al.*, 2017). The combined impact of anthropogenic activities has resulted in the overexploitation of natural resources, a rise in pollution and waste disposal into the marine environment and the subsequent loss and fragmentation of marine habitats, threatening marine ecosystems and species diversity on a global scale (Coll *et al.*, 2010; Dafforn *et al.*, 2015a; Mercader *et al.*, 2017a). Therefore, a clear understanding of the small and large-scale impacts of artificial structures on marine biota and environmental processes is important, in order to effectively mitigate for the potential negative impacts associated with marine urban sprawl (Bishop *et al.*, 2017).

1.2 Impacts of Coastal Armouring

Although terrestrial and marine habitats can be naturally fragmented and linear, the addition of artificial structures causes greater spatial disconnection affecting an organisms movement, genetic structure of the population and the flow of organic detritus and nutrients, influencing trophic connectivity (Bishop *et al.*, 2017). This could affect the trophic interactions between marine species and society, with 10 – 12 % of the human population relying on the economic output of fisheries and aquaculture (Food and Agriculture Organization, 2014; Bishop *et al.*, 2017). The extent of fragmentation and therefore, the overall impact depends on a number of factors which include: the spatial configuration of the fragmented habitats (Ricketts, 2001), the dispersive capabilities of the species (Wiens *et al.*, 1997), their interaction and reliance on the habitat and the individuals behaviour (Goodsell *et al.*, 2007). In Sydney Harbour, Australia, intertidal assemblages present in natural habitat patches were more species diverse than mixed patches adjacent to artificial structures and completely fragmented patches (Goodsell *et al.*, 2007). Sea defences also prevent the landward expansion of coastal habitats, which in combination with rising sea levels and the increasing severity of storms, restricts coastal habitats into a narrow band and causes habitat loss; a process referred to as coastal squeeze (Doody, 2013; Pontee, 2013).

In addition to spatial disconnection and coastal squeeze, there is increasing evidence that artificial structures do not support the same assemblages that previously thrived in the natural habitat (Hunter & Sayer, 2009; Lam *et al.*, 2009; Bulleri & Chapman, 2010). Differences in size, composition (Goodsell *et al.*, 2007), texture (Coombes *et al.*, 2015), topographic complexity (Myan *et al.*, 2013), surface area, orientation (Glasby & Connell, 2001) and age of the installed artificial structure (Bulleri & Chapman, 2010; Bishop *et al.*, 2017) all contribute to the formation of 'urbanised ecosystems' that often favour non-indigenous species (Airoldi *et al.*, 2015). Artificial structures also cause fluctuations in the surrounding physico-chemical conditions, aiding the development of niche habitats (Bulleri & Airoldi, 2005). For example pilings and pontoons can support 2.5 times more invasive species than adjacent natural reef and have been referred to as invader hotspots (Glasby *et al.*, 2007; Dafforn *et al.*, 2009; Dafforn, 2017). Differing hydrodynamics associated with floating and fixed structures is a key abiotic factor contributing to community development, which strongly influenced biofouling assemblages on artificial devices deployed in the Gulf of Aqaba (Perkol-Finkel *et al.*, 2006a; Megina *et al.*, 2016). In Victoria Harbour, Hong Kong species assemblages on vertical sea walls were compared to natural, rocky habitat (Lam *et al.*, 2009). Each site supported several common species however, percentage cover of the chiton *Acanthopleura japonica* and hooted oyster (*Saccostrea cucullata*) was greater on the artificial sea wall (Lam *et al.*, 2009). Differences in zonation patterns were also observed suggesting that this was caused by the vertical orientation of the sea wall, affecting localised abiotic factors such as light availability, temperature and humidity (Lam *et al.*, 2009).

The impact of coastal armouring on adjacent habitats should also be considered as artificial structures can cause reduced light availability, variations in flow regimes, sediment movements and an increased risk of pollution incidents via leaching (Dafforn *et al.*, 2015b; Bishop *et al.*, 2017; Heery *et al.*, 2017). The formation of secondary artificial reefs can have positive effects on the local environment including an increase in nutrient availability via the establishment of periphyton and biofouling invertebrates that attract predatory species (Krone *et al.*, 2013; Reubens *et al.*, 2014; Nall *et al.*, 2017). Therefore, it is important to consider the potential impact pathways of a coastal development on a site specific basis.

1.3 Mitigation, Compensation and Habitat Restoration

Once the impact pathways of a proposed development have been identified, mitigation measures should be considered or compensation techniques, if mitigating the impact is not possible. Mitigation is defined as 'the act of making any impact less severe' (Elliott & Cutts, 2004; Elliott *et al.*, 2007a). Compensation can refer to economic, resource or ecological compensation and is broadly defined as 'to make up or make amends for damage' (Elliott & Cutts, 2004; Elliott *et al.*, 2007a). Ecological compensation includes the concept of biodiversity offsetting, whereby a site of ecological equivalent to the site impacted by proposed works is created (Pöll *et al.*, 2016). The ideal scenario for the conservation of a deteriorated site is to allow the natural habitat to recover under its own defensive mechanisms, with no intervention from humans, known as the 'do nothing' approach (Hoggart *et al.*, 2014; O'Shaughnessy *et al.*, 2020). However, different circumstances may require intervention as a result of public safety concerns (Cals *et al.*, 1998), infrastructure deterioration (Liversage & Chapman, 2018), energy developments or extent of habitat degradation past a recoverable state (Dafforn *et al.*, 2015b; Morris *et al.*, 2018; O'Shaughnessy *et al.*, 2020).

Habitat or ecological restoration via anthropogenic intervention refers to the process of assisting and managing the recovery of an ecosystem that has been degraded, damaged or destroyed by anthropogenic or natural processes (Seaman, 2007). It can be subdivided into three types: response to a degraded or anthropogenically changed environment; response to a single stressor; and habitat enhancement or creation (Elliott *et al.*, 2007a). The methods of habitat restoration vary according to circumstance but are consistent with the overriding aim to encourage ecosystem development back to its original, self-sustaining state with little assistance once established (Seaman, 2007). More specifically, the restored ecosystem should be similar in species composition, population density and biomass structure to the ecosystem present prior to deterioration or similar to a comparable site (Elliott *et al.*, 2007a). The success of habitat restoration can be monitored via both biotic and abiotic factors within that habitat; as highlighted in Ferrario *et al*., (2016), the role of biotic factors in the success of habitat restoration has been largely overlooked and requires further attention (Ferrario *et al.*, 2016).

1.4 Ecological Engineering

Ecological-engineering (eco-engineering) refers to the modification of planned or existing structures integrating ecological theory into structural design to influence physico-chemical processes (Type A), or direct engineering of biota via replanting or restocking (Type B) (Elliott *et al.*, 2016; Morris *et al.*, 2016; Dafforn, 2017). For example, Type A eco-engineering includes adding texture to a sea wall via small indents, larger pits or water holding features, such as flower pots (Morris *et al.*, 2017; Strain *et al.*, 2018a, 2018b). The aim of an eco-engineering project is to reduce stressors within an environment and act as a subsidy, increasing ecosystem functions; this relationship is known as the stress-subsidy hypothesis (Odum *et al.*, 1979a; Hanley *et al.*, 2017). Eco-engineering can include soft engineering, which describes temporary or 'soft' techniques to aid rehabilitation of a site (Elliott *et al.*, 2016), such as encouraging coastal plant growth (Arkema *et al.*, 2013) and sand nourishment processes (Stive *et al.*, 2013) to aid dissipation of wave energy and protect coastlines from erosion (Strain *et al.*, 2018b). Alternatively, it may involve hard engineering techniques, which refer to the introduction of permanent physical features such as concrete groynes (Elliott *et al.*, 2016), gabion baskets (O'Shaughnessy *et al.*, 2020) and artificial floating islands (AFIs).

Approximately 60 % of research in this field has been conducted in Australia, Israel, Europe and North America, in intertidal or subtidal regions (Strain *et al.*, 2018b). As 'ecologically stressed' coastal habitats it has been hypothesised that intertidal and subtidal zones will show greater positive results from eco-engineering solutions than habitats with fewer stressors (Bertness & Callaway, 1994; Strain *et al.*, 2018b). In heavily modified coastal water bodies such as marinas and docks, eco-engineering offers a means of enhancing existing or planned structures to benefit local biodiversity, while maintaining the integral anthropogenic function of the structure (Martins *et al.*, 2010; Browne & Chapman, 2011; Naylor *et al.*, 2012a). In some cases, the installation of artificial habitats with no anthropogenic function maybe necessary to support target species and prevent further declines in biodiversity.

Artificial habitat creation in marine environments was first introduced in Japan and was quickly adopted as a method of improving ecotourism (Shani *et al.*, 2012), recreational fishing and diving (Kirkbride-Smith *et al.*, 2013), aquaculture outputs (Seaman, 2007) and as a restoration method to support biodiversity (Perkol-Finkel *et al.*, 2006a; Dafforn *et al.*, 2015a).

Type A eco-engineering projects in the marine environment have largely focused on the socio-economic benefits and ecosystem services provided by the addition of artificial reefs (Rilov & Benayahu, 2000; Perkol-Finkel *et al.*, 2006b; Oricchio *et al.*, 2016). A number of factors can influence the colonisation of artificial reefs including: composition of the substratum (Burt *et al.*, 2009), microscale roughness (Sempere-Valverde *et al.*, 2018), shape (Perkol-Finkel *et al.*, 2006b), age (Perkol-Finkel & Benayahu, 2005), interstitial space between deployed material (Sherman *et al.*, 2002) and location (Kienker *et al.*, 2018). In addition, the size of the artificial reef can directly impact on the biomass, density and species diversity of the community assemblage associated (Bohnsack *et al.*, 1994; Abelson & Shlesinger, 2002). This is known as carrying capacity, which in ecological terms refers to 'the number of individuals in a population that the resource of a habitat can support' (Cohen, 1997; Elliott *et al.*, 2007a). For example, Rounsefell (1972) concluded that artificial reefs must be a minimum of 5700 m^3 to support a self-sustaining fish population (Rounsefell, 1972).

Alternatively, to artificial reefs, the deployment of AFIs is considered a low cost and energy eco-technology that may also provide ecosystem services and socio-economic benefits for local communities. They have largely been installed in freshwater environments assessing the treatment of aquaculture wastewater, sewage, rivers and lakes (Li *et al.*, 2010; Ning *et al.*, 2014; Pavlineri *et al.*, 2017). While there is considerable research on AFIs in freshwater and estuarine systems (Nakamura & Mueller, 2008; Yeh *et al.*, 2015; Chee *et al.*, 2017), there is a lack of research on AFIs installed in the marine environment; both exposed and heavily modified coastal water bodies.

1.4.1 Artificial Floating Islands

When considering the installation of AFIs to support habitat restoration, it is important to understand the ecosystem development of isolated islands and associated biotas, described in the 'island biogeography theory' and lessons learnt since its articulation. Geographically isolated islands develop distinct biotas (Buffon, 1761), which vary according to island characteristics such as size, resource availability (Forster, 1778), habitat heterogeneity and anthropogenic disturbance, plus individual species characteristics such as dispersal capacity, adaptive evolution and interspecific competition (Brown & Lomolino, 2000). The degree of isolation and area of an island are key factors that affect species composition, referred to as the species-isolation and species-area relationships (Losos & Ricklefs, 2009). The 'equilibrium theory' also predicted the species richness of isolated islands based on island characteristics and the interaction between immigration, extinction and evolution (Lomolino, 2009; Losos & Ricklefs, 2009). In context to habitat restoration, island biogeography theory has been considered when designing nature reserves, as it supports the creation of one large area in comparison to several small areas (Higgs, 1981). For this thesis, the theory highlighted the need to evaluate how the structure of AFIs including established vegetation and size may influence the islands ecological succession.

1.4.1.1 Structure

AFIs broadly consist of an integrated connection grid and buoyant matrix, where the selected growing medium can be attached with pre-established emergent vegetation (Figure 1.1) (Burzaco & Frog Environmental, 2016). Over time, the vegetation grows extensive root systems through the woven plastic matrix and into the water column. The matrix is a robust and flexible material allowing it to support vegetation growth and withstand harsh environmental conditions (Floating Island International, 2013). It is a soilless structure that once established, forms a localized ecological community within the submerged roots of the selected plants and on the surface of the structure itself; these include algal communities, biofilms, zooplankton, macroinvertebrates and epibiotic species (Yeh *et al.*, 2015). The thickness of the AFI can be adjusted according to the habitat it is deployed; for instance, intertidal habitats require a thicker planting matrix than reservoirs lacking water flow.

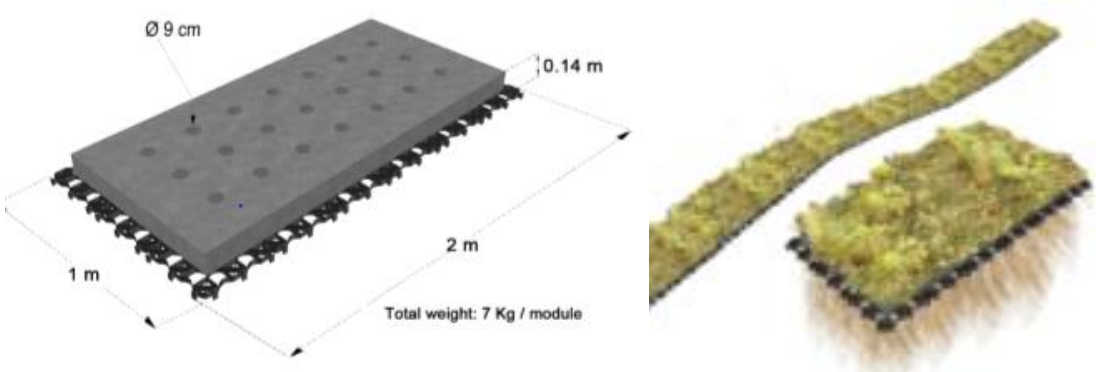


Figure 1.1: *Left* – 2 m² matrix unit with 21 planting holes, 9 cm in diameter. *Right* - Plants can also be pre-grown on coir matting and attached to the matrix unit (Burzaco & Frog Environmental, 2016).

1.4.1.2 Phytoremediation

Like a naturally occurring wetland system, AFIs are deemed to have a net positive effect on the local environment by improving water quality, via the removal of suspended solids and organic matter and biosynthesis of nutrients effectively purifying the surrounding water body (Floating Island International, 2013; Chang *et al.*, 2014; Overton *et al.*, 2015). As part of chlorophyll biosynthesis, excess nutrients such as nitrogen (N) and phosphorus (P) are incorporated into plant tissue. Plants also remove nutrients and contaminants by rhizofiltration, whereby the contaminant is stored within the roots (Dushenkov *et al.*, 1995; Verma *et al.*, 2006; Bonanno & Lo Giudice, 2010). The assimilation of available ammonia is controlled by a number of different processes including nitrification, denitrification and anaerobic ammonium oxidation (Pavlineri *et al.*, 2017). The retention of pollutant loads and water quality improvement is dependent on the ratio between the total area of the watershed and created wetland (Carleton *et al.*, 2001). Approximately 0.1 – 1 % of the watershed should be converted to wetland in order to detect tangible water quality improvement (Ham *et al.*, 2010). In addition, the older the wetland the greater the retention of pollutant loads as the system progresses towards an advanced state (Moreno *et al.*, 2007).

The installation of AFIs to improve water quality has applications in both natural systems, heavily modified water bodies and in the treatment of wastewater from multiple sources including aquaculture, agriculture, household, livestock, meat processing and refinery plants (Yeh *et al.*, 2015). Measuring the retention of pollutant loads of different halophytes when hydroponically grown in saline water is therefore important in order to create efficient AFI treatments. Sea aster (*Tripolium pannonicum*) and common glasswort (*Salicornia europaea*) were grown hydroponically in salinity treatments of 10 PSU in order to assess their growth when exposed to different forms of N: ammonium (NH_4^+), nitrate (NO_3^-) and ammonium nitrate (NH_4NO_3) (Quintã *et al.*, 2015). Common glasswort in the ammonium treatment had the lowest dry weight biomass and the highest dry weight biomass in the ammonium nitrate treatment by the end of the experiment (Quintã *et al.*, 2015). In contrast, sea aster had the highest dry weight biomass in the ammonium treatment and the lowest in the ammonium nitrate treatment by the end of the experiment (Figure 1.2) (Quintã *et al.*, 2015).

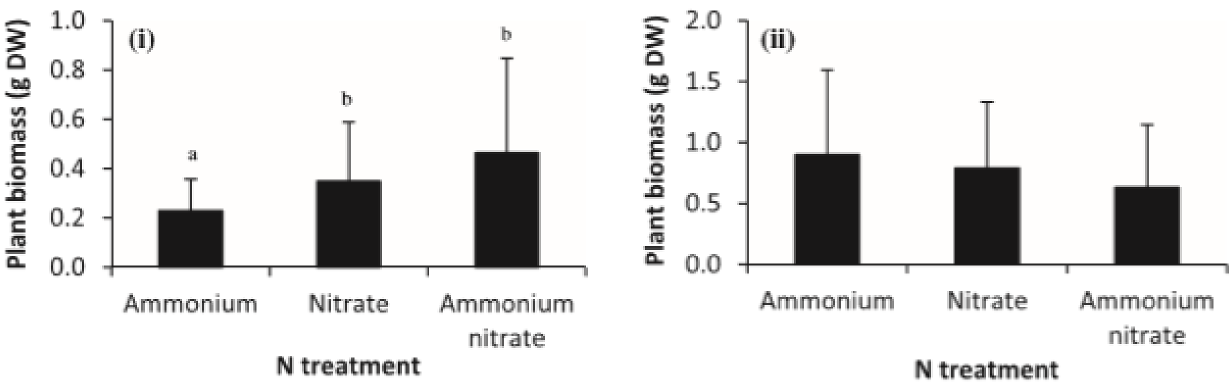


Figure 1.2: A comparison of the total dry weight biomass (g) of common glasswort (*Salicornia europaea*) (i) and sea aster (*Tripolium pannonicum*) (ii) after exposure to ammonium, nitrate and ammonium nitrate treatments over the course of a 223 day experiment (Quintã *et al.*, 2015).

Sea aster showed no treatment preference and outperformed common glasswort (Quintã *et al.*, 2015). More research is required on saline hydroponic bioremediation and plant growth when using halophytes that occupy different elevational gradients of temperate saltmarsh ecosystems.

1.4.1.3 Habitat Creation

The directional change in community composition following a sequence of disturbance events over time is a process referred to as succession (MacMahon, 1980; Greene & Schoener, 1982). Disturbance events include fluctuations in energy utilisation measurable via total biomass, changes in species composition and structural and functional characteristics of a site (Vinogradov & Shushkina, 1984; Prach & Walker, 2011). The duration of succession from early colonisation in the initiation state to its equilibrial climax community varies across different habitats (Margalef, 1989; Sandin & Sala, 2012) and will play a key role in the development of epifaunal and terrestrial habitats created by AFI installations. In aquatic environments, once a structure is immersed in the waterbody a conditioning layer of dissolved organic material will coat the surface (Taylor *et al.*, 1997) followed by microorganisms, phytoplankton and larvae creating a biofilm (Callow & Callow, 2002; Salama *et al.*, 2018). The biofilm aids colonisation of 'soft' or 'hard' biofouling organisms (Callow & Callow, 2002; Yan *et al.*, 2009) which form an epifaunal community changing through time as a result of biotic and abiotic disturbances during pre and post-settlement processes (Vance, 1988; Fraschetti *et al.*, 2002; Oricchio *et al.*, 2016). Succession is not always predictable (Clenn-Lewin, 1980) however, considering the mechanisms, stages and trajectories of specific sites over time, could aid the approach to restoration efforts and determine the desired successional stage (Rey Benayas *et al.*, 2009; Prach & Walker, 2011).

In Yundang Lagoon, a saline heavily modified water body in Xiemen Island, China, the biofouling communities on three AFIs transplanted with sea purslane (*Halimione portulacoides*) were assessed during a three year deployment (Xie *et al.*, 2019a). The abundance, biomass and composition of biofouling invertebrates varied according to season and location of the AFI installation as hydrodynamics, dissolved oxygen availability and extent of eutrophication varied across the lagoon (Xie *et al.*, 2019a). For example, increasing water temperatures caused a shift in the dominant fouling species from *Corophium uenoi*, *Grammaropsis laevipalmata* and *Ampithoe valida* to the black striped mussel (*Mytilopsis sallei*) (Xie *et al.*, 2019a). It is important to consider the colonisation of nonindigenous species (NIS) during the planning stages of an AFI as artificial structures tend to be fouled by NIS, acting as a potential propagule for their dispersal (Glasby *et al.*, 2007). Floating structures such as pontoons often installed in shallow waters close to the shore have been shown to recruit a higher number NIS than native species (Hurlbut, 1991; Glasby *et al.*, 2007).

Alternatively, an 11.9 m^2 AFI was deployed in a freshwater pond to determine if the total biomass of bluegill (*Lepomis macrochirus*) and large-mouth bass (*Micropterus salmoides*) increased as a result of the installation (Pardue, 1973). After 2.5 years of growth, the overall fish biomass increased by 20 %. The AFI provided an attachment surface for periphyton, increasing nutrient availability and the carrying capacity of the pond, in addition to shelter, reducing predation risk (Pardue, 1973; Neal & Lloyd, 2018). In addition, a 4.5 m^2 AFI was installed in Chicago River to assess fish species richness and abundance associated with the AFI, in comparison to a local dock and an open water site (Yellin, 2014). The total number of fish in association with the AFI was 40.8 % higher in comparison to the local dock. However, there were no significant difference in species diversity (Yellin, 2014). Unlike in the previous study, the abiotic conditions at the three comparison sites were less controlled and reliant on attracting fish using bait in the minnow traps. This experiment was also undertaken within a freshwater system, highlighting the lack of research on the interaction of fish with AFIs installed in marine environments.

AFIs are increasingly being installed to provide additional refuge, nesting substratum and roosting sites for birds (Azim *et al.*, 2005; Zhao *et al.*, 2012; Lu *et al.*, 2015). For example, 10 AFIs with woven palm trees creating a shelter were installed in Arrowhead Marsh, California to assess utilisation by the California ridgeway's rail (*Rallus obsoletus obsoletus*); an endangered species with high mortality rates due to increased inundation of intertidal habitat during the winter (Overton *et al.*, 2015). In 2010 and 2011, the AFIs were used 300 times more frequently than expected, largely during the daytime and correlating with the tidal regime (Overton *et al.*, 2015). Further research is required to determine whether the presence of AFIs over a longer time period would reduce mortality rates of California ridgeway's rail and potentially breeding success. In contrast, 60, 8.64 m^2 AFIs were installed over 17 years in the breeding territories of black throated loons (*Gavia arctica*) in Scotland, to provide nesting sites during periods of flooding since 1976 (Hancock, 2000). As a result of their installation, chick production increased by 44 % between 1992 – 1995 (Hancock, 2000) (Figure 1.4).

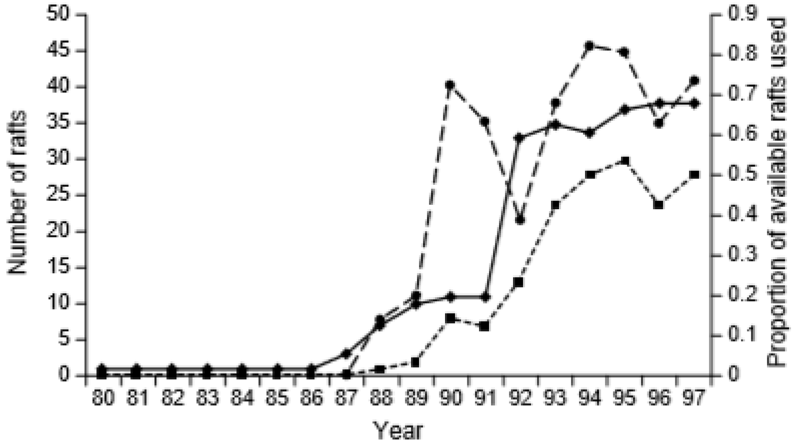


Figure 1.3: A comparison of the number of rafts (diamond) available and the proportion of those rafts being used (square) by black throated loons (*Gavia arctica*). The number of rafts used within a year is shown as a circle (Hancock, 2000).

Other similar examples include great crested grebes (*Podiceps cristatus*) in Hogganfield Havens, Scotland and Caspian terns (*Hydroprogne caspia*) in Dutchy Lake, Oregon (Floating Island International, 2008; Glasgow City Council, 2016). In the latter example, AFIs were used to attract Caspian terns and encourage breeding activity away from the migration routes of chinook salmon (*Oncorhynchus tshawytscha*) and rainbow trout (*Oncorhynchus mykiss*) juveniles. This example was effective due to the lack of natural nesting substratum available for Caspian terns to breed (Floating Island International, 2008) and demonstrated the broad application of AFIs in relation to bird and fish conservation.

1.5 Aims and Hypotheses

The aim of this thesis was to assess AFIs installed in heavily modified coastal water bodies to answer the overarching question 'Can artificial floating islands be used as a restoration tool in heavily modified coastal water bodies to increase their ecological potential?'. Heavily modified water bodies are surface waters that have been physically altered by anthropogenic activities, substantially changing its hydrogeomorphological characteristics (Borja & Elliott, 2007; Temino-Boes *et al.*, 2018). The thesis was motivated by knowledge gaps on viable compensation techniques for large-scale marine renewable infrastructure projects such as Tidal Lagoon Power, that have the potential to cause coastal habitat loss. The following six testable hypotheses were identified, focusing on halophytes, aquatic invertebrates, fish, birds and the public perception of AFIs:

- Halophytes present at the lower limit of saltmarsh will grow more successfully via salinity stimulated growth in the high salinity hydroponic treatment, in comparison to halophytes present in the upper limits of saltmarsh.
- The fouling community assemblages of AFIs is distinctly different on the horizontal surface in comparison to the vertical edge.
- Fish will change their vertical distribution in the tank with an AFI present in comparison to without an AFI present. Vegetated AFIs installed in heavily modified coastal water bodies will attract a higher number of fish than pontoons and unshaded sites which lack structures at the surface.
- Vegetated AFIs installed in heavily modified coastal water bodies attract a higher density and species diversity of birds than alternative hard structures within the same survey area.
- The majority of the respondents will be aware of the ecological functioning role of AFIs and would support their installation in coastal environments.

In order to address the overarching question and testable hypotheses the thesis was subdivided into seven chapters described below:

Chapter 2 monitored the growth of five halophytes. It assessed if common glasswort (*Salicornia europea*), sea rush (*Juncus maritimus*), sea aster (*Tripolium pannonicum*), common cordgrass (*Spartina anglica*) and sea purslane (*Halimione portulacoides*) could be recommended for planting in AFIs installed in heavily modified coastal water bodies. This was based on two, 8 week experiments where each species was hydroponically grown in two salinity treatments in order to measure and compare plant growth via fresh weight change and dry weight measurements. The study also assessed the root/shoot (R/S) of each species, the establishment of roots through the matrix and the leaf length change. Visual counts of three AFIs installed in heavily modified coastal water bodies with the five halophytes transplanted were conducted during the course of the deployment. In addition, the halophytes that survived the AFIs installation were removed to measure the fresh weight and dry weight of the leaves, stems and roots and assess which halophytes established successfully.

Chapter 3 assessed the biofouling communities on the inner and outer horizontal surface and vertical edge of two AFIs. The size and dry weight of blue mussels that fouled on the AFI was determined and compared across the three sections. The succession of fouling organisms was discussed based on remote underwater video footage collected as part of Chapter 4. The chapter determined if the AFI could support native fish populations as a feeding site, based on the invertebrate community assemblages that colonised the AFIs. It also made recommendations on how to manage biofouling on AFIs in heavily modified coastal environments, in order to ensure that the buoyancy of the AFI is not compromised.

Chapter 4 used remote underwater video footage to assess differences in the fish relative abundance (MaxN), species richness and behaviour in association with three AFIs, hard structures and unshaded areas in heavily modified coastal water bodies. The chapter also investigated the abiotic and biotic factors influencing fish assemblages and behaviour, including life cycle stage, food availability, shelter and water chemistry. As part of a collaboration with Bristol Aquarium, the vertical distribution and shoaling behaviour of 13 native fish in the absence and presence of an AFI was assessed under controlled conditions in a tank experiment.

Chapter 5 used vantage point surveying techniques to monitor bird behaviour, density and species diversity on the AFIs in comparison to hard structures and open water habitats. Ethograms were also conducted to gain more detail on individual species behaviour. A habitat complexity assessment was used to compare the two installation sites and address potential factors influencing differences in bird habitat utilisation. This chapter determined how AFIs may be used by coastal bird populations installed in heavily modified coastal environments.

Chapter 6 gained an understanding of the public perception of coastal habitat loss and the use of eco-engineering methods such as AFIs in coastal environments. An online survey consisting of eight questions and 200 respondents determined the public awareness of local habitat restoration or creation projects, the ecological functioning role of AFIs and whether the respondent would support AFI initiatives as a method of habitat creation within coastal environments. Further, the study aimed to assess whether public awareness correlated with proximity of residency from the coast.

Chapter 7 combined the ecological and social ecosystem services identified during this study and evaluated the pros and cons of installing AFIs in heavily modified coastal water bodies. The chapter indicated the limitations of this study and provided recommendations for future research based on knowledge gaps on AFIs as an eco-engineering method. Practical recommendations for future project management were also identified based on logistical challenges faced during the installation of AFIs in The Prince of Wales Dock and Swansea Marina.

1.6 Artificial Floating Island Deployments

Three artificial floating islands (AFIs) were installed in heavily modified coastal water bodies in Swansea. Two 8 m² AFIs (commercially sold as Biohavens®) were installed on 28th and 29th September 2017; one located in Swansea Marina and one in The Prince of Wales Dock. A 13.2 m² AFI was also installed on 17th May 2018 in The Prince of Wales Dock (Table 1.1). The size of the AFIs installed was experimental and determined based on the requirements to test the hypotheses and due to feasibility and available funding. The locations of the AFIs were selected as both Swansea Marina and The Prince of Wales Dock are subject to the influx of seawater from Swansea Bay, via the swing gate entrance to the Port of Swansea or the Tawe Barrage and associated weirs. In addition, each AFI was attached to an existing fixed structure ensuring that they did not obstruct recreational activities and were accessible for monitoring.

Table 1.1: The size, deployment and removal dates, location, fencing installation and monitoring completed on the three artificial floating islands (AFIs) deployed in The Prince of Wales Dock and Swansea Marina.

AFI size (m²)	Deployment date	Location (degrees, minutes, seconds)	Fencing (Yes/No)	Monitoring completed	Removal date
8	28/09/2017	The Prince of Wales Dock: 51°37’10.6”N 3°55’30.0”W	Yes	Halophytes, fish and birds	04/06/2019
8	29/09/2017	Swansea Marina: 51°36’56.3”N 3° 56’26.0”W	Yes	Halophytes, epibenthic invertebrates, fish and birds	05/06/2019
13.2	*13/11/2017 *30/04/2018 17/05/2018	The Prince of Wales Dock: 51°37’09.8”N 3°55’29.8”W	No	Halophytes, epibenthic invertebrates, fish and birds	03/06/2019

* Failed deployment dates.

The 8 m² AFI in The Prince of Wales Dock and in Swansea Marina consisted of four 1 m x 2 m units, that were two matrix layers thick and connect via the integrated connection grid (Chapter 1; Figure 1.1). These units are only suitable for deployment in non-tidal areas. The 8 m² AFI in The Prince of Wales Dock was installed using 6 mm stainless long-link chain fed through four anchor points and attached to a large, moored buoy and two 20 kg concrete weights to fix the AFI into position. In contrast, the 8 m² AFI in Swansea Marina was installed using 6 mm long-link chain fed through all four anchor points and attached to two pilings present in the designated boom area. During the first 6 months of the two AFIs deployment, fencing was attached to reduce bird activity and allow time for the five halophyte species transplanted on the AFIs to establish roots through the matrix. The fencing was removed in May 2018.

The second AFI in The Prince of Wales Dock was 13.2 m^2 and consisted of four 1.5 m x 2.2 m units that were four layers thick and lacked an integrated grid. Unlike the 8 m^2 AFIs, each unit had four 19 mm plastic conduits running at right angles along its length and width, which can be inserted with cable for installation. The AFI was intended for installation below the primary and secondary weirs by the Tawe barrage; a location exposed to both the ebb and flow tide of Swansea Bay and the seaward flow of the River Tawe. After two attempts to install the AFI at this location, the AFI was installed in The Prince of Wales Dock (details provided in Appendix 1). The 13.2 m^2 AFI in The Prince of Wales Dock was similarly installed using 6 mm long-link chain however, it was directly attached to crimped wire that ran through the internal structure of the AFI, a large moored buoy and a 250 kg anchor to fix the AFI in position. The AFI installations are referred to throughout this thesis, with data displayed in Chapter 2, 3, 4 and 5 collected concurrently during the AFIs deployment.

Chapter 2: Hydroponically grown halophytes: a comparison of salinity tolerance

Abstract

Halophytes are able to withstand sodium chloride concentrations that 99 % of flora cannot. Therefore, halophytes are of keen interest for planting on artificial floating islands (AFIs) as a method of habitat creation in marine environments. AFIs consist of an integrated connection grid, buoyant matrix and growing medium. The key objective of this study was to compare the plant growth of halophytes, focusing on root establishment through Biohaven® matrix material in both a laboratory and fieldwork experiment. The testable hypothesis was that halophytes present at the lower limit of saltmarsh will grow more successfully via salinity stimulated growth in the high salinity hydroponic treatment, in comparison to halophytes present in the upper limits of saltmarsh. Sea aster (*Tripolium pannonicum*), sea purslane (*Halimione portulacoides*), common cordgrass (*Spartina anglica*), sea rush (*Juncus maritimus*) and common glasswort (*Salicornia europaea*) were transplanted into individual Biohaven® matrix units and hydroponically grown in two salinity treatments (15 and 30). The fresh weight, dry weight, stem, root and leaf length and leaf width of each plant was measured. Two experimental phases were run in spring and summer. In the field, visual counts of the five halophytes were recorded from 1st June 2018 – 2nd May 2019 on three AFIs installed in The Prince of Wales Dock and Swansea Marina. The halophytes present during the successful removal of two AFIs were collected and the fresh weight and dry weight measured for the leaves, stems and roots for comparison. Fresh weight change of the halophytes in the 30 salinity treatment was significantly lower than the 15. Sea purslane was the only halophyte to increase in fresh weight in the 30 salinity treatment in spring. The dry weight of roots protruding outside the matrix for sea purslane was significantly higher than the other halophytes. Root length was also significantly affected by salinity when comparing monocotyledons and dicotyledons species. This could be due to the low Na^+/K^+ ratio associated with the exclusion of Na^+ and Cl^- ions by monocotyledons species in comparison to dicotyledons. There was a significantly more sea rush growing on the AFI present in the low salinity environment in comparison to the four other halophytes. Based on these results, the hypothesis was rejected and sea purslane was recommended for future AFI installations in high salinity environments.

2.1 Introduction

Halophytes account for 1 % of flora and can be defined as plants that are able to ‘complete a life cycle in salt concentrations of at least 200 mM sodium chloride’ (Flowers *et al.*, 1986). This is the equivalent of 11.69 on the dimensionless practical salinity scale (PSU) (Lewis, 1980; Perkin & Lewis, 1980; Solan & Whiteley, 2016). They grow in a range of habitats including mangrove forests, tidal saltmarshes and estuaries associated with transitional and coastal water bodies (Flowers & Colmer, 2015). Due to a variety of adaptive mechanisms exposure to fluctuating salinity conditions is a subsidy for halophytes, that successfully maintain nutrient uptake, growth and reproduction rate under otherwise stressful conditions for non-adaptive species (Elliott & Quintino, 2007; Solan & Whiteley, 2016). This effectively reduces competition and is referred to as the stress-subsidy gradient (Odum *et al.*, 1979b). Non-adaptive species or glycophytes are unable to regulate their internal osmotic pressure effectively in response to ambient salinities equal to 3.5 PSU (Waisel, 1972), resulting in ion toxicity and an inability to perform vital biological processes (Mittler, 2002). As a result, salinity can indirectly control the structure of plant communities and the boundaries of species distribution based on individual stress tolerance (Pennings *et al.*, 2005; Solan & Whiteley, 2016).

In saltmarsh communities differing exposure to physical and geochemical stress including flooding and salinity, plus the competitive abilities of individual species have been identified as key factors controlling zonation across elevational gradients (Chapman, 1974; Pennings *et al.*, 2005; Perillo *et al.*, 2018). Salinity exposure can vary as a result of precipitation, evapotranspiration, local tidal regimes and anthropogenic pressures such as abstraction and contaminant loading (Elliott & Whitfield, 2011; Wolanski & Elliott, 2015; Solan & Whiteley, 2016). In order to understand stress tolerance and the unique adaptive mechanisms associated with halophytes, studies have examined several aspects of their physiology including photosynthetic rate (Lovelock & Ball, 2002), responses to oxidative stress (Jithesh *et al.*, 2006), flooding tolerance (Flowers & Colmer, 2008), growth in highly saline soils (Boesch *et al.*, 1994; Yeo, 1998; Zhu, 2001; Aslam *et al.*, 2011) and their application in the treatment of aquaculture wastewater (Quintã *et al.*, 2015; De Lange & Paulissen, 2016). The latter studies have largely focused on species including common glasswort (*Salicornia europaea*) and sea aster (*Tripolium pannonicum*) and how they respond to aquaculture wastewater that has been directly incorporated into the growing medium (Quintã *et al.*, 2015; De Lange & Paulissen, 2016).

Therefore, to assess the use of halophytes in artificial floating islands (AFIs) further research is required on halophytes grown in differing salinities within a hydroponic system.

2.2 Halophytes

Salt tolerant plants have been recognized and described since 1563, however, not until 1809 did Pallos produce the term 'halophyte', grouping these highly specialized plants together (Waisel, 1972). Chemopodiaceae consists of the largest number of halophytes with 550 species included in this family of angiosperms and less than 5 % placed in additional family groups; Poaceae, Fabeaceae and Asteraceae (Aronson, 1985; Aslam *et al.*, 2011). Research on the physiology of halophytes remained limited until the 1970s, with little known about their adaptive mechanisms and unique physiology (Waisel, 1972; Flowers *et al.*, 1977; Flowers & Colmer, 2008). Knowledge gained on salinity tolerance in plants in recent years has been driven by the requirement to understand natural and anthropogenic processes causing increased soil salinity, as estimates suggest up to 50 % of arable land will be affected by 2050 threatening global food supply and agricultural profits (Wang *et al.*, 2003; Butcher *et al.*, 2016).

Generally, plants respond to salinity via avoidance, resistance or tolerance (Waisel, 1972). They have three response levels: cellular, tissue and the whole plant. The basic mechanism of salt tolerance involves the restricted accumulation and sequestration of inorganic ions, which allows the individual to maintain their internal osmotic balance against heightened external salinities (Flowers & Yeo, 1986; Aslam *et al.*, 2011). Salt tolerance can vary between halophyte species depending on the extent to which ions are accumulated (Munns, 2002; Neves *et al.*, 2007). For example, obligatory halophytes only grow in saline soils and exhibit salinity stimulated growth, preferential halophytes exhibit salinity stimulated growth in saline soils but also grow in non-saline environments and facultative halophytes grow optimally in non-saline soils (Chabreck, 1984).

When exposed to a saline medium, the response of halophytes can largely be associated with the regulation and compartmentalization of Na^+ and Cl^- ions, causing fluctuations in the sodium/potassium ratios (Na^+/K^+) (Flowers & Colmer, 2015; Bose *et al.*, 2015). This is caused by an increase in Na^+ accumulation, in order to maintain water potential gradients for effective plant growth and water uptake (Flowers *et al.*, 1977; Gorham *et al.*, 1980; Neves *et al.*, 2007). The extent to which Na^+ is accumulated in the plant can vary according to whether it is a monocotyledon or dicotyledon species, with the former being highly selective for K^+ ion uptake (Albert, 1975; Flowers & Colmer, 2008; Flowers *et al.*, 2015). These biochemical processes can also have a knock on effect on other macronutrients, such as calcium (Ca) and magnesium (Mg), which have largely been studied in relation to growth deficiencies (Gul *et al.*, 2000) and enzyme activity (Neves *et al.*, 2007; Bose *et al.*, 2015). In addition to high salinity tolerance, emergent halophyte species in the lower saltmarsh successfully grow in reduced soils caused by regular flooding seaward of the mean high water (Armstrong *et al.*, 1985; Colmer & Flowers, 2008; Perillo *et al.*, 2018).

2.3 Aims and Objectives

The aim of this study was to determine which of the five selected halophyte species could be recommended for planting in AFIs installed in saline environments. The testable hypothesis was that halophytes present at the lower limit of saltmarsh will grow more successfully via salinity stimulated growth in the high salinity hydroponic treatment, in comparison to halophytes present in the upper limits of saltmarsh. The root growth of each halophyte was specifically of interest, as quick establishment is important for long term vegetative cover and roots add complexity to the structure that could support aquatic invertebrate and fish communities. In order to test the hypothesis, five halophyte species were exposed to two salinity treatments in a laboratory experiment and during field installations with the following objectives:

1) To measure and compare plant growth via fresh weight change at the beginning and end of the experiment.
2) To compare the dry weight of stems, leaves and roots inside and protruding outside of the matrix at the end of the experiment.
3) To determine the root/shoot (R/S) ratio at the end of the experiment.
4) To compare plant establishment in the AFI by measuring root length outside (underneath) the matrix at the end of the experiment.
5) To measure and compare leaf length change.
6) To compare the number of halophytes of each species that grew successfully on the AFIs during installation in The Prince of Wales Dock and Swansea Marina and the dry weight of the leaves, stems and roots of halophytes collected during the AFIs removal.

The information gained from this study will aid future research and projects that require vegetative cover on AFIs installed in saline environments.

2.4 Materials and Methods

2.4.1 Laboratory Experiment

The halophyte species were selected for this experiment based on their presence at different elevational gradients in saltmarsh habitat present in south Wales, performance in salinity tolerance research (Lv *et al.*, 2012; Quintã *et al.*, 2015; De Lange & Paulissen, 2016) and recommendations provided by Frog Environmental for planting in AFIs in saline environments (Frog Environmental, 2017). This included species naturally present in the lower (common glasswort and common cordgrass, *Spartina anglica*), middle (sea aster and sea purslane, *Halimione portulacoides*) and upper (sea rush, *Juncus maritimus*) elevations of temperate saltmarsh. The plants were collected from Llanrhidian Marsh, situated west of the village of Crofty, south Wales (51°64'N, -4°14'E). In order to account for seasonal bias on plant growth, the experiment was split into two phases: spring (16th April – 15th June 2018) and summer (1st August – 26th September 2018). For the spring and summer experiment, 12 similarly sized individuals of sea aster, sea purslane, common cordgrass and sea rush were collected on 2rd April 2018 and 18th July 2018 respectively. Due to the absence of common glasswort in April, it was only included in the summer experiment. Plants were collected on the same day in preparation for each experiment, to minimise differences in salinity exposure as a result of rainfall and evaporation in the saltmarsh.

2.4.1.1 Plant Species Ecology

From the dandelion and daisy family (*Asteraceaea*) sea aster is the only herbaceous perennial of this family found seaward of mean high water in saltmarsh habitats. As a hemicryptophyte, its overwintering buds are positioned at soil level and it has little vegetative spread capacity (Clapham *et al.*, 1942). The plant is semi succulent with a range of wide and narrow lanceolate leaves and a salinity tolerance of 5 on the Ellenburg indicator values (Clapham *et al.*, 1942). Sea purslane is a perennial that forms part of the Goosefoot family (*Chenopodiaceae*). It is an evergreen shrub typically found fringing intertidal pools that are largely inundated during high tide (Andrades-Moreno *et al.*, 2013). They are also described as phanerophyte as the overwintering buds are located above ground and exposed (Chapman, 1950). They have a salinity tolerance value of 6 under the Ellenburg indicator values (Chapman, 1950). As a rhizomatous perennial herb, common cordgrass has regularly been used to stabilise wet mudflats (Raybould *et al.*, 1991; Adam, 1993). From the family *Poaceae* common cordgrass was produced by chromosomal doubling from Townsend's cordgrass (*Spartina townsendii*), which is a hybrid of small cordgrass (*Spartina maritima*) and smooth cordgrass (*Spartina alterniflora*) (Ayres & Strong, 2001; Perillo *et al.*, 2018). It is a hemicryptophyte with a salinity tolerance of 7 on the Ellenburg indicator values (Raybould *et al.*, 1991; Adam, 1993). From the family *Juncaceae*, sea rush is also a rhizomatous plant that forms tussocks in the upper saltmarsh margins along the high tide mark (Snogerup, 1993). The tussocks are tightly grouped with slow spreading capacity. It is an herbaceous perennial with a salinity tolerance of 5 on the Ellenburg indicator values like sea aster (Snogerup, 1993). In the family *Amaranthaceae*, common glasswort is an herbaceous annual associated with the lower saltmarsh (Ball & Tutin, 1959; Dalby, 1989). It is also described as a therophyte as it can survive harsh conditions in seed form and has a high salt tolerance, scoring 9 on the Ellenburg indicator values (Ball & Tutin, 1959; Dalby, 1989).

2.4.1.2 Sampling Site

Llanrhidian Marsh is located south of the River Loughor and forms part of the southern side of Bury Inlet. The tide extends from Loughor Bridge to Pontarddulais approximately 6 km away, with high water spring tides controlled by the naturally fluctuating ground height of the eastern saltmarsh (Pye & Blott, 2009). This region of the northern Gower coastline previously consisted of steep sloping cliffs, with the development of Llanrhidian Marsh partly due to the formation of beach and dune systems later in the Holocene era, restricting the mouth of the Inlet (Pye & Blott, 2009). Common cordgrass was introduced to Landimore Marsh, west of Llanrhidian Marsh in 1935 and has since colonised the entire estuary (Pye & Blott, 2009). The sediment is dominated by fine sands of approximately 125 μm grain size (Carling, 2009), allowing easy removal of individual plants in the field.

Individual plants were transplanted into 10 cm diameter pots, using the associated sediment and peat free compost. During a two week acclimatisation period, the plants were watered with mild saline (5) solution to ensure salinity exposure was consistent (Quintã *et al.*, 2015; De Lange & Paulissen, 2016). From this point onwards practical salinity units with dimensionless numbers will be referred to throughout (Lewis, 1982). At the end of the acclimatisation period, six plants of each halophyte species were randomly selected for each experiment; total of 24 plants for the spring experiment and 30 plants for the summer experiment. Each plant was carefully washed to remove any soil and lightly blotted. The fresh weight, stem height, root length, leaf length and width, and the number of stems and roots were all measured and counted for each plant prior to installation. Three individuals of each species were grown hydroponically in a 15 salinity treatment and three individuals in a 30 salinity treatment for each experiment; total of six replicates per species, across the spring and summer experiments.

2.4.1.3 Experiment Preparation

The two experiments were conducted in a laboratory environment to enable control of the temperature, humidity, salinity, nutrient concentrations and light. The air and water temperature varied between 20 – 25 °C. Three VIPARSPECTRA Reflector Series 450 W lights were used in the experiment. The photosynthetically active radiation (PAR) varied from 300 – 2560 μmol m^2 s^{-1} across the length of the plants. This range varied according to the height of the individual plant and therefore, the distance away from the light units. Oxygen saturation was controlled with an Aquarline Hailea Aco-9620 air pump that maintained 70 – 100 % oxygen saturation in each growing container. The mean pH across the two phased experiment was 7.67 and ranged from 6.70 - 8.48. Each plant was transplanted into 8 cm diameter pre-cut holes in the AFI Biohaven® matrix (commercially sold by Frog Environmental), using a mix of washed horticultural grit and peat free compost. The two layered matrix units were approximately 19.5 cm x 19.5 cm x 10 cm. To keep the units buoyant, polystyrene (19.5 cm x 4.5 cm x 5 cm) was attached to two lengths of the matrix using wooden skewers (Figure 2.1).

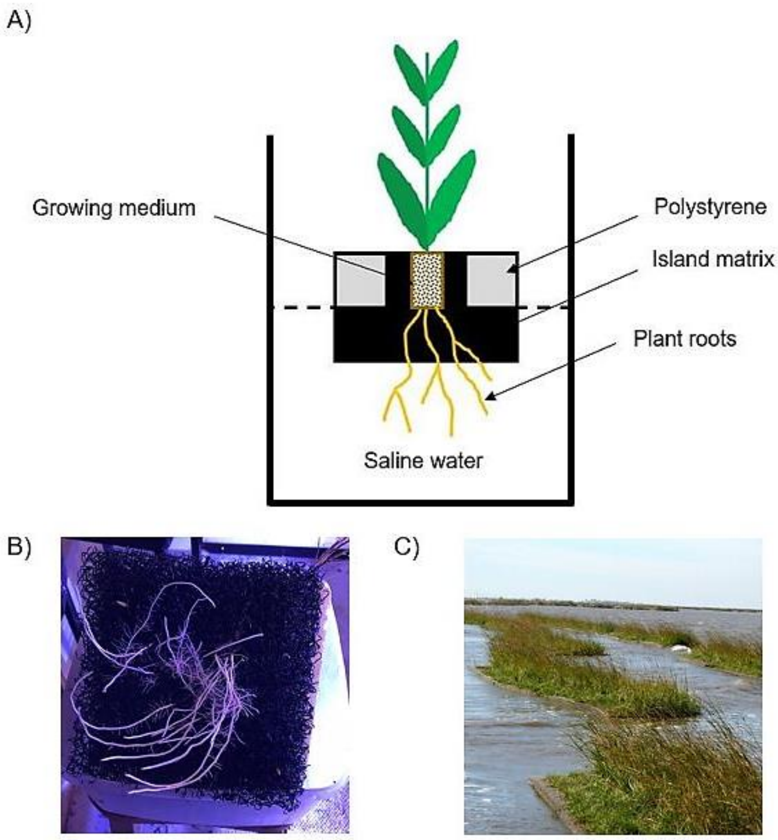


Figure 2.1: A) Schematic diagram of the Biohaven® island matrix units used in this study. Each unit consisted of two layers of non-woven recycled plastic matrix, polystyrene providing buoyancy and horticultural grit mixed with peat free compost, as a growing medium. Individual halophytes were transplanted into one unit and exposed to 5 L of saline water (15 or 30 salinity treatment), which was added to a 10 L container. B) Sea rush roots that grew through the matrix unit during the summer experiment. C) Artificial floating islands installed for wave absorption (Frog Environmental, 2016a).

The matrix units were placed in 10 L containers with 5 L of saline water. On the basis that the seawater had approximately 150 µmol/L of total organic nitrogen, 630 µL of Ionic Hydro Grow Nutrient Solution was added at the beginning of the experiment, resulting in a total organic nitrogen concentration of 450 µmol/L in 5 L. This was calculated using % weight/volume provided with the product. For the plants grown in 15 salinity treatment, 800 µL of the solution was used. The nutrient solution was added on the first and fourth week of the experiment, to ensure that nutrient availability was not a limiting factor on plant growth.

2.4.1.4 Data collection

Biometric measurements (stem height, number of stems, root length of the roots protruding outside of the matrix, number of roots protruding outside of the matrix, leaf length and width) and factors affecting water chemistry (water temperature, pH and redox potential) were monitored once per week. Air temperature was recorded daily. At the end of the eight week experiment, the fresh weight of each plant was measured for the total plant, leaves, stems, protruding roots outside of the matrix and for the root mass inside the matrix. Each plant was then dried at 60 °C for 48 hours and the same characteristics measured for the dry weight (Dodkins & Mendzil, 2015).

2.4.2 Field Experiment

Two, 8 m² AFIs (Biohavens®) were installed on 28th and 29th September 2017; one located in Swansea Marina and one in The Prince of Wales Dock. During deployment of the AFI in Swansea Marina the water salinity ranged from 9 – 15.67. The water body is subject to freshwater input from the River Tawe and oil spill contamination from recreational boats in the marina. In The Prince of Wales Dock the water salinity ranged from 28 – 32.25 with freshwater input limited to rainfall. Two weeks prior to the installations 32 similarly sized plants of sea aster, sea purslane, common cordgrass, sea rush and common glasswort were collected from Llanrhidian marsh and transplanted into 9 cm diameter biodegradable coir pots; 16 plants of each species were planted on each 8 m² AFI. Each plant was regularly watered with mildly saline (5) water. A 13.2 m² AFI was also installed on 17th May 2018 in The Prince of Wales Dock. As this AFI was initially proposed for installation in a tidal location 48 individual plants of sea aster, sea purslane, common cordgrass and sea rush were pre-grown in 9 cm diameter biodegradable coir pots from seeds to allow the roots to penetrate through the coir pot. Due to the late germination of common glasswort, 48 coir pots with multiple seeds were prepared. Each plant was regularly watered with mildly saline (5) water. The coir pots were inserted into the AFI matrix units on the day of installation. While monitoring the AFIs visual counts of the halophytes growing on the matrix were completed from 1st June 2018 – 2nd May 2019 (The Prince of Wales Dock, n = 10; Swansea Marina, n = 10).

The large (13.2 m²) AFI in The Prince of Wales Dock was removed successfully on 3rd June 2019 and the AFI in Swansea Marina on 5th June 2019. Due to the weight of blue mussels (*Mytilus edulis*) on the small (8 m²) AFI in The Prince of Wales Dock, the matrix units split while attempting to lift the AFI out of the water. This prevented the collection of halophytes from the small AFI in The Prince of Wales Dock. The plants present in the matrix of the large

AFI in The Prince of Wales Dock and AFI in Swansea Marina were removed carefully and stored in sample bags. The wet weight of each plant was measured and subdivided into leaves, stems and roots. All of the plants were dried at 60 °C for 48 hours and the dry weight of the leaves, stems and roots was measured.

2.4.3 Statistical Analysis

Prior to statistical analysis, the data were tested for normality using the Anderson – Darling normality test and equal variance via the Levene's test. Data were square-root transformed where necessary to achieve equal variance. In order to test the hypothesis of this study, the response of the five halophytes to the two salinity treatments was compared between individuals of each species and between species. The relationship between the dry weight of the stems, leaves and roots inside and protruding outside of the matrix of each species at the end of the experiment and the two salinity treatments was tested by applying binomial generalised linear models, followed by a Tukeys pairwise comparison test to compare species. For non-parametric data, Mann-Whitney U Test and Kruskal Wallis followed by Nemenyi's multiple comparisons testing was used to compare the change in fresh weight, root length, leaf length and R/S across all species and salinity treatments. These non-parametric tests were also used on the halophyte count data collected from the field experiment. Data analysis was conducted in R 3.5.1. Statistics Software, Minitab 18 (Minitab Ltd, Coventry, United Kingdom) and PRIMER 6 (PRIMER-e, Auckland, New Zealand). Prior to analysis in PRIMER, the biometric dataset was normalised and a multidimensional scaling plot (MDS) on Euclidian distance were produced. ANOSIM was also calculated to compare biometric variables based on species and salinity.

2.5 Results

2.5.1 Laboratory Experiment

2.5.1.1 Fresh Weight Change

The fresh weight change of the halophytes was significantly lower in the 30 salinity treatment than the 15 salinity treatment; when pooling data from the spring and summer experiments (Kruskal Wallis, $<p = 0.001$, $n = 6$). Sea purslane was the only species during the spring experiment to increase in mean fresh weight, for both salinity treatments (Figure 2.2). In the 15 salinity treatment, the mean fresh weight change was 330.40 ± 65.94 % and 38.81 ± 25.28 % in the 30 salinity treatment.

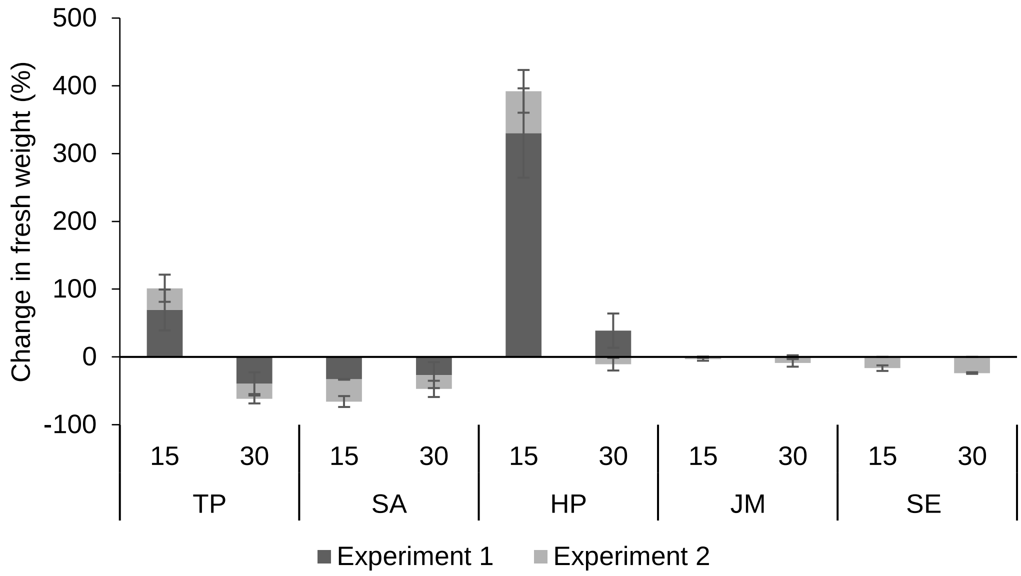


Figure 2.2: Mean change in total fresh weight (%) of sea aster (TP), common cordgrass (SA), sea purslane (HP), sea rush (JM) and common glasswort (SE) during the spring (16th April – 15th June 2018) and summer (1st August – 26th September 2018) experiments for the 15 and 30 salinity treatments (n = 3, mean ± standard error; common glasswort only for summer experiment).

In the summer experiment, sea purslane and sea aster had a significantly lower fresh weight in the 30 salinity treatment in comparison to the 15 salinity treatment (Mann-Whitney U Test, p = 0.041; p = 0.002, n = 3). Sea purslane and sea aster were also significantly higher in fresh weight change in comparison to common cordgrass in the 15 salinity treatment (Nemenyi's multiple comparison, p = 0.002; p = 0.005, n = 3).

2.5.1.2 Dry Weight

For the spring experiment, sea purslane roots protruding out of the matrix accounted for 11.04 ± 1.08 % of the overall dry weight in the 15 salinity treatment and 1.16 ± 0.48 % in the 30 salinity treatment (ANOVA, p = 0.002, n = 3). Stem dry weight of sea purslane also varied between salinity treatments; 41.69 ± 3.77 % in the 15 salinity treatment and 60.33 ± 3.22 % in the 30 (ANOVA, p = 0.039, n = 3). During the summer experiment, there were no significant differences in stem, leaf and root dry weight between the two salinity treatments. When pooling data from the spring and summer experiments, sea purslane had a significantly higher overall dry weight of roots protruding outside of the matrix in comparison to sea aster, common cordgrass, sea rush and common glasswort (ANOVA and Tukey's pairwise comparison, p = 0.024; p = 0.015; p = 0.001; p = 0.019, n = 6; Figure 2.3). In addition, the stem dry weight of sea purslane was also significantly higher in the 30 salinity treatment in comparison to the 15 (ANOVA, p = 0.047, n = 6).

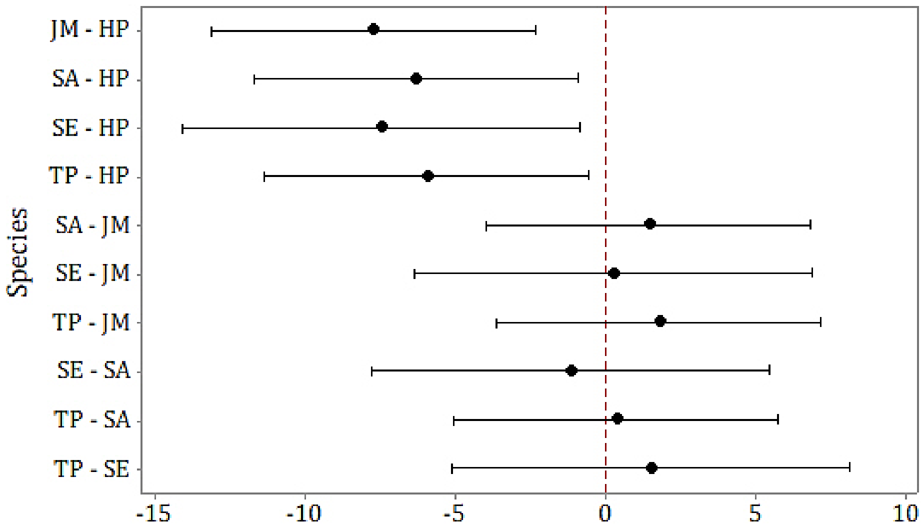


Figure 2.3: Tukey's pairwise comparison test illustrating differences in the mean root dry weight protruding out of the matrix between sea aster (TP), sea purslane (HP), common cordgrass (SA), sea rush (JM) and common glasswort (SE, n = 3). Intervals not containing zero correspond to means that are significantly different (95 % confidence intervals). Data pooled from both experiments (n = 6).

2.5.1.3 Root/Shoot

When calculating the R/S, the roots inside and outside of the matrix were included. The R/S of sea purslane and sea aster in the higher salinity treatment was significantly lower than sea rush in the higher salinity treatment (Kruskal Wallis pairwise comparisons using Tukey and Kramer, $p = 0.003$ and $p = 0.014$, n = 6). In the spring experiment, sea purslane had a R/S of 0.10 in the high salinity treatment and sea rush had a R/S of 14.43, indicating that sea purslane had a lower overall root dry weight.

2.5.1.4 Root Length

When comparing the root length of plants in the 15 and 30 salinity treatment during both experiments, the root length was significantly shorter in the 30 salinity treatment (Kruskal Wallis, $p = 0.005$, n = 6). Common cordgrass also had a significantly shorter root length in the lower salinity treatment than sea purslane during both experiments (Kruskal Wallis pairwise comparisons using Tukey and Kramer, $p = 0.025$, n = 6). Focusing on the spring experiment, sea aster and sea purslane had grown roots through the AFI matrix by week one. The mean root length of sea aster in the high salinity was 7.87 ± 6.43 cm and the mean root length of sea purslane was 20.76 ± 7.43 cm. The mean root length of sea aster in the low salinity treatment was 17.17 ± 11.02 cm and the mean root length of sea purslane was 31.3 ± 3.56 cm. In comparison, common cordgrass and sea rush had grown roots through the matrix by week three.

During the summer experiment, all five halophytes had grown roots through the AFI matrix material by the end of the first week, except sea rush in the 30 salinity treatment (Figure 2.4; Figure 2.5). Sea asters mean root length in the 15 salinity treatment was 13.01 ± 2.46 cm and 6.79 ± 0.70 cm in the 30 salinity treatment. This constitutes a 48.8 % decline in root length in the 30 salinity treatment, but this result was not significant (Mann-Whitney U Test, p = 0.1, n = 3).

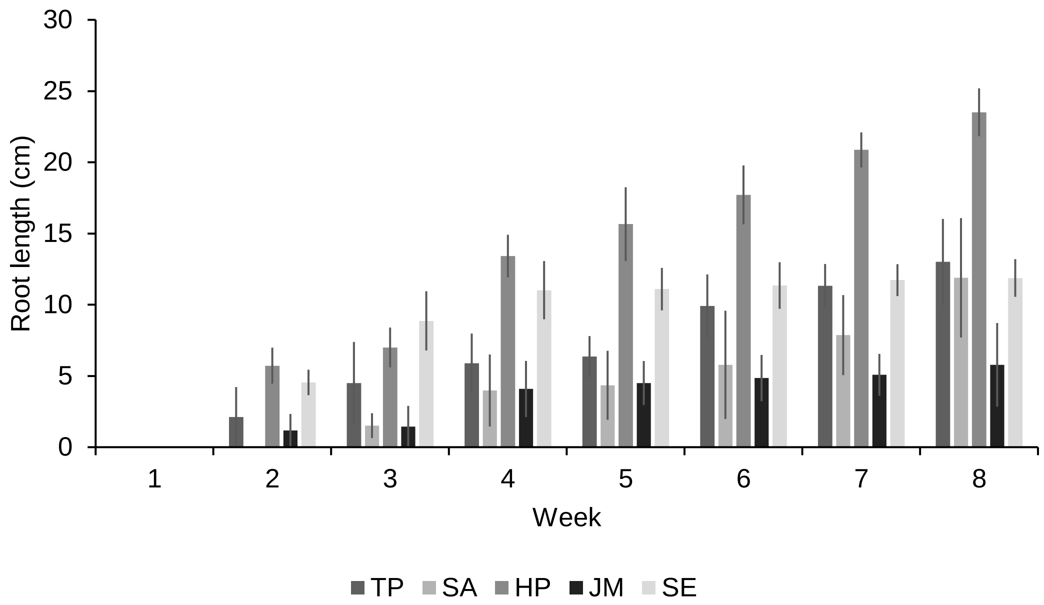


Figure 2.4: The mean root length of sea aster (TP), common cordgrass (SA), sea purslane (HP), sea rush (JM) and common glasswort (SE) measured each week during the summer experiment, in the 15 salinity treatments (n = 3).

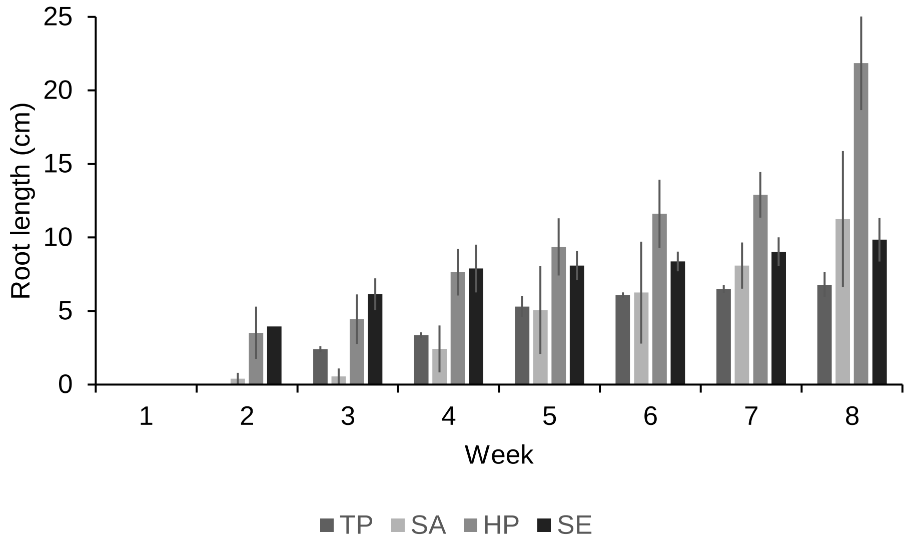


Figure 2.5: The mean root length of sea aster (TP), common cordgrass (SA), sea purslane (HP) and common glasswort (SE) measured each week during the summer experiment, in the 30 salinity treatments (n = 3).

2.5.1.5 Leaf Length

Overall leaf length was not significantly affected by the salinity treatments, including data for sea aster, sea purslane and common cordgrass (Kruskal Wallis, $p = 0.128$, $n = 6$). Analysing data from the spring experiment, the leaf length change between treatments and species was significantly different (Kruskal Wallis, $p = 0.021$, $n = 3$). This was due to comparisons between leaf length change of sea aster in the low salinity treatment and sea purslane in the high salinity treatment (Kruskal Wallis pairwise comparisons using Tukey and Kramer, $p = 0.017$, $n = 3$). Therefore, there was no marked difference in leaf growth as a result of the salinity treatments when comparing individuals of the same species.

2.5.1.6 Combined Biometric Variables

Utilising the biometric data that was collected for sea aster, sea purslane and common cordgrass including leaf width, stem, leaf and root length protruding outside the matrix, there were clear dissimilarities between the three species (ANOSIM, sea aster – common cordgrass, $R = 0.486$; sea aster – sea purslane, $R = 0.562$; common cordgrass and sea purslane, $R = 0.833$, $p = 0.01$, $n = 6$; Figure 2.6).

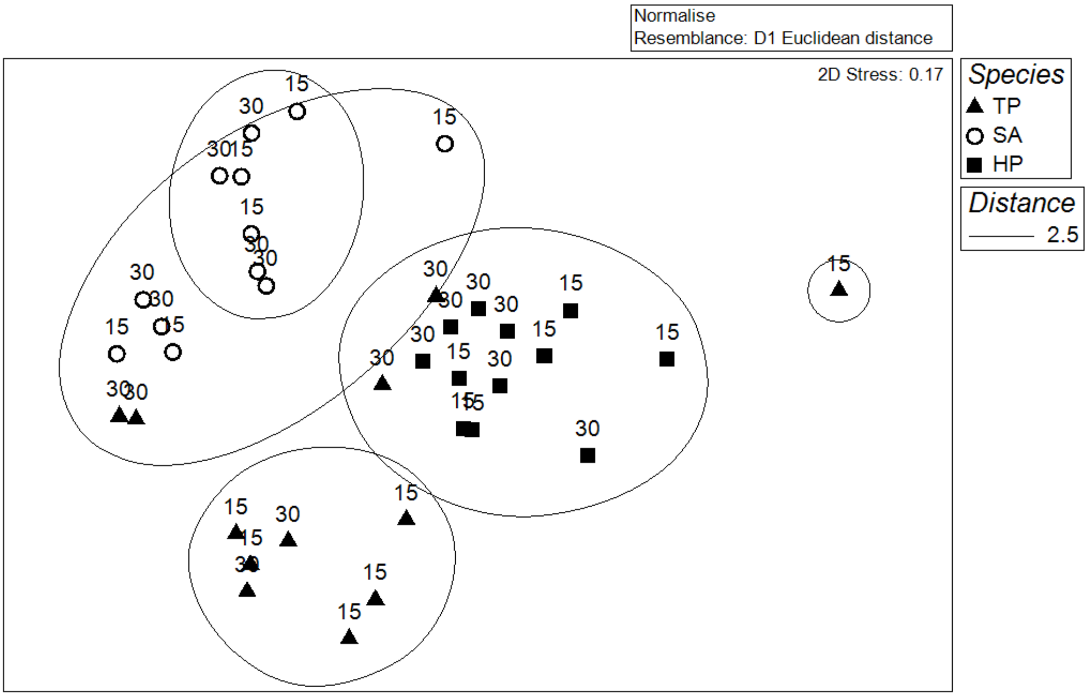


Figure 2.6: Multidimensional scaling plot of Euclidean distances between sea aster (TP), common cordgrass (SA) and sea purslane (HP) based on multivariate biometric characteristics (leaf width, stem, leaf and root length) measured in the 15 and 30 salinity treatments. Data were normalized prior to analysis and shows data from week eight in the spring and summer experiment (n = 6).

As sea rush and common glasswort lack easily measurable and comparable leaves, these species were not included in this analysis. The MDS showed that salinity did not control differences in biometric variables when comparing the same species and that this was driven by physiological differences of each species (Figure 2.6).

2.5.2 Field Experiment

2.5.2.1 Visual Counts

There was a significantly more halophytes growing on the 8 m² AFI in Swansea Marina in comparison to the 8 m² AFI in The Prince of Wales Dock (Mann Whitney U Test = 568, p <0.001; n = 10; Figure 2.7). Sea rush had significantly more individual plants than common glasswort, sea aster and sea purslane on the AFI in Swansea Marina (Kruskal Wallis = 20.504, df = 4, p < 0.001; Nemenyi multiple comparison, sea rush and common glasswort, p <0.001; sea rush and sea aster, p = 0.002, sea rush and sea purslane, p = 0.023; n = 10). On average, 13.4 ± 0.43 individual sea rush plants were growing on the AFI in Swansea Marina in comparison to 4.6 ± 1.99 common glasswort, 4.9 ± 0.72 sea aster and 6.6 ± 1.25 sea purslane. Common cordgrass had an average of 7.2 ± 1.17 plants (mean ± standard error). The final halophyte count on 2nd May 2019 recorded 15 sea rush, eight common cordgrass, six sea aster and six sea purslane plants growing on the AFI.

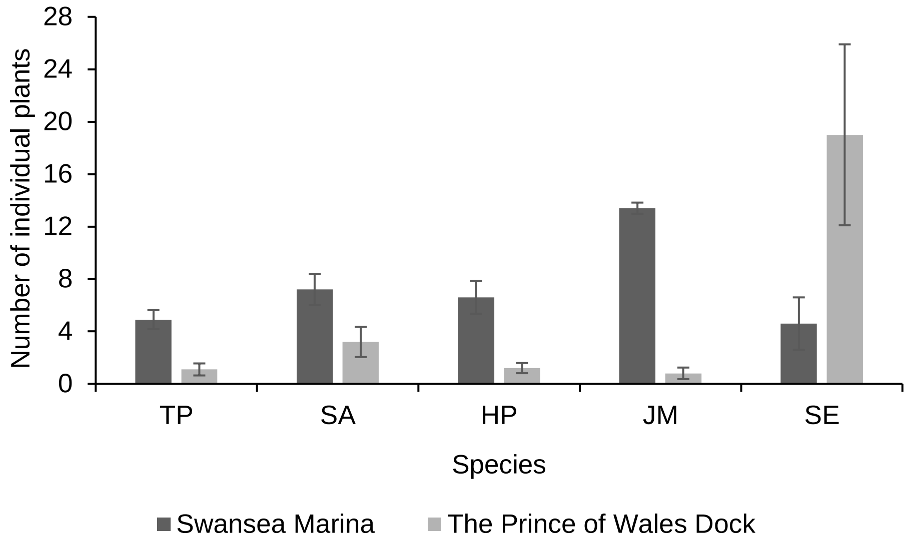


Figure 2.7: The average number of plants growing on the 8 m² artificial floating island in Swansea Marina and The Prince of Wales Dock from 1st June 2018 – 2nd May 2019 (n = 10).

There was no significant difference between the number of individual plants of each species on the small AFI in The Prince of Wales Dock (Kruskal Wallis = 4.187, df = 4, p = 0.381). On average 19 ± 6.90 common glasswort, 3.2 ± 1.15 common cordgrass and 1.2 ± 0.39 sea purslane individual plants were present on the small AFI in The Prince of Wales Dock (mean ± standard error). Common glasswort re-germinated in late May 2018; 50 individual plants were recorded in July (Figure 2.8). By 6th February 2019 there were no plants present on the small AFI in The Prince of Wales Dock. On 13th November 2018 two sea purslane plants and one common cordgrass plant was recorded.

Figure 2.8: *Left* – The large artificial floating island (AFI) in The Prince of Wales Dock on 1st June 2018. *Middle* – The small AFI in The Prince of Wales Dock on 25th July 2018 with blue mussel (*Mytilus edulis*) shells and plant growth dominated by common glasswort. *Right* – Common glasswort growth on the small AFI in The Prince of Wales Dock on 1st June 2018.

On the large AFI in The Prince of Wales Dock there was significantly less common glasswort than the four remaining halophytes transplanted, as common glasswort failed to germinate (Kruskal Wallis = 22.93, df = 4, $p < 0.001$; n = 10). However, there were no significant differences between the halophytes that successfully grew on the AFI. There was an average of 33.6 ± 6.19 sea purslane, 31.3 ± 6.87 sea rush, 30.7 ± 7.20 sea aster and 24.8 ± 7.74 common cordgrass (mean ± standard error). During the final count on 2nd May 2019, 18 sea purslane, three sea aster and two sea rush individual plants were recorded.

2.5.2.2 Dry weight

During the removal of the large AFI in The Prince of Wales Dock on 3rd June 2019 three of the key species assessed as part of the laboratory experiment were still growing on the AFI: sea purslane, sea rush and sea aster. Common saltmarsh grass (*Puccinellia maritima*) was also present however, common cordgrass and common glasswort were both absent. Sea purslane was the dominant species on the AFI contributing 87.18 % of the total dry weight collected followed by common saltmarsh grass (Table 2.1). The stems of sea purslane were the heaviest feature when compared with the leaves and roots. The salinity in The Prince of Wales Dock ranges from 28 – 32.25 (Appendix 3). During the removal of the AFI in Swansea Marina on 5th June 2019 four of the key species assessed as part of the laboratory experiment were still growing on the AFI: sea rush, common cordgrass, sea aster and sea purslane. Common saltmarsh grass, creeping saltbush (*Atriplex prostrata*), sea plantain (*Plantago maritima*) and lesser swine-cress (*Lepidium didymium*) were also present. Sea rush was the dominant plant species on the AFI contributing 69.08 % of the total dry weight collected followed by common saltmarsh grass (Table 2.1). The roots of sea rush accounted for most of the dry weight in comparison to the stems. The salinity in Swansea Marina ranged from 9 – 15.67 (Appendix 3).

Table 2.1: A comparison of the leaf, root and stem dry weights of sea purslane, sea rush, common saltmarsh grass and sea aster collected from the large artificial floating island in The Prince of Wales Dock on 3rd June 2019. Additionally, for comparison the leaf, root and stem dry weights of sea rush, common saltmarsh grass, common cordgrass, sea aster, creeping saltbush, sea plantain and sea purslane collected from the artificial floating island in Swansea Marina on 5th June 2019.

Site/Species	Dry weight (g)			
Prince of Wales Dock	Leaf	Stem	Root	Total
Sea purslane	23.67	123.57	29.9	177.14
Common saltmarsh grass	N/A	11.39	2.22	13.61
Sea rush	N/A	7.39	1.78	9.17
Sea aster	0.91	0.65	1.71	3.27
Swansea Marina				
Sea rush	N/A	94.74	324.07	418.81
Common saltmarsh grass	N/A	72.33	16.34	88.66
Common cordgrass	13.36	4.97	32.42	50.75
Creeping saltbush	5.78	8.65	3.36	17.79
Sea aster	7.34	5.57	4.16	17.07
Sea plantain	N/A	7.16	1.09	8.25
Sea purslane	2.66	1.61	0.54	4.81
Lesser swine-cress	0.04	0.10	0.03	0.17

2.6 Discussion

This study aimed to test the hypothesis that halophytes present in the lower limit of temperate saltmarsh (common glasswort and sea aster) would grow more successfully via salinity stimulated growth in a high salinity hydroponic treatment, in comparison to halophytes present in the upper limit of temperate saltmarsh (sea rush). This was achieved by measuring changes in fresh weight, dry weight and biometric variables of five halophytes that colonise different elevations of temperate saltmarsh, when grown hydroponically in a low and high salinity treatment. In addition, the five halophyte species were transplanted and grown on AFIs installed in a high (The Prince of Wales Dock) and low (Swansea Marina) salinity environment in heavily modified coastal water bodies. The number of individual plants were counted during the course of the deployment and collected once removed, to compare the dry weight of the leaves, stems and roots. The data collected from this study will aid future research and projects that require vegetative cover on AFIs installed in saline environments.

2.6.1 Laboratory Experiment

2.6.1.1 Fresh Weight Change

Salinity as well as season were identified as factors impacting on the fresh weight change of sea purslane and sea aster. During the summer experiment both halophyte species significantly increased their fresh weight in the 15 salinity treatment but decreased in the 30, suggesting that they are either obligatory or preferential halophytes (Chabreck, 1984). The overall growth of sea purslane in a previous study increased when exposed to salinities up to 12, before declining thereafter (Redondo-Gomez *et al.*, 2007; Benzarti *et al.*, 2014b) and decreased in fresh weight when exposed to saline solutions between 1.5 – 3 % concentrate (Ramani *et al.*, 2006a), supporting the results of this experiment. A key adaptive mechanism of sea purslane when exposed to high salinity is the excretion of salt through epidermal bladders on the upper and lower surface of their leaves, preventing accumulation in young tissues (Jensen, 1985; Freitas & Breckle, 1992a). More than 80 % of ions present in the leaves of sea purslane are stored in epidermal bladders avoiding excess ion accumulation (Freitas & Breckle, 1992b). As a halophyte often lining channels and pools inundated during high tide in the mid-region of temperate saltmarsh (Redondo-Gomez *et al.*, 2007), sea purslane is well adapted to fluctuating environmental conditions including salinity and water exposure (Beeftink, 1977; Freitas & Breckle, 1992b). Unlike sea purslane, sea aster has no morphological characteristics that enable it to exclude salt and therefore, has developed biochemical adaptive mechanisms of accumulating ions in vacuoles and osmolytes (Ramani *et al.*, 2006b). The species is present in the mid-region of temperate saltmarsh and subject to similar fluctuating conditions to sea purslane.

In addition to their adaptive mechanisms, *Atriplex* species have been recognised as nitrophilic (Osmond *et al.*, 1969; Smirnoff & Stewart, 1985) and sea purslane has exhibited a higher salinity tolerance with increasing nitrate availability (Jensen, 1985). As nitrogen availability was not a limiting factor in this experiment, nutrient supply could have aided growth for sea purslane via increasing salinity tolerance. Comparing the fresh weight change in spring and summer, sea purslane grew more in the spring experiment in comparison to the summer. In natural saltmarsh, the maximum growth rate of sea purslane is from winter – spring, declining in summer due to increasing temperatures, radiation and declines in water availability (Neves *et al.*, 2007). As a perennial, seasonal fluctuations in a controlled laboratory experiment were not expected however, potential differences in exposure to environmental conditions prior to collection could have influenced the results of this experiment.

2.6.1.2 Dry Weight

The dry weight of sea purslane roots protruding through the AFI matrix was significantly higher compared to the four other halophytes. In previous experiments, sea purslane has increased in root dry weight more rapidly in comparison to *Limoniastrum monopetalum,* that allocated more dry weight to other aerial plant components (Neves *et al.*, 2007). Sea purslane also had a higher shoot dry weight in the 30 salinity treatment. This stimulated plant growth via moderate salinity exposure, can result in 30 % higher whole plant dry weight values of sea purslane than individuals grown in freshwater (Redondo-Gomez *et al.*, 2007; Benzarti *et al.*, 2012, 2014a). The stimulated plant growth in Redondo-Gomez *et al*. (2007) correlated with fluctuations in photosynthetic rate, by the regulation of stomatal conductance and CO_2 concentrations in the leaves (Redondo-Gomez *et al.*, 2007; Benzarti *et al.*, 2014a).

Sea aster has shown low tolerance for waterlogged conditions, with significantly smaller roots than other halophytes, which was associated with an increase in iron and manganese concentrations in the plant shoots (Cooper, 1982). Common cordgrass as a species associated with the lower saltmarsh is adapted to waterlogged conditions due to their well-developed root system and high oxygen transportation capacity, however, the halophyte has exhibited lower growth rates in waterlogged soils (Holmer *et al.*, 2002).

Juncus species have been associated with high concentrations of proline (Boscaiu *et al.*, 2013), the most common osmolyte, to maintain osmotic balance in response to salinity exposure and decreased in height and total biomass with salinities >10 (Greenwood & MacFarlane, 2009). Common glasswort has an optimal growth and photosynthetic rate at salinities between 6 – 23, accumulating a high concentration of Na^+ ions in the cell vacuoles of the shoot endodermis (Lv *et al.*, 2012). However, due to the short life span of common glasswort, once the plants had flowered in late August individuals in this experiment quickly became dry. The results of this study are therefore a misrepresentation of their ability to grow successfully in the AFI matrix.

2.6.1.3 Root/Shoot (R/S)

Overall, the salinity treatments affected the R/S of the halophytes, with variations in R/S between dicotyledon species (sea aster, sea purslane and common glasswort) compared to monocotyledon (common cordgrass and sea rush). Dicotyledon species tend to contain higher

Na^+ concentrations and have a lower selectivity for K^+ ions, whereas monocotyledons tend to accumulate lower concentrations of Na^+. For example, the average Na^+/K^+ ratios of the monocotyledon species were six fold lower than the dicotyledons, indicative of the exclusion of Na^+ and Cl^- ions and accumulation of K^+ ions (Gorham *et al.*, 1980; Gil *et al.*, 2014). This salt induced nutritional imbalance is likely to have resulted in differences between the R/S of the halophytes in this study. The root growth of sea aster has previously declined in response to increased salinity resulting in a lower R/S ratio than plants grown in a low salinity (Montfort & Brandrup, 1927; Adam, 1993). This was coupled with a stimulation in shoot growth. In contrast, species such as big cordgrass (*Spartina cynosuroides*) and smooth cordgrass (*Spartina alterniflora*) exposed to high salinities experienced reduced shoot growth and had a high R/S (Parrondo *et al.*, 1978; Adam, 1993). As Na^+ and Cl^- ion exposure influences the uptake of other essential macronutrients, salinity can affect the allocation of carbohydrates to aerial parts of a plant, impacting on the growth of stems, leaves and roots (Adam, 1993).

2.6.2 Root Length

Overall, root length of the plants was significantly shorter in the 30 salinity treatment in comparison to the 15, with sea purslane and common cordgrass most affected in relation to root length outside the matrix. Sea purslane produced the longest roots in both salinity treatments. *Atriplex* species form dense low growing foliage that sprout thin roots across the stem structure (Paraskevopoulou *et al.*, 2015) and have proven to be resilient to stressful conditions including non-uniform salinity exposure (Bazihizina *et al.*, 2009). For example, when exposed to both 0.6 and 40 salinity treatments within a hydroponic system, old man saltbush (*Atriplex nummularia*) was able to maintain a stable photosynthetic rate plus root and stem growth (Bazihizina *et al.*, 2009). In addition, sea purslane successfully established roots through a shallow green roof system with a depth of 10 cm, highlighting the species ability to grow in shallow, artificial substrata (Paraskevopoulou *et al.*, 2015).

Common cordgrass in temperate saltmarsh forms dense monospecific swards via rapid growth of thick rhizomes and roots (Institute of Terrestrial Ecology, 1990), that grew through the AFI matrix in both salinity treatments during the summer experiment. This could be due to the large air ducts that transport oxygen through aerenchyma lacunae, allowing the species to grow extensive and well aerated root systems under submersion (Waisel, 1972; Howes & Teal, 1994).

2.6.2.1 Leaf Length

The leaf length of sea aster, sea purslane and common cordgrass was not significantly affected by salinity in both experiments. For sea aster and sea purslane, this could be due to the compartmentation of Na^+ and Cl^- ions into the cell vacuole (Munns, 2002). For example, Na^+

ion concentrations in sea purslane have reached 14.6 mg g^{-1} during the summer (Neves *et al.*, 2007). In order to maintain a lower water potential inside the plasmalemma and prevent osmotic flux of water out of the cells, organic osmolytes are formed in the cytosol (Ramani *et al.*, 2006a; Burg & Ferraris, 2008). Adaptive mechanisms such as this may have reduced the potential impact of NaCl exposure on leaf growth and photosynthetic rate.

2.6.3 Field Experiment

2.6.3.1 Visual Counts

Overall, a greater number of halophytes were able to successfully grow on the AFI in the low salinity environment of Swansea Marina. This was also demonstrated by sea purslane and sea aster in the laboratory experiment. However, sea rush and common cordgrass were the dominant halophytes on the AFI in the low salinity environment; the two monocotyledon species. The rigid nature of sea rush stems may be able to withstand regular use of the AFI by large wildfowl such as mute swans (*Cygnus olor*). In addition, both halophytes have thick rhizomes and root systems that provide a strong anchor into the substratum once established. On the small AFI in high salinity environment of The Prince of Wales Dock there was no halophyte growth during the last four months of its deployment and when it was removed. This was partly due to the weight of blue mussels that had fouled underneath the AFI causing the matrix to sit low in the water and left vulnerable to overtopping by the surrounding waterbody. The majority of the coir matting and therefore, plant growth was removed during the winter period. Common glasswort did, however, re-germinate in Spring 2018 highlighting the halophytes potential as vegetative cover on AFIs installed in high salinity environments.

The germination of common glasswort on natural saltmarsh is inhibited by high salinities and generally in European habitats occurs in early spring (Ajmal Khan & Weber, 1986; Singh *et al.*, 2014). Excluding common glasswort that failed to germinate on the large AFI, four halophytes established in the high salinity environment of The Prince of Wales Dock which was installed eight months after the small AFI and had at thicker matrix that sat higher on top of the waterbody. The difference in season that the AFIs were deployed (8 m^2 AFIs in September and 13.2 m^2 in May), variations in float height on the water as a result of the thicker structure of the 13.2 m^2 and fluctuations in environmental conditions could have influenced the successful establishment of each halophyte species based on their individual growth phenology.

The impact of heavy biofouling on the underside of AFIs and on the installation chain in highly productive environments must be controlled for AFIs to provide a sustainable and long term ecological engineering solution. Regular cleaning of the installation chain should be implemented as part of a maintenance plan and consideration of the matrix buoyancy required, as this can be altered prior to installation. The buoyancy of Biohavens® for example is added to the plastic matrix via pumped closed cell polyurethane foam which can be mediated based on the buoyancy required.

2.6.3.2 Dry Weight

In the high salinity environment of The Prince of Wales Dock four species were collected on the large AFI after its 13 month deployment. Notably sea purslane contributed the most to the total dry weight of plants collected from the AFI which typically colonises the middle section of temperate saltmarsh. This may be due to stimulated plant growth as a result of the high salinity also observed during the laboratory experiment (Redondo-Gomez *et al.*, 2007; Benzarti *et al.*, 2012, 2014a). In the low salinity environment of Swansea Marina, eight species were collected on the AFI after its 20 month deployment. Sea rush contributed the most to the total dry weight of plants collected which typically colonises the upper sections of temperate saltmarsh. The dry weight of the roots accounted for the majority of the plants weight. Although all the plants were washed carefully to remove excess substratum attached to the roots, it was particularly difficult to remove the thick clay that enveloped sea rush from Llanrhidian marsh. Common saltmarsh grass also germinated on both AFIs and therefore, should be considered for future AFI installations in saline environments.

2.7 Conclusion

This laboratory experiment was conducted in order to determine which halophyte species or combination of species, would be recommended for vegetated AFI deployments in enclosed, saline environments. Based on the results of this experiment the hypothesis can be rejected, as common glasswort and sea aster that colonise the lower limits of temperate saltmarsh did not demonstrate more successful growth in high salinity environments in comparison to halophytes that colonise middle or upper regions. Sea purslane outperformed the other halophytes in the spring experiment, as it was the only species to increase in fresh weight in both salinity treatments. Sea purslane also established quickly into the matrix material and had the longest average root length by the end of each experiment. Therefore, sea purslane could add complexity to the underside of the AFIs in high salinity environments via root growth, potentially creating habitat for aquatic invertebrates and shelter for fish populations. Future studies should focus on planting common glasswort with an improved experimental design that considers seasonal timing and the life cycle of the halophyte. Due to common glassworts high salinity tolerance and visual observations during the experiment, common glasswort may successfully grow in hydroponic saline environments, unlike the results of this study. More research is required on AFIs in exposed saline environments and how fluctuating abiotic factors such as nitrogen availability and dissolved oxygen influence salinity tolerance of halophytes that are hydroponically grown.

Chapter 3: Floating invertebrate oases: characterisation of biofouling communities on artificial floating islands

Abstract

Artificial floating islands (AFIs) are an ecological engineering method used to create habitat, improve water quality and support localised biodiversity in aquatic environments. The overall aim of this study was to investigate the development of biofouling communities on AFIs installed in heavily modified coastal water bodies. The testable hypothesis was that the fouling community assemblages would be distinctly different on the horizontal surface in comparison to the vertical edge. Scrape samples were collected from the 'inner, outer and edge' sections of two AFIs installed in The Prince of Wales Dock and Swansea Marina, to compare the fouling community assemblages at the end of their deployment period. The AFI installed in The Prince of Wales Dock was exposed to salinities ranging from 28 – 32.25 and was fouled by 20 taxa; 15 of which were recorded on the edge of the AFI. Blue mussels (*Mytilus edulis*) were the dominant epibenthic species, with significantly larger individuals sampled in the inner and outer sections, in comparison to the edge. The AFI had significant differences in community assemblage in the inner and outer sections, in comparison to the edge, which was controlled by the abundance of *Balanus crenatus*, *Jassa marmorata* and sea vast tunicate (*Ciona intestinalis*). Japanese skeleton shrimp (*Caprella mutica*) were also present. The AFI in Swansea Marina was exposed to salinities ranging from 9 – 15.67 and was fouled by a total of 9 taxa, with Australian tubeworms (*Ficopomatus enigmaticus*) dominating the samples. The community assemblages were also significantly different in the inner and outer sections in comparison to the edge, which was controlled by the abundance of bay barnacles (*Amphibalanus improvisus*) and *Melita palmata*. Therefore, the hypothesis was accepted and the study concluded that AFIs have the potential to support biodiverse fouling communities via the primary settlement of ecosystem engineers. For future AFI installations in heavily modified coastal water bodies, it is recommended that a management plan is implemented to monitor and clean the installation chain when required, while retaining the secondary reef feature on the underside of the AFI. In addition, a Biosecurity Risk Assessment should be produced before installation to ensure the AFI does not facilitate the spread of non-indigenous species.

3.1 Introduction

Biofouling refers to the colonisation of artificial structures such as buoys, pontoons, pilings and revetments by micro or macroorganisms in freshwater, brackish and marine environments (Melo & Bott, 1997). In a marine context, once a clean surface is immersed a conditioning layer of dissolved organic material will coat the structure (Taylor *et al.*, 1997) followed by microorganisms, phytoplankton and larvae creating a biofilm (Callow & Callow, 2002; Salama *et al.*, 2018). Once a biofilm has been established a macrofouling community may develop consisting of 'soft' (algae, soft corals and sponges etc.) or 'hard' (barnacles, mussels and tubeworms etc.) fouling organisms (Callow & Callow, 2002; Yan *et al.*, 2009). The assemblage of biofouling communities can vary according to pre and post-settlement processes (Fraschetti *et al.*, 2002; Oricchio *et al.*, 2016). Pre-settlement referring to the survival and dispersal of larvae controlled by current regimes, water chemistry and predation (Oricchio *et al.*, 2016). The physical characteristics of the structure itself such as surface complexity, colour, spatial orientation and fixture design are also important in determining the colonisation of different epibiotic species (Holloway & Connell, 2002). Post settlement processes refer to the long term variability of abiotic factors such as light availability, water chemistry and hydrodynamics (Perkol-Finkel *et al.*, 2006a; Glasby *et al.*, 2007). These directional changes in community composition over time as a result of disturbance events is the process of succession, from early colonisation in the initiation state to its equilibrial climax community (Greene & Schoener, 1982; Sandin & Sala, 2012).

The development of secondary reefs on artificial structures increases nutrient concentrations in the water column as biofouling invertebrates defecate, enriching localised communities (Langhamer, 2010; Coates *et al.*, 2014; Nall *et al.*, 2017). The potential growth of macroalgae and colonisation of invertebrates also provide feeding sites for predators (Lubbers *et al.*, 1990) and refuge sites for prey, reducing predation risk (Irlandi *et al.*, 1995; Clynick *et al.*, 2008). However, in many cases artificial structures form dissimilar community assemblages in comparison to natural habitats, affecting localised species interactions and potentially larger scale trophic dynamics (Ambrose & Anderson, 1990; Nall *et al.*, 2017). It is therefore important to gain an understanding of the costs and benefits on the ecological and socio-economic system when introducing artificial structures into the marine environment (Elliott *et al.*, 2016).

Community assemblages on artificial structures can vary based on the season it was deployed (Rajagopal *et al.*, 1997) and the duration of submersion, which can affect recruitment of larvae and colonisation (Satheesh & Wesley, 2011). In the Skagerrak Sea the temporal recruitment of biofouling organisms on artificial panels was examined with blue mussels (*Mytilus edulis*) dominating initial recruitment in June, followed by bay barnacles (*Amphibalanus improvisus*) in August (Berntsson & Jonsson, 2003). The impact of heavily colonised artificial structures on ecohydrology, which refers to the physical conditions of the system such as hydrography and sedimentology should also be considered (Elliott *et al.*, 2016). For example, mussel farms produce large quantities of pseudo-faeces that will deposit on the seabed particularly in low current conditions (Crawford *et al.*, 2003) and can accumulate in the infrastructure of the farm, potentially creating anoxic conditions affecting water quality (Longdill *et al.*, 2007).

The depth of substratum in the water column and therefore, exposure to swash and light intensity variations (Kennelly, 1989) have also been highlighted as fundamental factors controlling the assemblage of biofouling communities (Glasby & Connell, 2001; Holloway & Connell, 2002). Floating devices exposed to high current velocities in Gulf of Eilat, Red Sea were fouled by ascidians, sponges and bivalves, in contrast to fixed structures deeper in the water column that were largely colonised by algae and corals (Perkol-Finkel *et al.*, 2006a), supporting previous studies (Holloway & Connell, 2002). In addition, the Pelamis P2 wave energy converter was deployed in Orkney, Scotland and monitored to assess the development of biofouling communities across different sub sections of the device (Nall *et al.*, 2017). Similarly to the floating devices in the Red Sea, scrape samples collected from the shallow sections of the Pelamis were dominated by algae species including *Ulva* and *Polysiphonia* and deeper sections were colonised by suspension feeders such as blue mussels, *Balanus crenatus* and European sea squirt (*Ascidiella aspersa*) (Nall *et al.*, 2017). When designing anthropogenic devices for deployment in aquatic environments, the potential impact of biofouling requires assessment, as it can reduce the efficiency of the device and potentially prevent it from functioning, increasing maintenance costs.

Artificial floating islands (AFIs) are an ecological engineering method which have largely been installed in freshwater habitats for water treatment (Lu *et al.*, 2015; Xie *et al.*, 2019a), to create new patch habitats (Overton *et al.*, 2015) and for aesthetic benefits, with limited installations in marine environments. They broadly consist of a buoyant mat, integrated connection grid, substratum and transplanted vegetation, suitably selected for the chosen location (Burzaco & Frog Environmental, 2016). The recycled plastic matrix and submerged roots of vegetated AFIs provide a structurally heterogeneous surface, encouraging the colonisation of micro and macroorganisms; these include ‘soft’ and ‘hard’ fouling species (Callow & Callow, 2002; Yeh *et al.*, 2015). AFIs installed in heavily modified coastal water bodies, substantially changed by morphological alteration could be a practical restoration tool to enhance benthic communities and increase their ecological potential (Borja & Elliott, 2007; Temino-Boes *et al.*, 2018; Buffagni *et al.*, 2019). Like pontoons, AFIs have vertical and horizontal attachment sites that may create two distinct biofouling communities (Pomerat &

Reiner, 1942; Bassindale *et al.*, 1948) due to fluctuating hydrodynamics and physcio-chemical conditions (Eckman, 1983; Glasby & Connell, 2001; Perkol-Finkel *et al.*, 2006a) and grazing preferences of local predators (Mook, 1981; Oricchio *et al.*, 2016).

Therefore, it is important to gain more information on: the biofouling communities that colonise AFIs in saline environments such as heavily modified coastal water bodies; the higher trophic level consumers AFIs could support; and variations in biofouling across the AFI that could potentially affect the long term stability of the installation and impact on ecohydrology within the system.

3.2 Aims and Objectives

The aim of this study was to investigate the development of biofouling communities on AFIs installed in heavily modified coastal water bodies. The testable hypothesis was that the fouling community assemblages of AFIs is distinctly different on the horizontal surface in comparison to the vertical edge. In order to test the hypothesis, scrape samples were collected from two AFIs with the following six objectives:

1) To characterise the benthic community colonising AFIs.
2) To compare the invertebrate community assemblages fouling the inner and outer horizontal surface and the vertical edge of the AFI.
3) To compare invertebrate communities colonising AFIs in brackish (Swansea Marina) and fully saline (The Prince of Wales Dock) aquatic ecosystems.
4) To compare size and dry weight of blue mussels that had fouled the inner and outer horizontal surface and vertical edge of the AFI in The Prince of Wales Dock.
5) To make recommendations on how to manage biofouling on AFIs in heavily modified coastal water bodies, in order to ensure that the buoyancy of the AFI is not compromised.

3.3 Materials and Methods

On 3rd and 5th June 2019, the 13.2 m² AFI in The Prince of Wales Dock and the 8 m² AFI in Swansea Marina were successfully removed and vertically lifted out of the water using cranes. This allowed the AFIs to be laid upside down for scrape sampling.

3.3.1 Scrape Sampling

Scrape samples were collected at the end of the AFIs deployment in The Prince of Wales Dock and Swansea Marina. As different sections of the AFIs may provide alternative habitats for biofouling species, scrape samples were collected from the inner and outer horizontal surface and vertical edge of each AFI. The outer section was defined as the 70 cm radius of the AFI (Figure 3.1). The inner section consisted of the remaining area in the centre and the edge was the vertical surface around the perimeter. Ten, 10 cm x 10 cm scrape samples were taken from the inner, outer and edge sections, resulting in a total of 30 samples per AFI and 60 samples in total. Due to the thickness of blue mussel growth on the AFI in The Prince of Wales Dock, a corer was used to collect samples. A small quadrat and scraping tool were used to collect samples from the AFI in Swansea Marina.

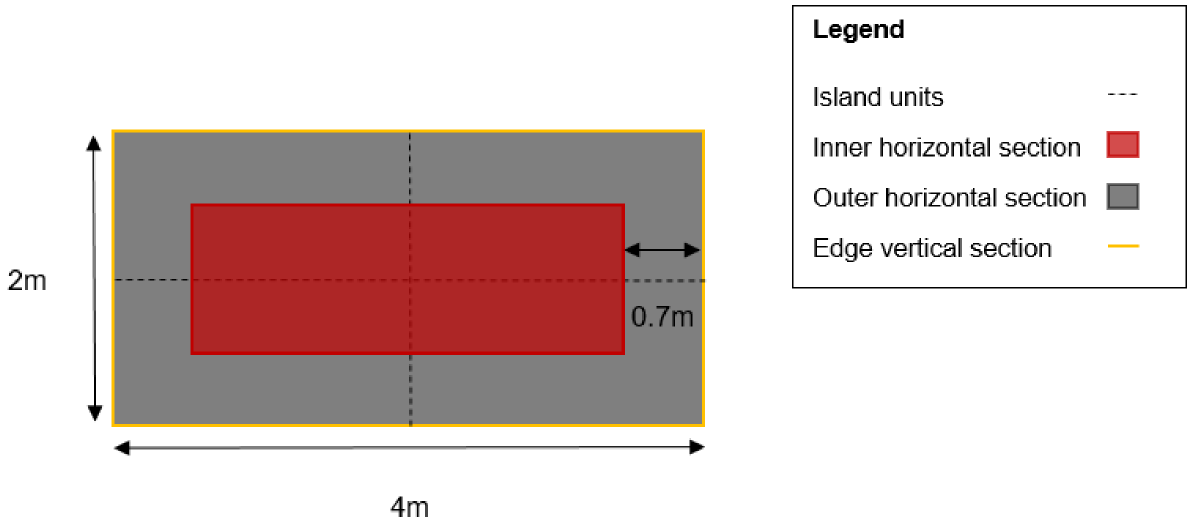


Figure 3.1: Schematic diagram showing the inner, outer and vertical sampling sections of the artificial floating islands (AFIs). The dimensions of the 8 m² AFI was used to demonstrate the sampling methodology of this study.

3.3.2 Sample Processing

Once back at the laboratory, the samples collected from the AFI in Swansea Marina were immediately preserved in 70 % ethanol. For samples taken from the large AFI in The Prince of Wales Dock, blue mussels were firstly counted and separated from any other taxa, which were immediately preserved in 70 % ethanol. Ten mussels were randomly selected from each sample, weighed, measured (length and width) and dried for 72 hours at 60 °C. Once the samples were dry the total dry weight, soft tissue and shell weight were recorded. All of the biota from each sample was examined under a microscope and identified to species level where possible.

3.3.3 Statistical Analysis

PRIMER v6 with PERMANOVA was used to assess the biofouling assemblages on the AFI. A square root transformation was applied on the species abundance data and Bray-Curtis similarity matrices were constructed for The Prince of Wales Dock and Swansea Marina datasets. PERMANOVA was used to test for significant differences in the community assemblages between the inner, outer and edge sections of the deployed AFIs and SIMPER provided information on the species contributing the most to identified dissimilarities between each section. This analysis was used to test the hypothesis that the community assemblages would be distinctly different on the horizontal surface in comparison to the vertical edge. One-way ANOVA and Tukey multiple comparisons test was used to determine if there was a significant difference in the abundance, length and width of blue mussels sampled in the inner, outer and edge sections of the AFI in The Prince of Wales Dock. Kruskal Wallis and Nemenyi's multiple comparison was used to determine if there was a significant difference in the dry weight of blue mussels in the inner, outer and edge section of the AFI as the data were non-parametric. Data analysis on the blue mussel datasets was conducted in R.3.5.1 Statistics Software.

3.4 Results

3.4.1 The Prince of Wales Dock

The entire underside of the AFI was extensively biofouled during installation in The Prince of Wales Dock. In total, 20 taxa were recorded across the three sections of the AFI, which included six crustaceans, two polychaetes, an ascidian, a calcarea, a bryozoan, an arachnid and an insect larvae (Figure 3.2). Eight algal species were also recorded including six green (*Cladophora sericea*, *Blindingia minima*, *Ulva compressa*, *Ulva intestinalis*, *Rhizoclonium riparium* and *Chaetomorpha ligustica*), a brown (channelled wrack, *Pelvetia canaliculata*) and a red algal species (*Ceramium secundatum*). A total of 10 taxa were sampled in the inner section, 11 in the outer and 15 at the edge, averaging at 4.8 ± 0.42 taxa per 100 cm^{-2} sample (mean ± standard error; n = 30). The total abundance of invertebrates was 120.7 ± 17.56 in the inner section, 104.8 ± 10.75 in the outer and 64.9 ± 8.15 at the edge (mean ± standard error; n = 10 for each section; Figure 3.2).

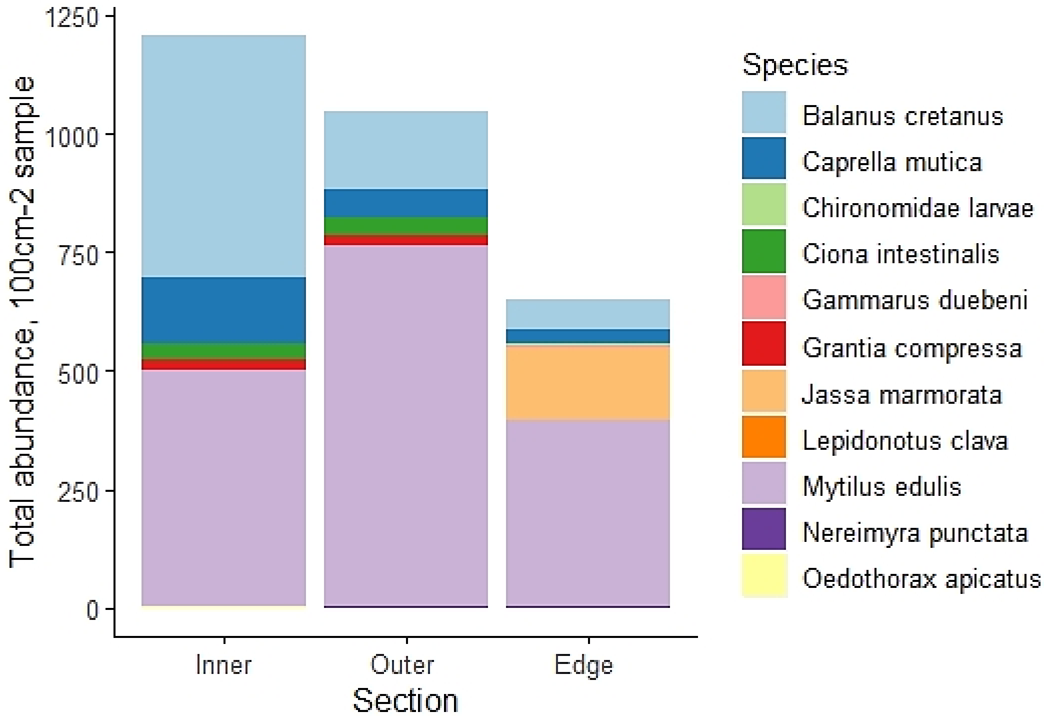


Figure 3.2: Total invertebrate abundance of the inner, outer and edge 100 cm^{-2} samples (n = 10) collected from the artificial floating island in The Prince of Wales Dock. As abundance was not recorded for algal species and bryozoans, they are not included in this figure.

The community assemblages of invertebrate species were significantly different at the inner and outer section, in comparison to the edge (PERMANOVA, inner and edge, $p = 0.002$; outer and edge, $p < 0.001$; Figure 3.3). The abundance of *B. crenatus* and blue mussels contributed the most to the dissimilarities between the inner and edge section (SIMPER, 22.37 % and 21.28 % respectively); overall dissimilarity of 57.83 %. The abundance of blue mussels and *Jassa marmorata* contributed the most to the dissimilarities between the outer and edge section (SIMPER, 27.91 % and 18.56 % respectively); overall dissimilarity of 54.23 %.

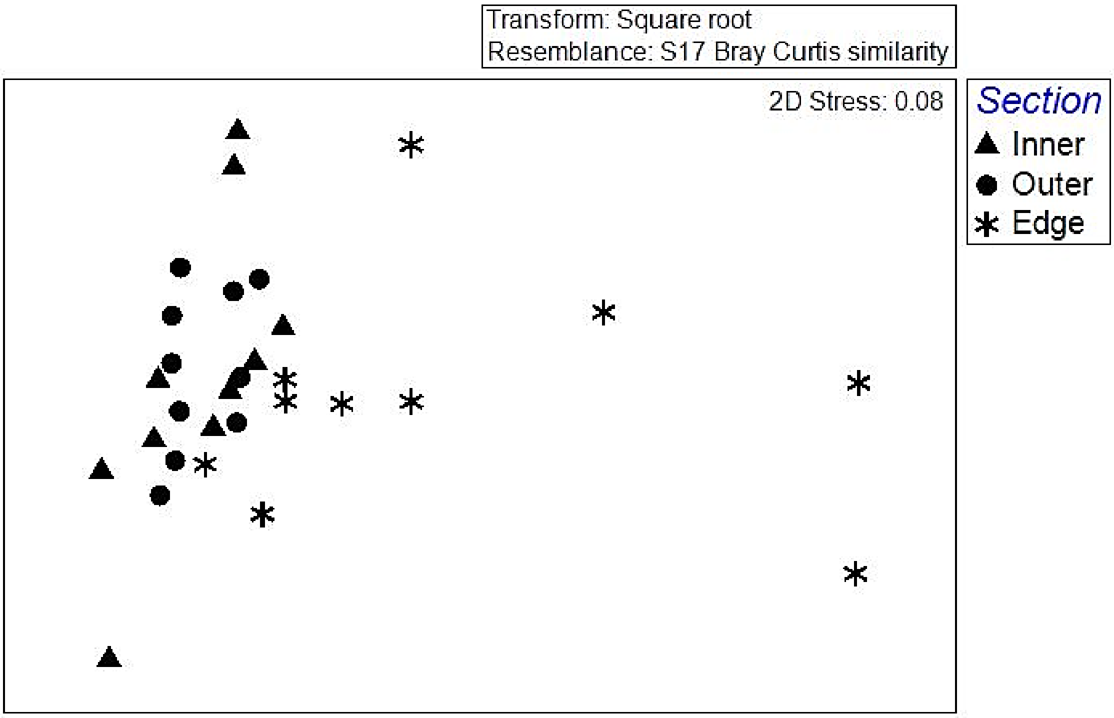


Figure 3.3: Multidimensional scaling plot of Bray Curtis similarity between the community assemblages (invertebrate abundance) at the inner, outer and edge section of the large artificial floating island in The Prince of Wales Dock (n = 10).

Blue mussels, *B. crenatus*, Japanese skeleton shrimp (*Caprella mutica*), sea vase tunicate (*Ciona intestinalis*) and *Nereimyra punctata* were recorded in all three sections of the AFI, with the two latter species in low abundance (Figure 3.2). Japanese skeleton shrimp is an invasive non-indigenous species in Europe. Both purse sponge (*Grantia compressa*) and sea vase tunicate established in the inner and outer sections of the AFI. Of the eight algal species identified on the AFI, seven were found at the edge including *B. minima* and *R. riparium*, which were found in 50 % of the edge samples.

3.4.1.1 Biofouling Succession

Using the remote underwater video footage collected as part of Chapter 4, the succession of visible macrofouling organisms underneath the AFIs could be established (Figure 3.4). This descriptive analysis excluded invertebrates <1 cm such as *B. crenatus*. Sea vase tunicate was the first invertebrate recorded fouling the underside of the small AFI, captured via Go Pro footage from April – June 2018. Two months later in July 2018, blue mussels now dominated under both the large and small AFI, outcompeting sea vase tunicates. By early September, the presence of Japanese skeleton shrimp was observed in the video footage with blue mussels. A second survey in September also revealed the presence of sea vase tunicates and purse sponge as an epibiont on the blue mussels.

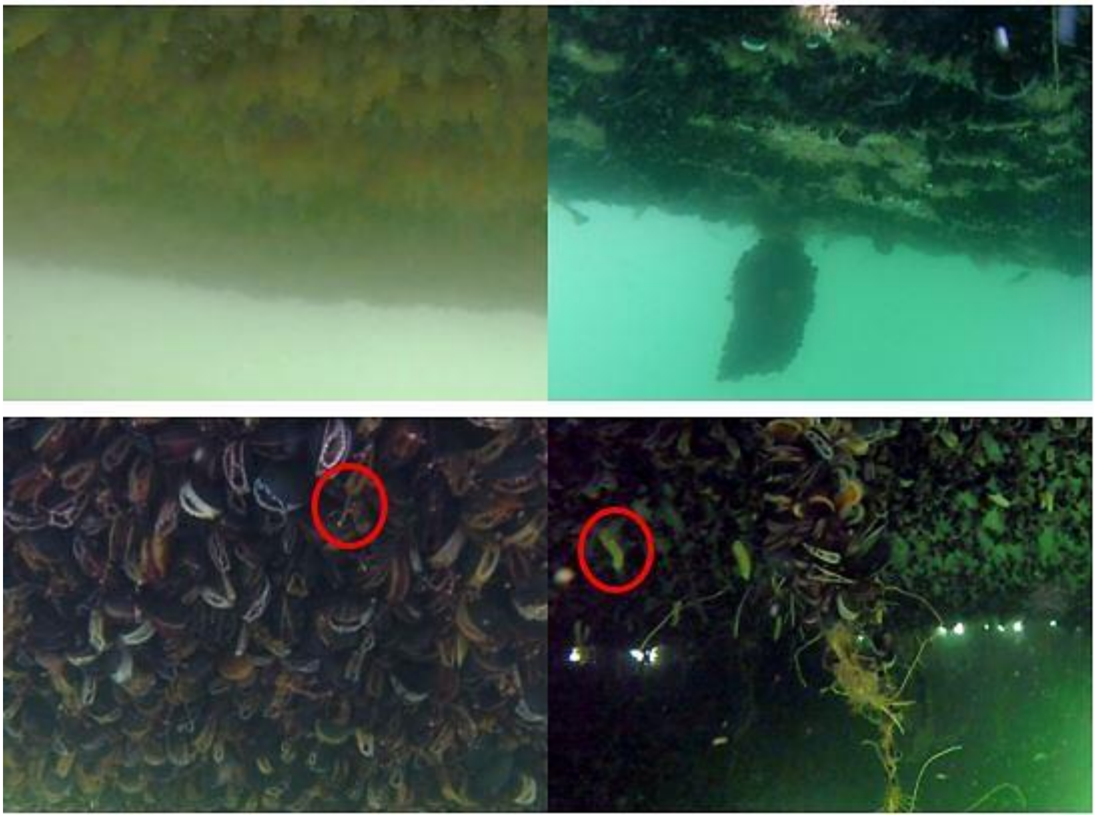


Figure 3.4: Images of the succession of biofouling underneath the artificial floating island (AFI) in The Prince of Wales Dock. Top left; Sea vase tunicate (*Ciona intestinalis*) covering the underside of the small AFI on 24th April 2018. Top right; Blue mussels (*Mytilus edulis*) outcompeting sea vase tunicates by 25th July 2018. Bottom left; Blue mussels covering the large AFI with Japanese skeleton shrimp (*Caprella mutica*) (circled) on 3rd September 2018. Bottom right; presence of purse sponge (*Grantia compressa*) (circled) and sea vase tunicates as epibionts on blue mussels by 17th September 2018.

3.4.1.2 Blue Mussels

There was a significantly higher abundance of blue mussels in the outer section in comparison to the edge (ANOVA, $p = 0.011$, Tukey multiple comparison, outer and inner, $p = 0.011$; $n = 10$; Table 3.1). The length of blue mussels was significantly larger in the inner and outer section of the AFI, in comparison to the edge (ANOVA, $p <0.001$; Tukey multiple comparison, inner and edge, $p <0.001$; outer and edge, $p <0.001$; $n = 10$). Similarly, there was significantly wider mussels present in the inner and outer sections in comparison to the edge (ANOVA, $p <0.001$; Tukey multiple comparison, inner and edge, $p <0.001$; outer and edge, $p = 0.008$; $n = 10$). This significant difference in size was also reflected in the soft tissue dry weight of the samples (Kruskal Wallis = 18, $p = <0.001$; Nemenyi multiple comparison, inner and edge, $p <0.001$; $n = 10$; Table 3.1).

Table 3.1: Comparison of the abundance, size, dry weight and reef height range (lowest - highest reef height measured once scrape samples removed) of blue mussels (*Mytilus edulis*) sampled in the inner, outer and edge sections of The Prince of Wales Dock artificial floating island.

Blue mussels	Inner	Outer	Edge
Abundance (mean, s.e; n = 10)	49.7 ± 7.79	76.1 ± 7.07	39.4 ± 9.59
Length (mean, s.e; n = 100)	4.06 ± 0.12	3.70 ± 0.13	3.01 ± 0.12
Width (mean, s.e; n = 100)	2.08 ± 0.06	1.91 ± 0.06	1.64 ± 0.06
Dry weight of soft tissue (mean, s.e; n = 100)	0.33 ± 0.03	0.20 ± 0.02	0.19 ± 0.02
Dry weight of shell (mean, s.e; n = 100)	1.62 ± 0.13	1.55 ± 0.14	1.35 ± 0.13
Reef height range (cm) Minimum – Maximum	1 - 15.2	1.6 - 13.1	0.1 - 10

3.4.2 Swansea Marina

The entire underside of the AFI in Swansea Marina was also biofouled. In total, nine invertebrate taxa were recorded across the AFI, including four crustaceans, two polychaetes, one bryozoan, one insect larvae and one clitellata species. Six taxa were recorded in the inner section, six in the outer and nine at the edge (Figure 3.5). The total invertebrate abundance per 100 cm^{-2} sample was 275.9 ± 39.45 in the outer section, 201 ± 31.02 in the inner section and 144.9 ± 21.51 at the edge (mean ± standard error; n = 10). The Australian tubeworm (*Ficopomatus enigmaticus*) was the dominant species in the three sections of the AFI and accounted for large differences in species abundance (Figure 3.5). Australian tubeworm is also an invasive non-indigenous species in Europe. *Conopeum seurati* was present in all of the samples but was not counted for abundance.

Figure 3.5: The presence of Australian tubeworm (*Ficopomatus enigmaticus*) in each section of the artificial floating island installed in Swansea Marina. *Left* - sample one collected in the inner section; *middle* – sample one collected in the outer section; *right* – sample one collected in the edge section.

The community assemblages of invertebrate species were significantly different at the inner and outer section, in comparison to the edge (PERMANOVA, $p = 0.003$ and $p < 0.001$ respectively; Figure 3.6 and 3.7). The abundance of Australian tubeworms contributed the most to the dissimilarities in community assemblage of the inner and the edge section (SIMPER, 35.88 %), followed by bay barnacles (SIMPER, 29.85 %); overall dissimilarity of 24.80 %. In the outer and edge section, the dissimilarities in community assemblage were driven by the same species as the inner section, with Australian tubeworms accounting for 37.73 % and bay barnacles, 22.30 %; overall dissimilarity of 26.51 %. The abundance of bay barnacles in the inner and outer section was significantly higher than the abundance in the edge section (Kruskal Wallis = 18.33, inner and edge, $p < 0.001$; outer and edge, $p = 0.006$; $n = 10$; Figure 3.6). *Melita palmata* was present in all sections of the AFI. Chironomidae larvae, Enchytraeidae species and ragworm (*Hediste diversicolor*) were only found in the edge samples and in low abundance.

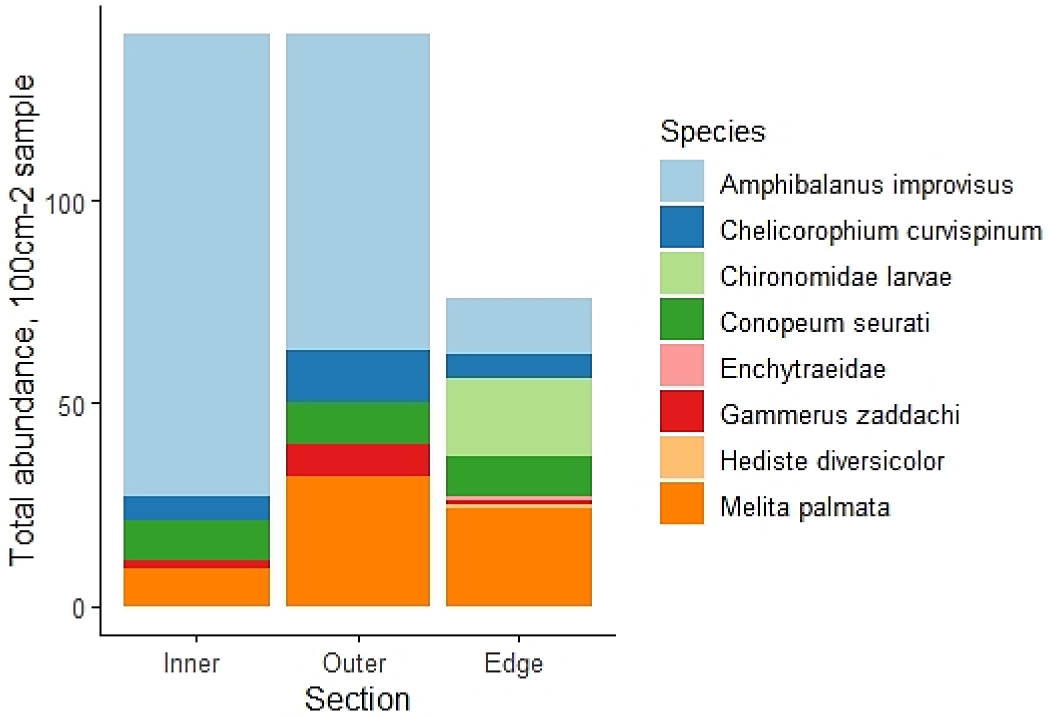


Figure 3.6: Total invertebrate abundance of the inner, outer and edge 100 cm^{-2} samples (n = 10) collected from the artificial floating island in Swansea Marina excluding Australian tubeworm (*Ficopomatus enigmaticus*).

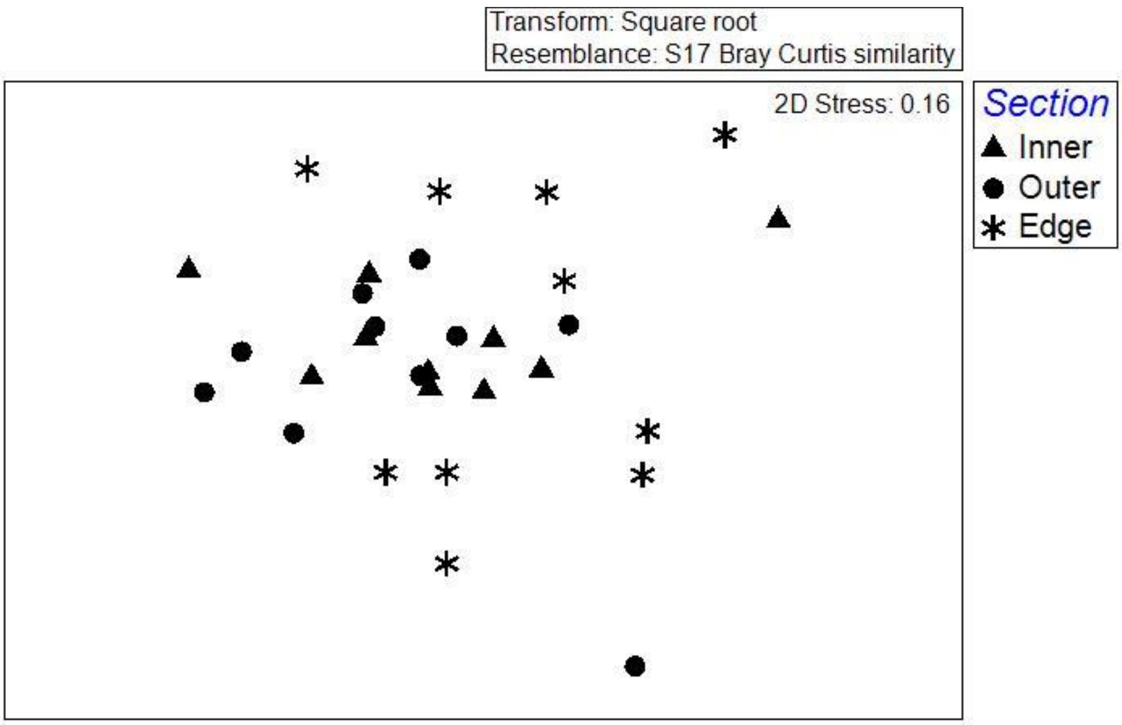


Figure 3.7: Multidimensional scaling plot of Bray Curtis similarity between the community assemblages (invertebrate abundance) at the inner, outer and edge section of the artificial floating island in Swansea Marina (n = 10).

3.5 Discussion

The biofouling communities were significantly different in the inner and outer section of the AFIs in comparison to the edge in The Prince of Wales Dock and Swansea Marina. The biofouling communities assessed at the end of the AFIs deployment was controlled by the colonisation of mussels and tubeworms, also referred to as foundation species (Dayton, 1972; Ellison *et al.*, 2005) or ecosystem engineers (Jones *et al.*, 1994, 1997; Heiman & Micheli, 2010). Ecosystem engineers create biogenic structures such as mussel beds, coral reefs and seagrass meadows that effectively stabilise and diversify the landscape (Dayton, 1972; Jones *et al.*, 1997; van der Zee *et al.*, 2015), facilitating the colonisation of species that depend on them for refuge from predators and potential desiccation, and food supply (Gutiérrez *et al.*, 2003; Donadi *et al.*, 2013; van der Zee *et al.*, 2015). The complex structure of biogenic reefs can also attenuate currents and waves in the intertidal zone, reducing the impacts of erosion on exposed sediment (Koch *et al.*, 2009). Therefore, the AFIs provided an appropriate substratum for colonisation of ecosystem engineers, that via their influence on trophic interactions and exposure to environmental stresses determine the long term development of community assemblages.

3.5.1 The Prince of Wales Dock

Blue mussels were the dominant fouling species on the AFI in The Prince of Wales Dock. The colonisation and growth of bivalves on floating structures is dependent on larval dispersal and fluctuating abiotic factors, including mass transfer rate, current velocity, water temperature and salinity (Wildish & Kristmanson, 1985; Glasby *et al.*, 2007; Oricchio *et al.*, 2016). The early settlement of blue mussels on hard, man-made structures (Joschko *et al.*, 2008) such as offshore gas platforms (Stachowitsch *et al.*, 2002; van der Stap *et al.*, 2016), offshore wind farms (Lindeboom *et al.*, 2011; Krone *et al.*, 2013), wave energy devices (Nall *et al.*, 2017) and fish cages (Greene & Grizzle, 2007) has been frequently recorded. For example, analysis of biofouling communities at offshore gas platforms in the southern North Sea discovered that blue mussels were present largely on platforms closer to the shore at 0 – 20 m depths, in high currents where food was abundant in the water column (van der Stap *et al.*, 2016). This was similarly the case on the Palarmis P2 energy device where the presence of blue mussels was concentrated at 0.5 – 2 m depths (Nall *et al.*, 2017). In addition to depth and distance from the shore, research on mussel dominated communities found that mussel recruitment can also be facilitated by the presence of barnacles (Menge, 1976) and hydroids (Okamura, 1986), that create a rough settlement surface for larvae to attach. During the deployment of fish cages near the surface in the Gulf of Maine, USA hydroids were the primary settlers, followed by blue mussels that displaced other fouling invertebrates and became dominant for the following year

(Greene & Grizzle, 2007). Mussels also preferentially colonise pitted surfaces, highlighting the importance of substratum composition and texture also created by the conditioning layer and subsequent biofilm (Bayne, 1964; Seed, 1969; Okamura, 1986; Callow & Callow, 2002).

Alternatively, in this study sea vase tunicate colonised the AFIs before blue mussels (Figure 3.4). Primary settlement in most systems is dependent on food transport and nutrient supply, controlled largely by mass transfer rate and current velocities (Connell & Slatyer, 1977; Abelson & Denny, 1997; Perkol-Finkel *et al.*, 2006b). Sea vase tunicates are solitary filter feeding ascidians, typically found in the lower shore growing on hard substrata including both natural and artificial structures (Tolman, 2001; Paetzold *et al.*, 2012). The ascidian is predominantly associated with temperate climates, from Norway to the Mediterranean. In the last two decades however, it has been classified as an aquatic invasive nuisance successfully colonising a number of river systems in Canada and negatively impacting on the growth of blue mussels in the aquaculture industry (Tolman, 2001; Ramsay *et al.*, 2008). Unlike blue mussels that spawn during spring and summer, sea vase tunicates can reproduce throughout the year tolerating low water temperatures (8 – 12 °C) while spawning (Gulliksen, 1972; Seed, 1976; Svane & Havenhand, 1993). Therefore, during the six month period from the AFIs installation in September 2017 to the first underwater camera survey in April 2018, sea vase tunicate was able to colonise the AFI before blue mussels had spawned. By July in The Prince of Wales Dock, blue mussels had outcompeted sea vase tunicates. Their rapid growth and strong external shell can effectively displace neighbouring organisms on the substratum (Jackson, 1983; Okamura, 1986). Additionally, high stocking densities of blue mussels on mussel socks can reduce the growth rate of sea vase tunicates, as it creates less suitable conditions with reduced food availability and space (Ramsay *et al.*, 2008). The filter feeding activity of blue mussels also can deform sea vase tunicate larvae during spawning periods, limiting successful settlement and colonisation (Mileikovsky, 1974; Lehane & Davenport, 2004; Ramsay *et al.*, 2008).

Significantly larger blue mussel shells (length and width measurements) were recorded in the inner and outer sections of the AFI in comparison to the edge, with a high abundance of *B. crenatus* as an epibiont in the inner section (Figure 3.2; Table 3.1). This suggests that the edge of the AFI was either biofouled less rapidly than the inner and outer sections or exposed to differing abiotic and biotic stressors, impacting on mussel growth. Blue mussels at the edge of the AFI were vulnerable to predation by herring gulls (*Larus argentatus*) and lesser black-backed gulls (*Larus fuscus*). When blue mussels were exposed to common starfish (*Asterias rubens*) under controlled laboratory conditions, the shell growth was significantly lower than unexposed mussels (Reimer *et al.*, 1995). However, their weight remained similar to unexposed mussels demonstrating that the bivalve will actively change its morphology based on exposure to biotic stress (Reimer *et al.*, 1995). An additional study determined that mussels can thicken their shell and adductor muscles to increase the difficulty of being prised open by specific predators (Freeman, 2007). Fluctuating water temperatures and salinity as a result of rainfall and wave action created by high winds at the edge of the AFI, may also have caused unstable conditions within the swash zone. Blue mussels will temporarily close their shell valves in response to sudden changes in salinity, resulting in reduced feeding times and slower shell growth (Riisgård *et al.*, 2012). This has resulted in dwarfed blue mussel populations in the northern Baltic sea, that are exposed to low salinities of 6 – 8 PSU (Tedertgren & Kautsky, 1986; Vuorinen *et al.*, 2002; Riisgård *et al.*, 2012). Therefore, a combination of both biotic and abiotic stressors could have been responsible for the different sizes of blue mussels sampled across the AFI.

There was also a significant difference between the community assemblage of invertebrates in the inner and outer sections in comparison to the edge, controlled by the abundance of *B. crenatus*, *J. marmorata* and sea vase tunicate (Figure 3.2 and 3.3). Previously, the growth of *B. crenatus* and *Semibalanus balanoides* on blue mussels reduced the growth rate of blue mussels, due to interspecific competition for similar sized food particles (2 µm) (Barnes, 1959; Møhlenberg & Riisgård, 1978; Buschbaum & Saier, 2001). Although this relationship was not shown by the size of blue mussels in the inner sections, this could account for the lower abundance and potentially restricted reproductive output of blue mussels in the inner section of the AFI, in comparison to the outer section (Dittman & Robles, 1991; Buschbaum & Saier, 2001). Significant differences in community assemblage based on orientation of the fouled surface has also been recorded on pontoons (Pomerat & Reiner, 1942; Bassindale *et al.*, 1948). Abiotic parameters such as light intensity, wave exposure and the mass transfer of nutrients can vary based on orientation, either encouraging or discouraging the settlement of certain taxa (Glasby & Connell, 2001). For example, blue mussels have previously shown strong preference towards colonising horizontal substrata (Oganjan *et al.*, 2017). Additionally, at the edge of the AFI sufficient light intensities supported the growth of algal species and associated amphipods. The formation of an algal belt provided both shelter from predators and foraging opportunities for *J. marmorata* (Krapp-Schickel, 1993). *J. marmorata* was predominantly found inhabiting *R. riparium*; a filamentous green alga. Unlike the more widespread *Jassa falcata,* recorded at greater depths in the benthic community, *J. marmorata* tends to dominate in exposed, upper surface locations (Beermann & Franke, 2012; Beermann, 2014). Adult *J. marmorata* are larger in size and more rapidly reproduce than *J. falcata*, which could account for the absence of *J. falcata* on the AFI (Purz & Beermann, 2013; Beermann, 2014). Due to the quick colonisation of ascidians, bivalves and sponges, *J. marmorata* did not establish a tubular cover on the underside of the AFI, which can suppress the settlement of other sessile epibionts (Caspers, 1952; Beermann, 2014).

Japanese skeleton shrimp were the only non-indigenous species on the AFI in The Prince of Wales Dock and it was recorded in all three sections. The caprellid is native to sub boreal aquatic environments of north east Asia and was first recorded outside of its known distribution in 1970, quickly spreading throughout marine environments of the northern hemisphere over a 40 year period (Carlton, 1979; Ashton *et al.*, 2007; Cook *et al.*, 2007a). The species high reproduction and growth rate, broad tolerance of abiotic parameters and omnivorous feeding behaviour are some of the traits that have enabled Japanese skeleton shrimp to spread successfully across Europe (Boos *et al.*, 2011). The caprellid also has a broad diet feeding on macroalgae, diatoms and aquaculture feeding pellets in some instances (Cook *et al.*, 2007b). Where it has been identified as a non-indigenous species, it often features within heavily modified coastal water bodies (Ashton *et al.*, 2007; Cook *et al.*, 2007b; Kerckhof *et al.*, 2007). On the AFI in The Prince of Wales Dock no other competing caprellid species was identified suggesting Japanese skeleton shrimp was having a negligible impact on biofouling community development. In addition, it was identified during the collection of scrape samples on a pontoon present in The Prince of Wales Dock earlier in this study.

In addition, both Japanese skeleton shrimp and *J. marmorata* have previously been recorded colonising upper surface structures which generally have a lower predation risk than benthic communities (Greene & Grizzle, 2007). The presence of a high number of amphipods and bivalves fouling underneath the AFI confirmed that the islands could support fish populations as a feeding site. In addition, the presence of filter feeding organisms may have enhanced nutrient cycling processes, improving water quality and enriching localised invertebrate communities present on bottom sediment (Langhamer, 2010; Coates *et al.*, 2014; Nall *et al.*, 2017).

Blue mussels also heavily fouled the stainless steel chains used to install the AFI, adding a substantial amount of weight to the structure. Offshore trials of blue mussel seed production on polypropylene and steel hawser longlines confirmed the ability of blue mussels to grow successfully on steel (Buck, 2007). For future installations in heavily modified coastal water bodies, it is recommended that a management plan is implemented to monitor and clean the installation chain when required. This will prevent the structure from losing buoyancy and dipping in the water, that can cause damage to the coir matting and plants that may have established on the upper surface, while retaining the secondary reef feature on the underside of the AFI.

3.5.2 Swansea Marina

The serpulid Australian tubeworm predominantly fouled the AFI in Swansea Marina, creating a biogenic reef its underside. It is a calcareous reef-building polychaete characteristically found in brackish coastal marinas, lagoons and estuaries with high nutrient contents, forming globular aggregations on soft sediment or encrusting pilings, buoys or pontoons (Read & Gordon, 1991; Iribarne & Schwindt, 1998; Rolston *et al.*, 2009; Charles *et al.*, 2018). The native range of Australian tubeworms remains unclear, however it is suggested that the polychaete originated from Australia (Allen, 1953; Iribarne & Schwindt, 1998) and spread throughout subtropical and tropical water bodies of the southern hemisphere (Dixon, 1981). The Australian tubeworm extended its range to temperate waters of the northern hemisphere in 1921, where it was first discovered in Normandy, France and was recorded in London Docks by 1922 (Monro, 1924; Dixon, 1981; Schwindt *et al.*, 2001). The complex intertwined tube structures produced by Australian tubeworms can attenuate currents, altering flow regimes and sedimentation rates, influencing benthic community assemblages in localised habitats (Iribarne & Schwindt, 1998; Heiman & Micheli, 2010). In Mar Chiquita, Argentina for example the presence of Australian tubeworms on large embankments resulted in the accumulation of sediments in the reef that would otherwise be transported downstream (Iribarne & Schwindt, 1998). The consequent change in depth of the lagoon resulted in complications for recreational activities and potential long term impacts on invertebrate diversity (Iribarne & Schwindt, 1998). At an alternative reef in Elkhorn Slough, California three crustaceans were associated with Australian tubeworms in high abundance including *Monocorophium insidiosum*, *Melita nitida* and *Hemigrapsus oregonensis* (Heiman & Micheli, 2010). The complex structure of the biogenic reef provides shelter from predators and aids the retention of propagules, enhancing reproductive success of amphipods that brood their young (Heiman & Micheli, 2010).

In Swansea Marina there was a significant difference in the community assemblage of invertebrates in the inner and outer section of the AFI in comparison to the edge. This was largely controlled by the abundance of bay barnacles and *M. palmata*. The exact native origin of bay barnacles is also unclear, although it has been commonly associated with northern America (Leppäkoski & Olenin, 2000; de Rivera *et al.*, 2011; Wrange *et al.*, 2016). Like Australian tubeworms, it is common in brackish water bodies such as estuaries and has a broad salinity, temperature and pH tolerance (Pansch *et al.*, 2013; Wrange *et al.*, 2014, 2016). It was first recorded in the Thames River, United Kingdom in 1854 and has since been recorded in shallow, coastal environments worldwide (Kawahara, 1963; de Rivera *et al.*, 2011). On the AFI, the abundance of bay barnacles was significantly higher in the inner and outer sections in comparison to the edge. In Pärmu Bay located in the north-eastern Baltic sea wave exposure controlled the distribution of bay barnacles and the species also preferentially colonised vertical surfaces; contradicting the results of this study (Oganjan *et al.*, 2017). However, due to the integrated connection grid present on the outskirts of the AFI installed in Swansea Marina, it was difficult to achieve the same sample area as the inner and outer sections. This may explain the difference in barnacle abundance found across the three sections of the AFI.

M. palmata was the most abundant amphipod on the AFI and a higher number of individuals were sampled on the outer and edge sections. The species has previously been associated with Australian tubeworms in the Mar Chiquita lagoon in Argentina (Obenat *et al.*, 2006), Nazioni Lake in northern Italy (Mistri & Rossi, 1999) and in marinas around Normandy, France (Charles *et al.*, 2018). The higher abundance of *M. palmata* in the outer section may be due to the higher abundance of Australian tubeworms, as the complex structure of calcareous tubes allows the amphipod to retain propagules, enhancing reproductive success (Heiman & Micheli, 2010). In heavily modified coastal water bodies Australian tubeworm reefs enhance nutrient cycling, as they filter feed on suspended organic particulate and excrete inorganic nutrients into the system, improve water quality and provide feeding opportunities for fish populations (Keene, 1980; Davies *et al.*, 1989).

3.5.3 Comparison

Heavily modified coastal water bodies tend to be brackish and polluted environments regularly disturbed by anthropogenic activities and therefore, colonised by species with a broad ecological amplitude and high resistance to fluctuating abiotic parameters (de los Ríos *et al.*, 2016; Charles *et al.*, 2018). The distinct differences in species recorded fouling on the two AFIs could be due to the physiological limitations of individual species to salinity, with higher salinities in The Prince of Wales Dock and lower salinities in Swansea Marina (see Appendix 3). However, both blue mussels and Australian tubeworms have a broad salinity tolerance ranging from 10 – 30 (Hiscock & JNCC, 1996; Rolston *et al.*, 2009; Riisgård *et al.*, 2012). Therefore, anthropogenic activities including the spread of non-indigenous Australian tubeworm from recreational boats using Swansea Marina and Japanese skeleton shrimp in The Prince of Wales Dock, plus the blue mussel aquaculture farm present in the dock, have largely influenced the formation of biogenic reefs on the AFIs. The presence and management of non-indigenous species in heavily modified coastal water bodies is a key area of concern as the degraded habitats and constant boat traffic facilitates non-indigenous species establishment (Molnar *et al.*, 2008; Seebens *et al.*, 2013; Charles *et al.*, 2018). The results of this study highlight the potential for AFIs to facilitate the spread of non-indigenous species, acting as a propagule between sites. For future installations this must be managed via production of a

Biosecurity Risk Assessment and if required, an invasive non-native species management plan prior to an AFI deployment.

3.6 Conclusion

The community assemblages that colonised the two AFIs were significantly different on the inner and outer horizontal surfaces in comparison to the vertical edge, largely controlled by the absence and presence of barnacles, amphipods and algae. Therefore, the hypothesis of this chapter was accepted. The AFIs were colonised by mussels and tubeworms, both ecosystem engineers that facilitate the colonisation of a wide range of biofouling invertebrates. The high abundance of filter feeders fouling the AFIs could improve water quality and assist in localised cycling of nutrients in heavily modified coastal water bodies, providing ecosystem services. To ensure that the buoyancy of an AFI is not compromised by biofouling on the installation chain and to prevent facilitating the spread of non-indigenous species, a management plan is recommended that will ensure the production of a Biosecurity Risk Assessment prior to deployment and regular cleaning of the installation chain.

Chapter 4: Can artificial floating islands support 'essential fish habitats' associated with heavily modified coastal water bodies?

Abstract

Essential fish habitats (EFH) are defined as sites that are necessary for fish to spawn, breed, feed and grow to maturity, including the water and associated substrata. Marinas and docks, although largely uncomplex and enclosed ecosystems can create shallow water environments suitable for fish nursery development. The installation of ecological engineering methods such as artificial floating islands (AFIs) could be used to support EFH present in heavily modified coastal water bodies. This study tested the hypothesis that fish will change their vertical distribution in the tank with a 2 m² AFI present (deployment phase), in comparison to without an AFI present (reference phase). 21 artificial root bundles were attached to the underside of the matrix unit. The AFI was deployed in a 180 m³ tank at Bristol Aquarium and 13 native fish species were monitored remotely using underwater video cameras (Go Pro Hero 5 Session) to determine if their vertical distribution and shoaling behaviour changed; 5.5 hours of the reference and deployment phase footage was compared (n = 84). The field experiment tested the hypothesis that vegetated AFIs installed in heavily modified coastal water bodies will attract a higher number of fish than pontoons (control) and unshaded sites (reference) which lack structures at the surface. One 8 m² AFI was installed in Swansea Marina and two AFIs in The Prince of Wales Dock (TPoWD): 8 m² and 13.2 m². The AFIs, pontoons and unshaded sites were monitored using remote underwater video from March 2018 – May 2019 to assess the differences in fish relative abundance (MaxN), species richness and behaviour (n = 13, TPoWD; n = 14, Swansea Marina). At Bristol Aquarium, horse mackerel (*Trachurus trachurus*), gilthead sea bream *(Sparus aurata)* and Pleuronectiforme species (spp.) were all recorded significantly more in the middle and upper sections of the tank and European pollock (*Pollachius pollachius*), black sea bream (*Spondyliosoma cantharus)* and smooth-hound (*Mustelus*) spp. were recorded significantly more in the middle sections, during the deployment phase in comparison to the reference phase; therefore, the first hypothesis was accepted. In the field experiment, European sea bass (*Dicentrarchus labrax*), European eel (*Anguilla anguilla*) and mullet spp. (Mugilidae) were recorded at TPoWD and Swansea Marina. At TPoWD the fish MaxN was significantly higher at the small AFI and pontoon, in comparison to the unshaded site. At Swansea Marina the fish MaxN was significantly higher

at the AFI and unshaded site in comparison the pontoon. However, at TPoWD and Swansea Marina there was a significantly higher MaxN of juvenile European sea bass under the AFIs in comparison to the pontoon and unshaded site. Therefore, the second hypothesis was accepted in relation to European sea bass and rejected for mullet spp. and European eel. Juvenile European sea bass used the AFIs for shelter and feeding demonstrating a species-specific relationship with the structure. Water temperature, salinity and life stage were highlighted as key factors influencing the interaction of fish species with the AFIs. This demonstrated AFIs potential to support EFH by the provision of ecosystem services in heavily modified coastal water bodies.

4.1 Introduction

From the 1970s onwards, it became apparent that many commercial fisheries were operating unsustainably and at risk of collapse, resulting in national and international efforts to reverse the decline in global fish stocks (Ludwig *et al.*, 1993; Hilborn *et al.*, 2019). This included Atlantic herring (*Clupea harengus*) which largely collapsed in the northeast Atlantic, evident by the spawning-stock biomass declining from >2 x 10^6 tonnes in the 1960s to <50 x 10^3 tonnes by the mid-1970s (ICES, 1998; Dickey-Collas *et al.*, 2010). Legislation including the Sustainable Fisheries Act, 1996 in the United States (Levin & Stunz, 2005), the Common Fisheries Policy in Europe (Dickey-Collas *et al.*, 2010; Hilborn *et al.*, 2019) and the Fisheries Act 1981 in the United Kingdom all came into force to support the sustainable management of commercial fisheries. In addition to overfishing, heavy modification of shorelines (Munsch *et al.*, 2017), introduction of invasive non-indigenous species (Britton *et al.*, 2010) and temperature fluctuations caused by climate change (Baudron *et al.*, 2020) all contribute to changes in fish population dynamics and distribution. Therefore, as well as managing stock-recruitment relationships it is important to gain an understanding of possible interactions between fish and habitat that threaten productivity (Levin & Stunz, 2005).

Essential fish habitats (EFH) are referred to as areas or volumes of water and bottom substratum that are necessary for fish to spawn, breed, feed or grow to maturity (Levin & Stunz, 2005; Valavanis *et al.*, 2008; Institute of Estuarine and Coastal Studies, 2016). Shallow ecosystems associated with coastal and transitional water bodies often form nursery and feeding grounds for fish which are of cultural, ecological and economic significance (Munsch *et al.*, 2017). Therefore, coastal and transitional water bodies fall within the definition of EFH, which may also apply to heavily modified coastal water bodies if they can support fish populations during critical life stages. Factors affecting the recruitment, survival and distribution of fish in different habitats include water temperature (Gibson, 1994), salinity

(Marshall & Elliott, 1998), dissolved oxygen, hydrodynamics (Rijnsdorp *et al.*, 1985), food availability (Leggett & Deblois, 1994), predation, competition, vegetation and sediment structure (Gibson, 1994; Elliott & Hemingway, 2002). Fish have complex life cycles that require different resources at each life stage and occupy multiple ecological niches (Fukuhara, 1986; MacCall & Rothschild, 1987; Elliott & Hemingway, 2002). The subsequent change in abiotic and biotic factors created by the development of heavily modified coastal water bodies determines their ecological functioning role (Levin & Stunz, 2005) for a number of associated guilds, including diadromous species, estuarine residents, marine adventitious species, marine juvenile migrants and marine seasonal migrants (Elliott & Dewailly, 1995; Elliott & Hemingway, 2002).

Marinas and docks used for trade, tourism and recreational activity are one of the most common and largescale coastal developments (Beck & Airoldi, 2007; Dugan *et al.*, 2012; Sekovski *et al.*, 2012). For example, the growth of marinas and yacht harbours in France is estimated to increase by 1.5 – 2.6 yr^{-1} (European Environment Agency, 2006; Wolanski, 2006; Beck & Airoldi, 2007). In order to form a marina, initial dredging and filing operations are required which cause substantial habitat destruction (Wilson *et al.*, 2015; Selfati *et al.*, 2018). They are often vast infrastructure developments that include the construction of jetties, seawalls and floating pontoons (Dugan *et al.*, 2012). The loss of macroalgae naturally occurring in the littoral zone consequently reduces the spatial complexity of the system and the abundance of fish it may recruit (Hölker *et al.*, 2002; Lewin *et al.*, 2004; Okun & Mehner, 2005). The establishment of residential communities and associated boating activity can also result in high levels of chemical waste pollution (Vadeboncoeur *et al.*, 2001; Bech, 2002; Neira *et al.*, 2011), litter, noise and light disturbance, water level manipulation (Coops *et al.*, 2003; Kahl *et al.*, 2008) and the introduction of invasive non-indigenous species (Sekovski *et al.*, 2012; Bouchoucha *et al.*, 2016).

For example, industrial development around the Manchester Ship Canal and Salford docks resulted in heavy pollution from anthropogenic waste and a high sediment oxygen demand from the contaminated sediment in the system (Williams *et al.*, 2010). This combined with the high retention time of water in the dock depleted fish populations and caused stratification and bottom water anoxia, until restoration began in the 1980s (Williams *et al.*, 2010). In contrast, high angling activity in the Rideau and Ottawa river resulted in greater soil compaction, litter and reduced aquatic macrophyte density and diversity (O'Toole *et al.*, 2009). Enhanced lighting can also create optimal conditions for piscivorous predators, as the light attracts high abundances of small shoaling fish and could result in an unnatural top–down trophic system, mediating fish populations (Becker *et al.*, 2013). The combined impacts on the physico-chemical environment and trophic food webs could result in disconnection of the system to other localised habitats and the potential loss of native species (LaPoint *et al.*, 2015; Bishop *et al.*, 2017; Selfati *et al.*, 2018). For example, the installation of dykes in the Seine estuary caused common dab (*Limanda limanda*) populations to disappear (Elliott & Hemingway, 2002). The disconnection and fragmentation of coastal and transitional water bodies could also impact on their nursery function, particularly for diadromous species, marine juvenile migrants and marine seasonal migrants (Elliott & Dewailly, 1995; Elliott & Hemingway, 2002). Therefore, the ecological functioning role of coastal environments is continuing to deviate from previous baselines associated with habitat quality and productivity and it is becoming increasing more difficult to determine (Rose, 2000; Peterson & Lowe, 2009).

In heavily modified coastal water bodies, artificial structures such as 'biohuts' have been installed to provide shelter for rocky shore fish species (Bouchoucha *et al.*, 2016; Selfati *et al.*, 2018). 'Biohuts' are a type of fish aggregation device (FAD) (Gooding *et al.*, 1967) which can vary from objects moored on the seabed, suspended in the water column or floating at the surface and are largely used to attract target species for artisanal, sport or commercial fishing practices (Gooding *et al.*, 1967; Robert *et al.*, 2013; Montes *et al.*, 2019). The traditional technique has been used for decades to improve pelagic fishery yield and information has been gained on how the location (Friedlander *et al.*, 1994), size (Sinopoli *et al.*, 2011) and design (Workman *et al.*, 1985; Higashi, 1994) of FADs influences the associated species in relation to abundance, life cycle stage and spatial and temporal patterns of behaviour (Castro *et al.*, 2002a; Capello *et al.*, 2012). In the Gulf of Lions, France juvenile common two-banded sea bream (*Diplodus vulgaris*) associated with the 'biohuts' in high abundances (Bouchoucha *et al.*, 2016). In Marchica Lagoon, Morocco the endangered dusky grouper (*Epinephelus marginatus*) and comb grouper (*Mycteroperca acutirostris*) also used the 'biohuts' and fish abundance was higher around the 'biohut' in comparison to natural habitat in the outer lagoon (Selfati *et al.*, 2018). The provision of shelter that artificial microhabitats provide during the early stages of development, could add value to EFH, in heavily modified coastal water bodies. Artificial floating islands (AFIs) are an additional eco-engineering method that could be used to support diadromous species, marine juvenile migrants and marine seasonal migrants present in EFH.

4.2 Aims and Objectives

The aim of this study was to determine whether AFIs are a viable method of habitat creation in heavily modified coastal water bodies associated with EFH. The laboratory experiment tested the hypothesis that fish will change their vertical distribution in the tank with an AFI present, in comparison to without an AFI present. The field experiment tested the hypothesis that vegetated AFIs installed in heavily modified coastal water bodies will attract a higher number of fish than pontoons and unshaded sites which lack structures at the surface. In theory, the combination of fouling invertebrates and protruding roots from established plant growth will create a topographically complex structure on the underside of the AFI, that could provide shelter from predators and feeding opportunities in EFH for juvenile and adult fish populations. This was achieved with the following four objectives:

1) To assess if an installed AFI affects the vertical distribution and shoaling behaviour of 13 native fish species under controlled conditions in a tank experiment.
2) To assess the effect of AFIs and pontoons on the relative abundance (MaxN), species richness and behaviour of fish species in comparison to unshaded sites in heavily modified coastal water bodies.
3) To determine if AFIs could be used to support EFH often present in heavily modified coastal water bodies.
4) To investigate potential abiotic and biotic factors influencing fish species presence and behaviour, including life cycle stage, food availability, shelter and water chemistry.

4.3 Materials and Methods

4.3.1 Aquarium Experiment

In collaboration with Bristol Aquarium, a 1 m x 2 m Biohaven® matrix unit (commercially sold by Frog Environmental; Figure 3.1) was deployed into a 5 m x 9 m x 4 m tank on 18th March 2019, in order to assess the behaviour of 13 native fish species: 26 European sea bass (*Dicentrarchus labrax*), 18 horse mackerel *(Trachurus trachurus*), 12 European pollock (*Pollachius pollachius*), nine ballan wrasse (*Labrus bergylta*), eight lesser spotted catshark (*Scyliorhinus canicula*), five turbot (*Scophthalmus maximus*), five black sea bream *(Spondyliosoma cantharus),* four gilthead sea bream (*Sparus aurata*), four greater spotted catshark (*Scyliorhinus stellaris*), two common smooth-hound shark (*Mustelus mustelus*), two bib (*Trisopterus luscus*), a starry smooth-hound shark (*Mustelus asterias*) and a European plaice (*Pleuronectes platessa).* The AFI consisted of a non-woven plastic matrix, integrated connection grid and 21 planting holes, 9 cm in diameter. For the purposes of this experiment 21 bundles of 50 – 60 cm artificial roots were attached underneath using cable ties to create a complex 3-D structure. In the tank, the roots were suspended in the water column, mimicking a vegetated AFI in the field offering shelter and environmental enrichment to the species present. For aesthetic benefits in Bristol Aquarium, 2 m² of Astroturf and nine artificial flowers were attached to the upper horizontal surface of the AFI. The AFI was installed using the integrated connection grid on the outside of the island and clean rope to secure the AFI in position (Figure 4.1).

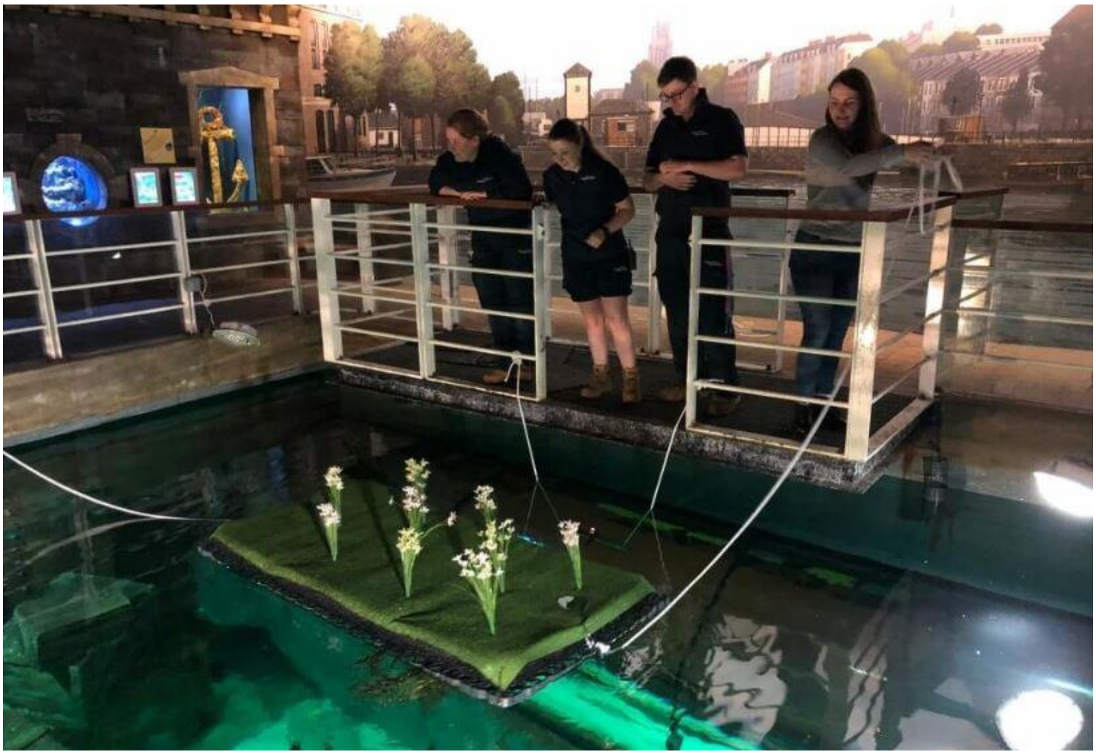


Figure 4.1: Deployment of the 2 m² Biohaven® matrix unit in the native fish tank at Bristol Aquarium on 18th March 2019. The artificial floating island consisted of a non-woven plastic matrix, integrated connection grid, 21 bundles of artificial roots, AstroTurf and nine artificial flowers.

The monitoring equipment consisted of a 0.5 m scaffolding pole, 8 mm polyester rope, Go Pro mount, waterproof case and four Go Pro Hero 5 Sessions. Prior to deployment, all of the equipment was submerged in a mix of safe 4 solution and water at a concentration of 1:100 for ten minutes. Once disinfected, the equipment was rinsed with water for one minute to ensure there were no contaminants entering the tank. After a trial period estimating the optimum angle for monitoring the entirety of the tank, one of the four Go Pros was suspended 50 cm into the water using the feeding platform from 13th – 15th March 2019. During the three days, a total of 15 hours of footage was collected in order to assess fish behaviour without the AFI in the tank; this will be referred to as the reference phase. Each Go Pro had a battery life of 2 hours and therefore, was replaced with another Go Pro every two hours; two full days with 6 hours of footage and one half day with 3 hours. For five days, the Go Pros were deployed with the AFI installed and collected 32 hours of video footage; 6.4 hours of footage per day. This period will be referred to as the deployment phase.

The salinity, pH, dissolved oxygen, ammonia, nitrate and nitrite concentrations were recorded once per week during the experiment. Water temperature was recorded daily (Table 4.1).

Table 4.1: Environmental parameters monitored once per week during the two week experiment from 11th – 22nd March 2019. These included water temperature, salinity, pH, dissolved oxygen, ammonia, nitrate and nitrite concentrations (data provided by Bristol Aquarium staff).

Week	Temperature (°C)	Salinity	pH	Dissolved oxygen (%)	Ammonia (mg/l)	Nitrate (mg/l)	Nitrite (mg/l)
1	14.9	30	7.7	97.7	0	25	0.1
2	14.4	30	8	91.2	0	25	0

4.3.1.1 Video Analysis

The Go Pro positioned during deployment allowed assessment of the entire width and depth of the tank, from the feeding platform to the back wall (8 m). The recorded video footage was analysed in nine, 8.5 cm x 4.7 cm subsections (Figure 4.2). Each section was numbered (1 – 9) and for every four minutes of footage the species richness, number of fish and behaviour was determined for each section. This was later grouped into upper (1 – 3), middle (4 – 6) and lower (7 – 9) for analysis. Behaviour was divided according to the swimming direction (up, down, straight or stationary) and number of individuals swimming together (individual, 1 – 2; small shoal, 3 – 4; medium shoal, 5 – 6 and; large shoal, 7 plus). This method of analysis was adopted for all the video footage collected during the experiment. In this study 5.5 hours of footage during the reference and deployment phase was analysed in relation to fish vertical distribution and shoaling behaviour (n = 84), in order to test the hypothesis that fish use the middle and upper sections of the tank more frequently with the AFI present in comparison to without. Due to the short time interval of data collection, autocorrelation may have impacted on the results of this study, however the high ‘n’ value was viewed as sufficient to minimise the impact.

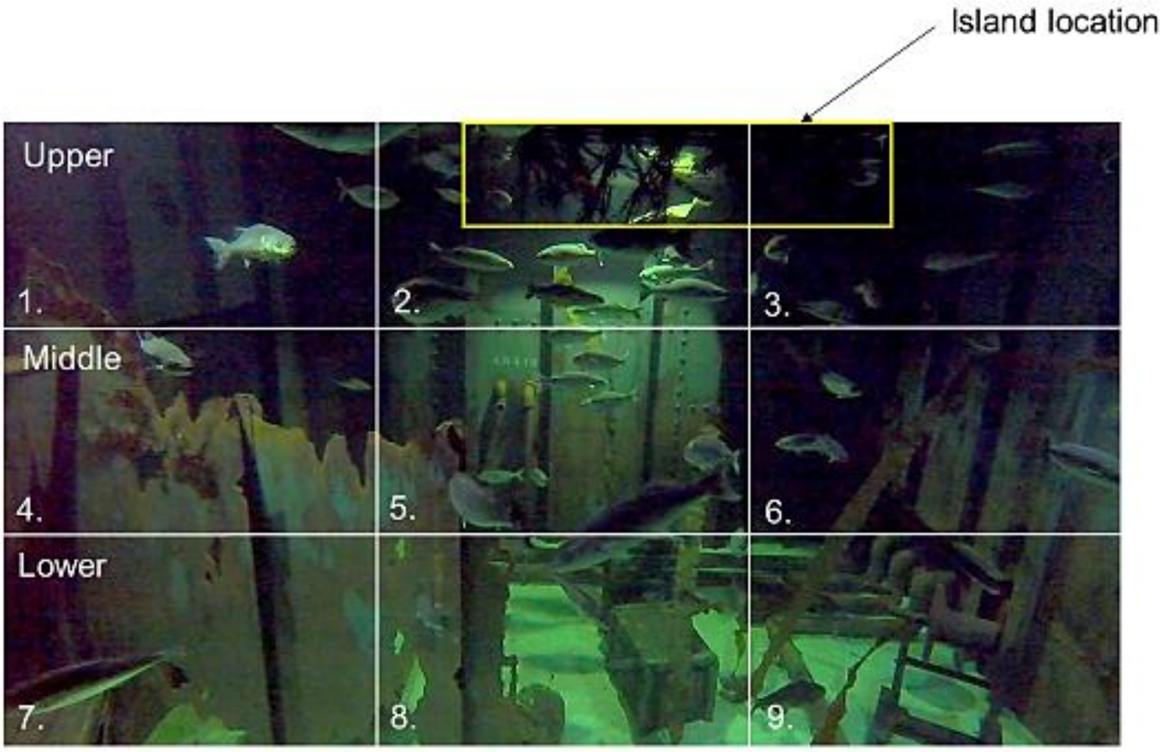


Figure 4.2: Analysis of the underwater video camera footage of the deployed 2 m² artificial floating island in Bristol Aquarium. It was firstly subdivided into nine 8.5 cm x 4.7 cm subsections and later grouped into the upper (1 – 3), middle (4 – 6) and lower (7 – 9) section.

When the AFI was deployed, additional analysis was undertaken recording fish in the interaction zone. The interaction zone was determined as a 3 m² area; 0.5 m either side of the AFI and 1 m below. Comparisons were also made between 1.2 hours of morning and afternoon video footage to assess if exposure time to the AFI influenced fish behaviour (n = 20).

4.3.2 Field Experiment

Two, 8 m² Biohavens® were installed on 28th and 29th September 2017; one located in Swansea Marina (51°36’56.3”N, 3° 56’26.0”W) and one in The Prince of Wales Dock (51°37’10.6”N, 3°55’30.0”W). A 13.2 m² Biohaven® was also installed on 17th May 2018 in The Prince of Wales Dock (51°37’09.8”N, 3°55’29.8”W; Figure 4.3) after complications attempting to install the AFI in a tidal location (see Appendix 1).

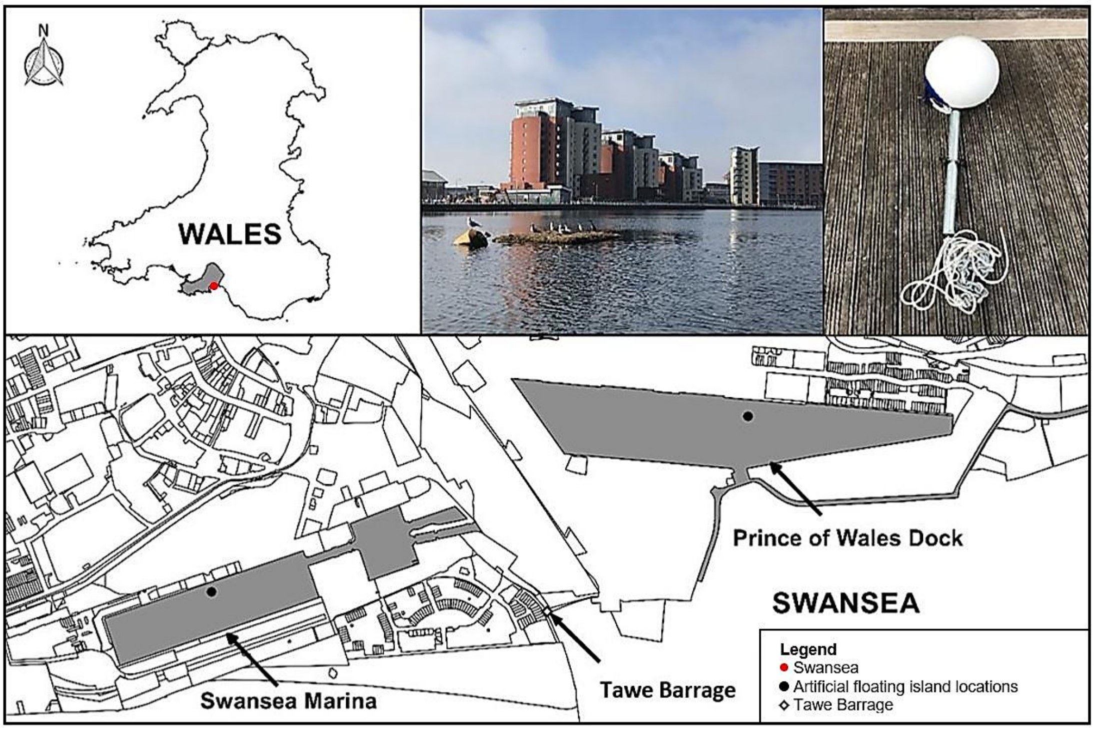


Figure 4.3: *Top left* – Wales and the county of Swansea (GIS Map, 2013). *Top middle* – Large artificial floating island (AFI) in The Prince of Wales Dock with a great cormorant (*Phalacrocorax carbo*) and five juvenile *Larus* species resting on it. *Top right* – Go Pro mount created to monitor fish at the unshaded sites. *Bottom* - AFI locations in Swansea Marina and The Prince of Wales Dock and the location of the Tawe Barrage, Swansea (HM Land Registry, 2014). Map produced in QGIS 3.4 Madeira.

Swansea Marina was previously the location of the south dock; a heavily used area for coal exports and copper imports from other coastal regions, such as Cornwall and Anglesey. After the docks closure in 1969, Swansea Council redeveloped the area into Swansea Marina in 1982. Boat birthing facilities were provided and flat accommodation converting the area from an industrial site into a leisure community. In 1992, the Tawe Barrage was also built in order to control boating traffic and water height in Swansea Marina (History Points, 2018). As a result of the tide overtopping the primary and secondary weirs located next to the Tawe barrage, water salinity can vary between 0.02 – 24.38. Water depth also fluctuates from 0.5 – 5 m (Swansea Council, 2016). In order to compare the relative abundance (MaxN; the maximum number of fish per frame (Unsworth *et al.*, 2014; Grimmel *et al.*, 2020)), species richness and behaviour of fish associated with the AFI in Swansea Marina, a section of pontoon 60 m away from the AFI was used as a control site. The pontoons consist of wooden platforms attached to polystyrene and concrete blocks for buoyancy. Similarly to AFIs, pontoons provide shelter and feeding opportunities for fish as a floating structure readily fouled by aquatic invertebrates. However, they lack vegetation cover and root growth that adds complexity to the AFI structure. In addition, pontoons are often used to access boats in heavily modified coastal water bodies and therefore, subject to greater noise disturbance. An unshaded location 60 m away from the AFI was used as a reference site, which lacked any floating structures at the surface to ascertain if fish activity varied in comparison to the AFI locations and control sites associated with the pontoons (Figure 4.4). The data collected may further evidence the use of floating structures as FADs in heavily modified coastal water bodies.

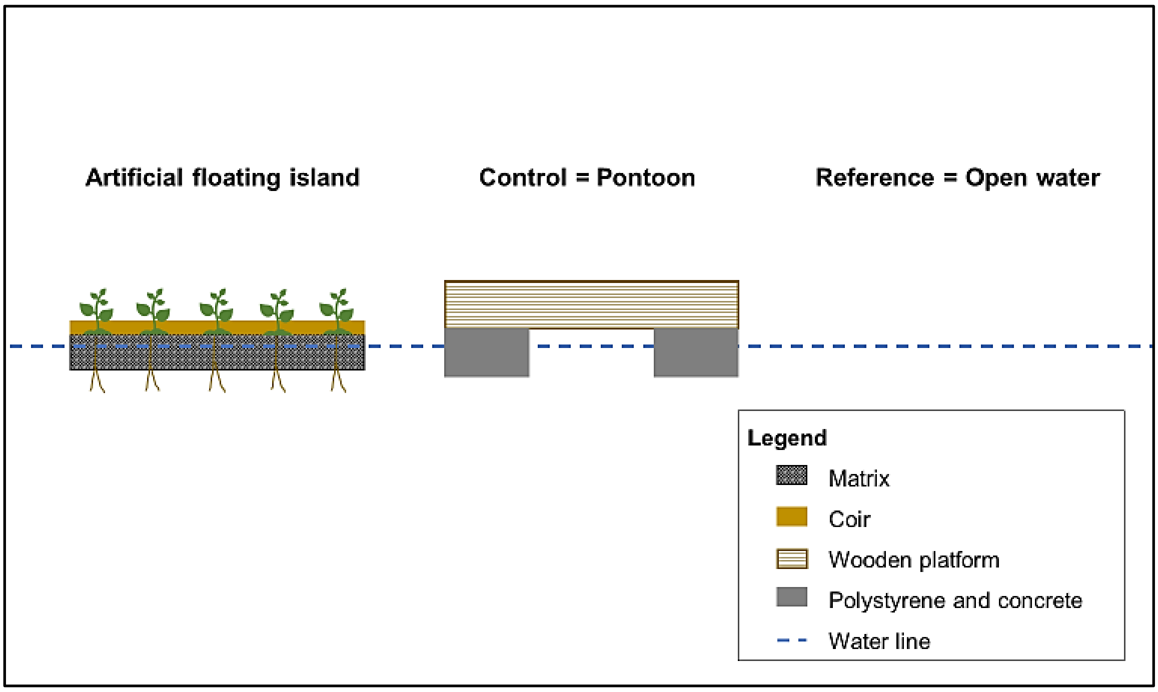


Figure 4.4: Schematic diagram showing the three comparative sites at The Prince of Wales Dock and Swansea Marina. These include the artificial floating islands, section of a pontoon as a control and a designated unshaded location.

The Prince of Wales Dock opened in 1881 and forms part of the Port of Swansea; one of five docks in south Wales. It was one of the first modern docks to open, exporting coal and copper across Europe. The water height in the dock is approximately 7.6 m and it covers around 0.11 km^2. It is now owned by Associated British Ports (ABP), who have minimised activity in The Prince of Wales Dock allowing Swansea Water Sports to run recreational boating activities. Due to the lack of boating activity, there is minimal disturbance influencing the spatial distribution of fish. In The Prince of Wale Dock, the section of pontoon used as a control was 200 m away from the two AFIs and the unshaded location used as a reference site was 100 m between both the AFIs and the pontoon. The monitoring locations were selected to ensure that there was suitable distance between the AFIs, control and reference sites to prevent potential in-combination effects and to test the hypothesis that vegetated AFIs attract a higher MaxN of fish in comparison to the control and reference sites.

4.3.2.1 Monitoring

Remote underwater video was used to determine the MaxN. species richness and behaviour of fish in association with the three AFIs, pontoons (control) and unshaded areas (reference) in Swansea Marina and The Prince of Wales Dock. Three Go Pro Hero Sessions were deployed in Swansea Marina, from 7th March 2018 – 2nd May 2019 (n = 14) and 25th April 2018 – 2nd May 2019 in The Prince of Wales Dock (n = 13). The larger AFI in The Prince of Wales Dock was monitored with an additional Go Pro Hero Session from 25th July 2018 – 2nd May 2019 (n = 10). The cameras were deployed 50 cm underwater using a 0.6 m galvanised scaffolding pole and 8 mm polyester rope which attached the system to the AFIs and pontoon. For the unshaded sites, the scaffolding pole was anchored in position with 12 mm short link chain and marked with an A1 buoy fender (29 cm x 37 cm). Once the Go Pros were deployed, each location was monitored for 1-2 hours per survey, alternating between Swansea Marina and The Prince of Wales Dock in either the morning or afternoon. A replicate was defined a monitoring day at each site.

Meteorological data and water chemistry parameters were monitored and recorded, prior to the deployment of the Go Pros underneath the AFIs, pontoons and in the unshaded locations. This included air temperature, humidity, wind speed, illumination, water temperature, salinity, pH and redox potential. The salinity at The Prince of Wales Dock ranged from 28 – 34 in comparison to Swansea Marina which ranged from 8 – 17 (see Appendix 3 for more information on seasonal variations in meteorological data and water chemistry parameters).

4.3.3 Statistical Analysis

Prior to statistical analysis, all of the data collected was tested for normality using the Shapiro Wilk normality test. For categorical data collected during the laboratory experiment, the Chi Squared test for goodness of fit and binomial post hoc testing was used to determine if there was a significant difference in the vertical distribution (lower, middle and upper sections) and shoaling behaviour (individual, small, medium and large shoal) of fish during the reference and deployment phase. A subset of laboratory data collected in the morning and afternoon of the AFIs deployment was reanalysed using Kruskal Wallis and Nemenyi’s multiple pairwise comparison testing, to determine if there was a significant difference in the abundance and vertical distribution of each species based on exposure time to the AFI. This analysis was used to test the hypothesis that fish will change their vertical distribution in the tank in the presence of an AFI.

Kruskal Wallis and Nemenyi's multiple pairwise comparison testing was also used to determine if there was a significant difference in fish MaxN and species richness at the AFIs, pontoon and unshaded site in The Prince of Wales Dock and Swansea Marina. Additionally, Chi squared test of independence and post hoc testing (Beasley & Schumacher, 1995; García-pérez & Núñez-antón, 2003) was used to determine if there was a significant difference in the MaxN of each species at the AFIs, pontoons and unshaded sites and behaviour of fish at each monitoring site. Data analysis was conducted in R 3.5.1 Statistics Software and SPSS. This analysis was used to test the hypothesis that vegetated AFIs installed in heavily modified coastal environments will attract a higher number of fish than pontoons and unshaded sites which lack structures at the surface.

4.4 Results

4.4.1 Aquarium Experiment

4.4.1.1 Ray-Finned Fish

Significantly more horse mackerel were recorded in the upper and middle sections of the tank during the AFIs deployment in comparison to the reference phase (χ^2= 100.25, df = 2, p <0.001; post hoc, upper section, p <0.001; middle section, p = 0.011; n = 84). The total number of horse mackerel recorded during the deployment phase was 48.96 % higher in the middle section and 107.97 % in the upper section, in comparison to the reference phase (see Appendix 2). In contrast, the number of fish recorded in the lower section of the tank was significantly lower in the deployment phase in comparison to the reference phase (χ^2 post hoc, lower section, p <0.001; n = 84). An average of 3.24 horse mackerel (17.99 % of 18 individuals; n = 84) were present in the 3 m^2 interaction zone during the AFIs deployment. Additionally, there were significantly less horse mackerel recorded commuting as part of a medium sized shoal in the deployment phase than the reference phase (χ^2 = 16.615, df = 3, p <0.001; post hoc, medium shoal, p = 0.006; n = 84).

There was a significantly more gilthead sea bream in the middle and upper sections of the tank in the deployment phase, in comparison to the reference phase (χ^2 = 56.318, df = 2, p <0.001; post hoc, middle section, p <0.001; upper section, p <0.001; n = 84). The number of gilthead sea bream in the lower section of the tank was significantly higher during the reference phase in comparison to the deployment phase, in both the morning and afternoon (Kruskal Wallis, morning = 9.75, df = 2, p = 0.021; afternoon = 17.18, df = 2, p <0.001; n = 20). In the afternoon, the number of gilthead sea bream was also significantly higher in the middle section of the tank when the AFI was deployed in comparison to the reference phase (Kruskal Wallis, df = 2, p = 0.004; Figure 4.5).

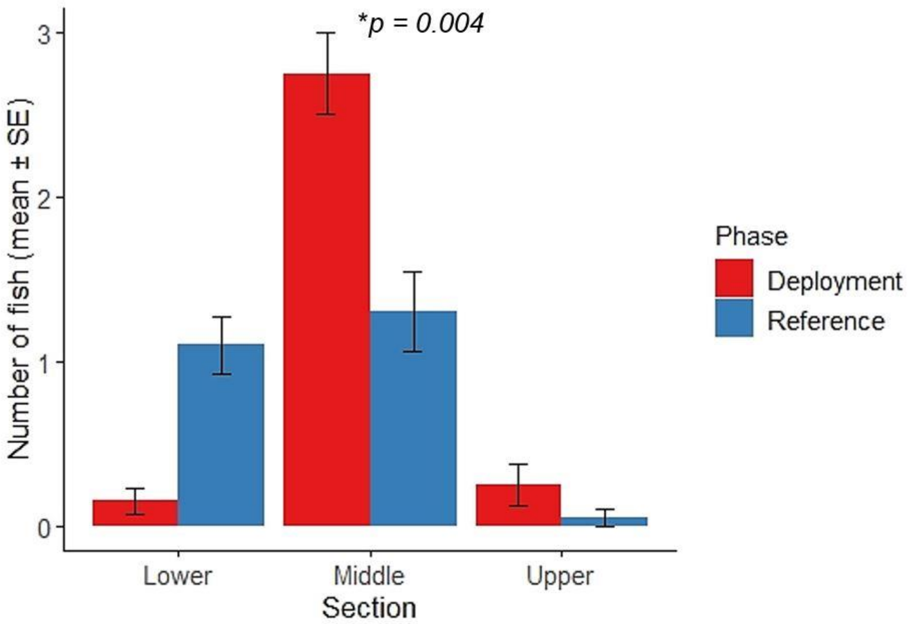


Figure 4.5: The number of gilthead sea bream (*Sparus aurata*) (mean ± standard error) recorded in the lower, middle and upper sections of the tank, during the afternoon (15.00 – 16.20) of the deployment phase and the reference phase at Bristol Aquarium (n = 20). *p value calculated using Kruskal Wallis and showed that significantly more gilthead sea bream were present in the middle section during the deployment phase in comparison to the reference phase.

In relation to shoaling behaviour, more gilthead sea bream were commuting as small shoals in the deployment phase in comparison to the reference phase (χ^2= 18.199, df = 4, p = 0.00; post hoc, small shoal, p = 0.02; n = 84).

There was significantly more European pollock recorded in the middle section of the tank during the deployment phase in comparison to the reference phase (χ^2 = 22.585, df = 2, p <0.001; post hoc, middle section, p <0.001; n = 84). Similarly, black sea bream was recorded significantly more in the middle section of the tank during the deployment phase (χ^2 = 18.031, df = 2, p <0.001; post hoc, middle section, p <0.001; n = 84). As there was only one European plaice, the species was grouped with turbot and will be referred to as Pleuronectiforme species (spp.) for the rest of the chapter. The Pleuronectiforme spp. were recorded more frequently in the middle and upper sections during the deployment phase in comparison to the reference phase (χ^2= 43.367, df = 2, p <0.001; post hoc, middle section, p = 0.002; upper section, p <0.001; n = 84). There was no significant change in the spatial distribution of European sea bass. During the deployment phase an average of 10.52 (40.48 % of total population, n = 84) European sea bass were present in the 3 m^2 interaction zone close to the AFI.

4.4.1.2 Cartilaginous fish

Of the four species of cartilaginous fish (Chondrichthyes), *Mustelus* (smooth-hound) spp. (common smooth-hound shark and starry smooth-hound shark) were recorded significantly more in the middle section of the tank during the deployment phase in comparison to the reference phase ($\chi^2 = 11.265$, df = 2, p = 0.003; post hoc, middle section, p = 0.004; n = 84).

4.4.2 Field Experiment

4.4.2.1 The Prince of Wales Dock

Three Actinopterygii species were recorded during the remote underwater video monitoring which included European sea bass, European eel and mullet (Mugilidae) spp. No fish were recorded underneath the large AFI and therefore, it has not been included in any further analysis. Notably six juvenile European eel were resting in the matrix material of the large AFI when it was removed from The Prince of Wales Dock on 3rd June 2019. These individuals were safely returned into the water once identified. Fish were present in the underwater video footage for over 16 weeks in The Prince of Wales Dock, from 1st June (Spring) to 17th September 2018 (Summer; Figure 4.6).

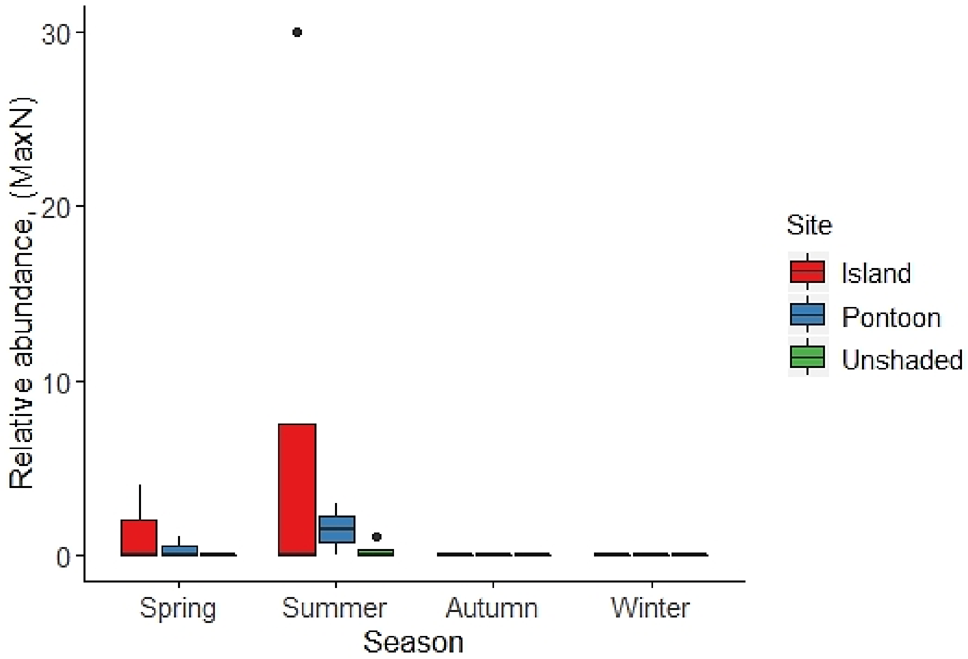


Figure 4.6: Fish relative abundance (MaxN) recorded in association with the small artificial floating island, pontoon and unshaded site, during video footage collected in spring, summer, autumn and winter 2018 (n = 12) at The Prince of Wales Dock. Additional survey conducted in spring 2019 not included.

The box and whisker plot includes the median (horizontal line), lower and upper quartiles (25 % and 75%, box) and the minimum and maximum values (whiskers).

Fish MaxN was significantly higher at the small AFI and pontoon, in comparison to the unshaded site (Kruskal Wallis = 26.256, df = 3, $p < 0.001$; Nemenyi's multiple comparison, small AFI and unshaded, p = <0.001; pontoon and unshaded, p = <0.001, n = 13). There was also a significantly higher MaxN in spring and summer in comparison to autumn and winter (Kruskal Wallis = 62.621, df = 3, $p <0.001$; Nemenyi's multiple comparison: spring and autumn; summer and autumn; spring and winter; summer and winter, $p <0.001$, n = 4; Figure 4.6). Note that the water temperatures in the dock were 18 ± 2.67 °C (mean ± standard error) in spring and 20.55 ± 1.32 °C (mean ± standard error) in summer, decreasing by 10 °C on average in autumn (see Appendix 3).

There was a significantly higher MaxN of European sea bass and unidentified shoaling fish recorded under the small AFI in comparison to the pontoon and unshaded site ($\chi^2 = 146.269$, df = 6, $p <0.001$; post hoc, European sea bass and small AFI, $p <0.001$; unidentified shoaling fish and small AFI, $p <0.001$; n = 13; Figure 4.7). The highest MaxN recorded during the monitoring period was a shoal of 30 juvenile European sea bass underneath the small AFI on 3rd September 2018. Juvenile European sea bass were only recorded under the small AFI, with a mean MaxN of 4.12 ± 1.08 (mean ± standard error). A mean MaxN of 2.23 ± 0.21 unidentified shoaling fish were recorded under the small AFI and 1.4 ± 0.4 under the pontoon (mean ± standard error; n = 13; Figure 4.7).

In contrast, there was a significantly higher MaxN of European eels and mullet spp. recorded under the pontoon in comparison to the small AFI and unshaded site ($\chi^2 = 146.269$, df = 6, $p <0.001$; post hoc, European eel and pontoon, $p <0.001$; mullet spp. and pontoon, p = <0.001; n = 13). The highest MaxN recorded under the pontoon was three mullet spp. on 25th July 2018. Mullet spp. and European eel were only recorded under the pontoon with a mean MaxN of 1.45 ± 0.17 for mullet spp. and 1.29 ± 0.07 for European eel (mean ± standard error). An unidentified shoaling fish was recorded on 9th August 2018 resulting in an overall MaxN of one for the unshaded site (Figure 4.7).

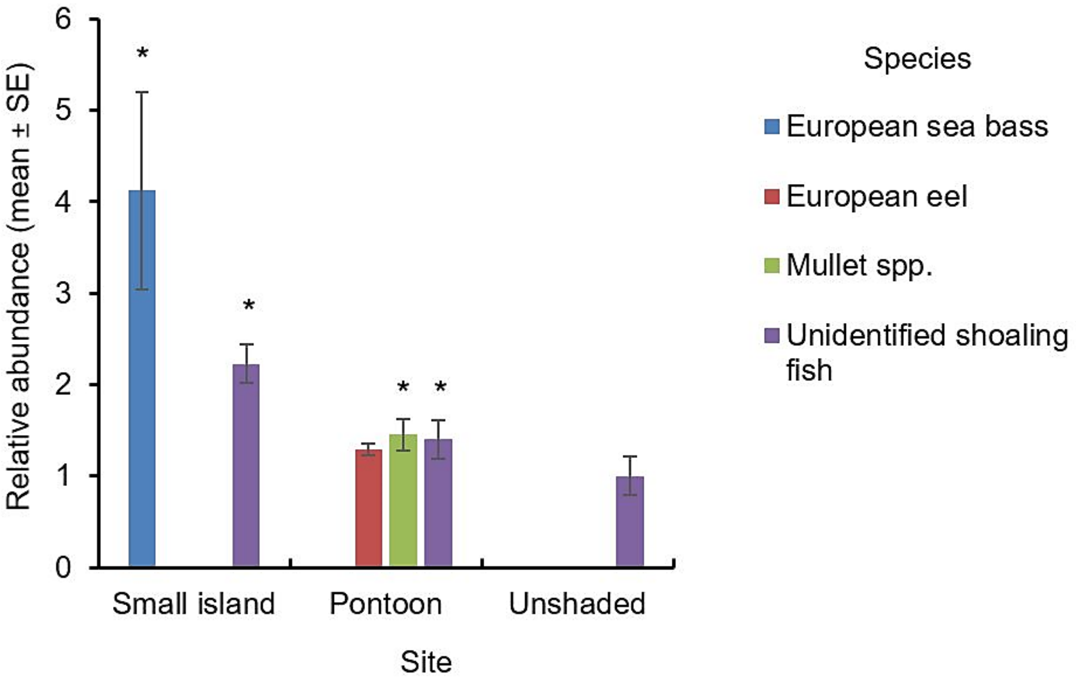


Figure 4.7: Relative abundance (MaxN, mean ± standard error) of European eel (*Anguilla anguilla*), European sea bass (*Dicentrarchus labrax*), mullet (Mugilidae) species (spp.) and unidentified shoaling fish in the underwater video footage at the small artificial floating island (AFI), pontoon and unshaded site at The Prince of Wales Dock (n = 13). * Significantly higher MaxN of European sea bass and unidentified shoaling fish at the small AFI and European eel and mullet spp. at the pontoon (p <0.001).

There was significantly more fish spp. commuting under the small AFI and pontoon in comparison to the unshaded site (χ^2 = 60.593, df = 4, p = <0.001; post hoc, small AFI, p <0.001; pontoon, p <0.001; n = 13). Mullet spp. were largely observed swimming under the pontoons and European eel used the pontoon for feeding and resting. European sea bass and unidentified shoaling fish were observed swimming and feeding on the small AFI (Figure 4.7; Table 4.2).

Table 4.2: Total number of recordings of European eel (*Anguilla anguilla*), European sea bass (*Dicentrarchus labrax*), mullet (Mugilidae) species (spp.) and unidentified shoaling fish either swimming, feeding or resting in the underwater video footage, associated with the small artificial floating island, pontoon and unshaded site in The Prince of Wales Dock (n = 13).

Species	Swimming	Feeding	Resting
European eel	26	42	35
European sea bass	84	54	0
Mullet spp.	36	1	0
Unidentified shoaling fish	68	17	0

Moon jellyfish (*Aurelia aurita*) and a sea gooseberry (*Pleurobrachia bachei*) were also identified during video analysis. Moon jellyfish were present throughout the year in The Prince of Wales Dock and sea gooseberries was present in June, September and November 2018. As invertebrates that lack the ability to visualise and interact with the AFIs, both moon jellyfish and sea gooseberry were not included in any further analysis. Moon jellyfish was the only taxon recorded underneath the large AFI during the monitoring period.

4.4.2.2 Swansea Marina

European sea bass and mullet spp. were the two taxa recorded in association with the AFI, pontoons and unshaded site in Swansea Marina. They were present in the underwater video footage from 1st June (Spring) to 17th September (Summer) 2018; the same 16 week period as The Prince of Wales Dock. Additionally, four European eels were resting inside the matrix unit of the AFI when it was removed from Swansea Marina on 5th June 2019 and safely returned to the water environment. Fish MaxN was significantly higher at the AFI and unshaded site in comparison the pontoon (Kruskal Wallis = 16.904, df = 2, $p < 0.001$; Nemenyi's multiple comparison, AFI and pontoon, $p < 0.001$; unshaded and pontoon, $p < 0.001$; n = 14). There was a significantly higher fish MaxN in spring and summer in comparison to autumn and winter (Kruskal Wallis = 45.253, df = 3, $p < 0.001$; Nemenyi's multiple comparison: spring and autumn, $p = 0.002$; summer and autumn; spring and winter; summer and winter, $p < 0.001$; n = 4; Figure 4.8). Note that water temperature in Swansea Marina was 17.07 ± 2.24 °C (mean ± standard error) in spring and 20.88 ± 1.51 °C (mean ± standard error) in the summer, decreasing by an average of 14 °C in autumn (see Appendix 3).

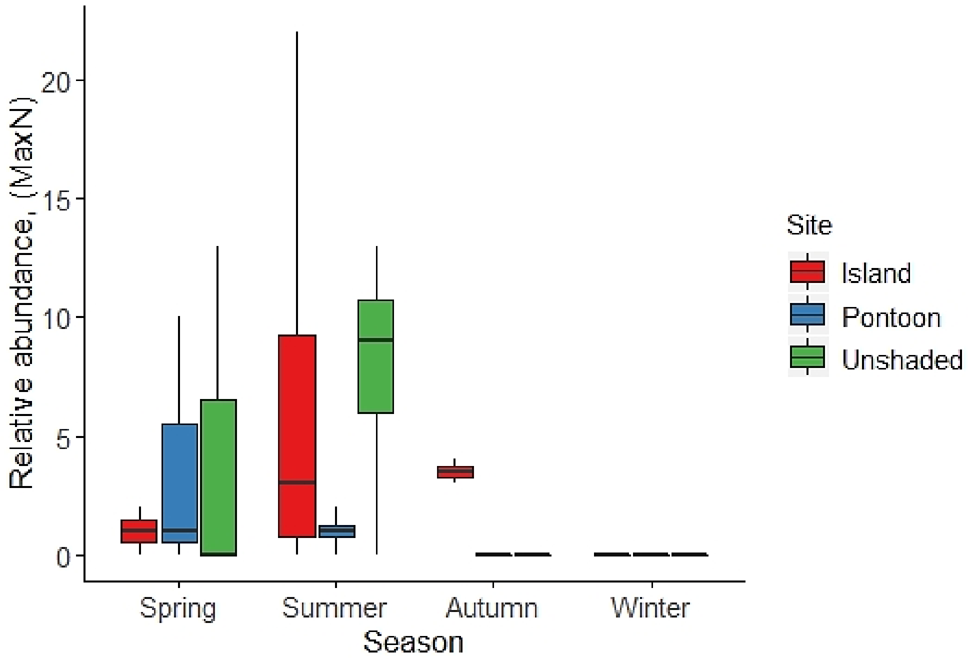


Figure 4.8: Fish relative abundance (MaxN) recorded in association with the artificial floating island, pontoon and unshaded site, during video footage collected in spring, summer, autumn and winter 2018 in Swansea Marina (n = 13). Additional survey conducted in spring 2019 not included. The box and whisker plot includes the median (horizontal line), lower and upper quartiles (25 % and 75%, box) and the minimum and maximum values (whiskers).

There was a significantly higher MaxN of European sea bass under the AFI in comparison to the pontoon and unshaded sites (χ^2 = 652.667, df = 4, p <0.001; post doc European sea bass and AFI, p <0.001; n = 14; Figure 4.9). The highest MaxN recorded during the monitoring period, was a shoal of 22 juvenile European sea bass underneath the AFI on 17th September 2018. Juvenile European sea bass were recorded associated with the AFI, pontoon and unshaded site. Overall, there were a higher number of European sea bass recorded under the AFI than the pontoon and unshaded site, averaging at 3.15 ± 0.18 (mean ± standard error, n = 14). There was a significantly higher MaxN of mullet spp. under the pontoons in comparison with the AFI and unshaded sites (χ^2 = 652.667, df = 4, p <0.001; post doc, mullet spp. and pontoon, p <0.001; n = 14; Figure 4.9). The highest MaxN recorded for mullet spp. was a shoal of ten individuals commuting under the pontoon on 1st June 2018. In addition, there was a significantly higher MaxN of unidentified shoaling fish at the unshaded site in comparison to the AFI and pontoon (χ^2 = 652.667, df = 4, p <0.001; post doc, unidentified shoaling fish. and unshaded site, p <0.001; n = 14; Figure 4.9). The highest MaxN recorded for unidentified shoaling fish was six individuals swimming in the unshaded site on the 27th July 2018.

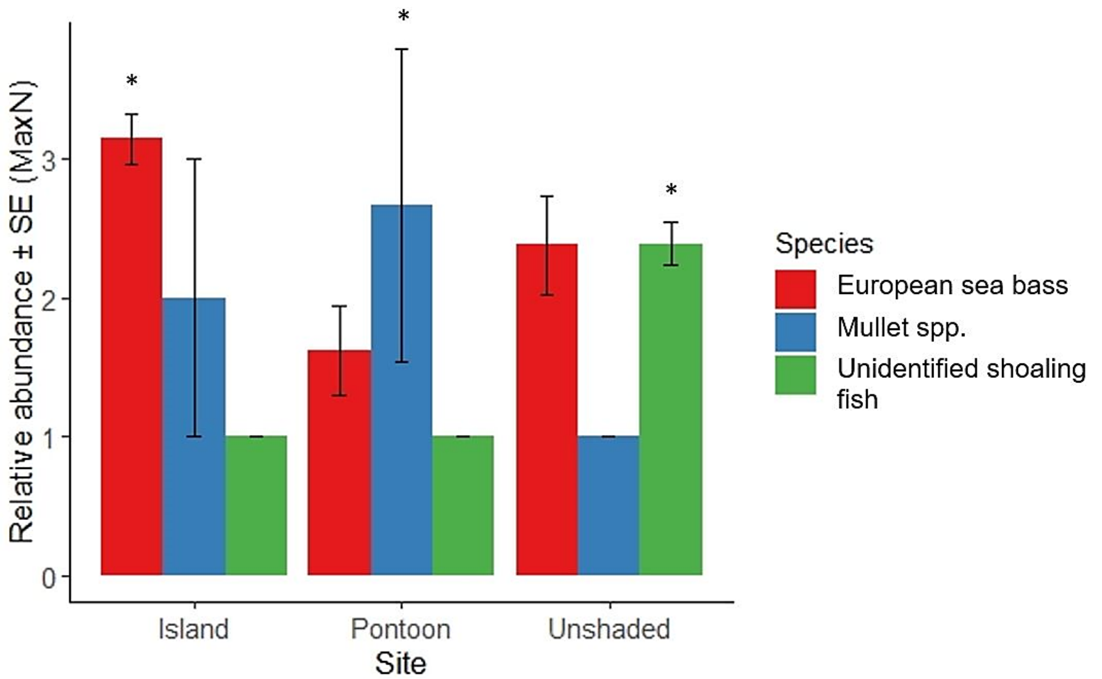


Figure 4.9: Fish relative abundance (MaxN, mean ± standard error) of European sea bass (*Dicentrarchus labrax*), mullet (Mugilidae) species (spp.) and unidentified shoaling fish at the artificial floating island, pontoon and unshaded site in Swansea Marina (n = 14). Significantly higher MaxN of European sea bass at the small AFI, mullet spp. at the pontoon and unidentified shoaling fish at the unshaded site ($p < 0.001$).

There was a significantly higher MaxN of fish feeding on the AFI than the pontoon and unshaded site ($\chi^2 = 299.711$, df = 2, $p < 0.001$; post hoc, AFI, $p < 0.001$; n = 14). In addition, there was a significantly higher MaxN of fish swimming under the pontoon and in the unshaded site, in comparison to the AFI ($\chi^2 = 299.711$, df = 2, $p < 0.001$; post hoc, pontoon, $p < 0.001$; unshaded, $p < 0.001$ n = 14; Table 4.3). European sea bass were observed swimming and feeding on the underside of the AFI and mullet spp. were observed swimming under the AFI and pontoon with no feeding activity observed.

Table 4.3: Total number of recordings of European sea bass (*Dicentrarchus labrax*), mullet (Mugilidae) species (spp.) and unidentified shoaling fish either swimming or feeding in the underwater video footage, associated with the artificial floating island, pontoon and unshaded site in Swansea Marina (n = 14).

Species	Swimming	Feeding
European sea bass	359	258
Mullet spp.	39	0
Unidentified shoaling fish	245	0

4.5 Discussion

Habitat creation using ecological engineering techniques such as artificial floating islands (AFIs) offers a potential method of adding complexity to heavily modified coastal water bodies, that are often associated with essential fish habitat (EFH). Currently, AFIs are largely installed in freshwater ecosystems to phytoremediate aquaculture wastewater and landfill leachate (Lu *et al.*, 2015), for aesthetic benefits in highly urbanised areas (Nakamura & Mueller, 2008; Burzaco & Frog Environmental, 2016) and to provide breeding grounds and resting places for birds (Hancock, 2000; Overton *et al.*, 2015). Floating islands created with natural weeds have also been used in freshwater lakes such as the Loktak lake in northwest India, as a fish aggregation device (FAD) for artisanal fishermen (Suresh, 2000). There is a plethora of evidence to show that fish associate with artificial or natural FADs such as objects moored on the seabed, suspended in the water column or floating at the surface (Nelson, 2003; Robert *et al.*, 2013; Montes *et al.*, 2019). The broad aim of this study was to determine whether vegetated AFIs are a viable method of habitat creation to support EFH that can form in heavily modified coastal water bodies.

4.5.1 Aquarium Experiment

Horse mackerel was the most abundant species that was recorded more frequently in the upper and middle sections of the tank during the deployment phase in comparison to the reference phase, supporting the first hypothesis of this study. During the reference phase, their activity was concentrated in the middle and lower sections of the tank. Horse mackerel is a pelagic-neritic, marine adventitious species that spawns at sea and seasonally migrates in large shoals between inshore and offshore regions (Lythgoe, 1971; Elliott & Dewailly, 1995; Elliott & Hemingway, 2002). In the summer months, horse mackerel migrate inshore and feed close to the surface on a range of prey including crustacea, fish fry and cephalopods (Lythgoe, 1971). During the winter months they migrate offshore and occupy deeper waters near to the seabed (Miller & Loates, 1997). Horse mackerel also shoal with other *Trachurus* spp. and in association with FADs (Castro *et al.*, 2002b), fish farms (Dempster *et al.*, 2002) and floating algae (Dooley, 1972; Kingsford, 1992). Research on the mechanisms behind adaptive shoaling focuses on predator-prey relationships (Pitcher, 1973; Seghers, 1974; Robertson *et al.*, 1976), the potential hydrodynamic benefits of swimming in association with a similar sized individual (Weihs, 1973; Pitcher *et al.*, 1985) and facilitating reproduction and migration (Robertson *et al.*, 1976). Horse mackerel have a number of natural predators including larger predatory fish, seals, whales and dolphins (Campbell, 2008). In the conditions of the experiment where all the species present were fed on a regular basis, there was no predation or foraging pressures on horse mackerel. However, the introduction of a new floating object in the tank may have stimulated a shoaling response close to the surface.

Gilthead sea bream and black bream used the middle section of the tank more frequently when the AFI was deployed in comparison to the reference phase, supporting the first hypothesis of this study. Gilthead sea bream is a demersal, marine adventitious species and black sea bream is a benthopelagic, marine juvenile migrant that both inhabit waters in close proximity of the seafloor, particularly sandy substratum and seagrass beds (Elliott & Dewailly, 1995; Guidetti, 2000; Elliott & Hemingway, 2002; La Mesa *et al.*, 2011). They are sedentary species that tend to swim individually or as smaller shoals and feed on a variety of prey including crustacea, molluscs and small fish (Lythgoe, 1971). Although there was no food resource available on the AFI installed during the experiment, the artificial root system may have attracted gilthead and black sea bream closer to the surface, due to its linear structure.

When the AFI was installed, gilthead sea bream swam more frequently as a small shoal than during the reference phase. Previously environmental enrichment such as ropes installed in sea cages associated with fisheries reduced aggressive behaviour of juvenile gilthead sea bream towards conspecifics and the housing net, improving the condition of their pectoral and caudal fins and modifying their horizontal distribution in the sea cage (Arechavala-Lopez *et al.*, 2019). Black sea bream commonly inhabit ship wrecks and have also been observed underneath unused sea cages, demonstrating that their association was not driven by feeding pellets given to the farmed species (Tuya *et al.*, 2006). Therefore, AFIs could offer an additional method of adding environmental enrichment to sea cages, improving the psychological and physiological state and habitat utilisation of key aquaculture species such as gilthead sea bream.

In the afternoon, the number of gilthead sea bream was also significantly higher in the middle section of the tank when the AFI was deployed in comparison to the reference phase. Fish activity including feeding, breeding, aggregating and resting are known to vary according to the diel cycle (Helfman, 1986). For example, the abundance and activity of zebrafish (*Girella zebra*), Southern eagle ray (*Myliobatis australis*) and blue-lined leatherjacket (*Meuschenia galii*) varied in the morning and afternoon in shallow water reefs of south-western Australia, potentially as a result of differences in digestive patterns (Birt *et al.*, 2012). Zebrafish have also demonstrated object recognition memory (May *et al.*, 2016). Therefore, the vertical distribution and shoaling behaviour of gilthead sea bream in this experiment may have varied due to exposure time with the AFI.

European pollock is a widespread benthopelagic, marine juvenile migrant species, inhabiting water bodies from Scandinavia to the Mediterranean (Lythgoe, 1971; Elliott & Hemingway, 2002). They are characteristically found close to the rocky shore and swim as individuals or small shoals (Dunn *et al.*, 1992; Charrier *et al.*, 2006). As part of this study, European pollock was observed more frequently in the middle section of the tank during the deployment phase in comparison to the reference phase, supporting the first hypothesis of this study. For fish species associated with benthic habitats, floating structures colonised by epibionts could potentially act as a substitute for the sea bed (Vandendriessche *et al.*, 2007). In the wild, *Pollachius* spp. such as saithe (*Pollachius virens*) have been monitored in high abundance close to sea cages harbouring rainbow trout (*Oncorhynchus mykiss*) (Carss, 1990; Bjordal & Skar, 1992; Dempster *et al.*, 2002), ship wrecks and offshore wind farms (Wilson & Elliott, 2009). The high abundance of *Pollachius* spp. by sea cages is largely due to the concentration of food pellets in contrast to other anthropogenic structures that provide both refuge and feeding opportunities on biofouling communities. Saithe also forage on sea lice associated with fish bred in the aquaculture sector (Carss, 1990). In the captive environment of the tank, European pollock may have changed its vertical distribution in the tank to mimic another species such as horse mackerel and gilthead sea bream or to use the AFI for refuge like observations in the field around anthropogenic structures (Wilson & Elliott, 2009).

European plaice and turbot are both demersal, marine juvenile migrants that feed predominantly on bivalves, polychaetes and crustaceans, referred to as zoobenthivores (Miller & Loates, 1997; Elliott & Hemingway, 2002; Elliott *et al.*, 2007b). During this study Pleuronectiforme species used the middle and upper sections more frequently in the deployment phase in comparison to the reference phase, supporting the first hypothesis of this study. They inhabit regions of the Northern Atlantic across to the Western Mediterranean, remain in shallow inshore waters as juveniles and migrate to deeper waters as adults (Lythgoe, 1971; Miller & Loates, 1997). AFIs offer refuge for pelagic species, which flatfish gain from burial under sediment and therefore, the utilisation of FADs by demersal species is currently understudied. The change in vertical distribution of European plaice and turbot during the deployment phase could be due to disturbance from staff, the public or in response to other species in the tank becoming increasingly active. This may have similarly been the case for the common smooth-hound and starry smooth-hound that used the middle section of the tank more frequently during the deployment phase in comparison to the reference phase. Both are demersal, marine adventitious species that inhabit coastal waters of the Mediterranean and feed on benthic invertebrates and small fish (Lythgoe, 1971; Elliott & Dewailly, 1995; Elliott & Hemingway, 2002).

4.5.2 Field Experiment

At The Prince of Wales Dock the fish relative abundance (MaxN) was significantly higher at the small AFI and pontoon, in comparison to the unshaded site. At Swansea Marina the fish MaxN was significantly higher at the AFI and unshaded site in comparison the pontoon. However, at The Prince of Wales Dock and Swansea Marina there was a significantly higher MaxN of juvenile European sea bass under the AFIs in comparison to the pontoon and unshaded site, with activity concentrated in the spring and summer months. Therefore, the second hypothesis was accepted in relation to European sea bass and rejected for mullet spp. and European eel. Fish species that interact with an installed FAD can be dependent on the season, abiotic conditions and life stage of the individual (Nelson, 2003). As structures that offer shelter and feeding opportunities, they have previously been associated with juveniles (Nelson, 2003), which could similarly be the case for AFIs. At both sites juvenile phase European sea bass swam underneath and fed directly on the underside of the AFIs. European sea bass are a commercially exploited demersal, marine juvenile migrant, with a large geographical range due to their euryhaline and eurythermic capabilities, from the North East Atlantic to the Mediterranean sea (Elliott & Hemingway, 2002; Sanchez Vazquez & MIunoz-Cueto, 2019). During their juvenile phase (1 – 5 years) European sea bass inhabit inshore, shallow coastal lagoons and estuaries and seem to favour *Spartina* marshes (Kelley, 1988; Colclough *et al.*, 2005); a genus of halophyte that was also used to vegetate the installed AFIs.

Juvenile fish often use the upper water column during early stages of development and as such, some species have formed adaptive mechanisms to support this behaviour (Zaitsev, 1970; Castro *et al.*, 2002b; Vandendriessche *et al.*, 2007). For example, most round fish have developed an air sac close to their dorsal fins that allows them to stay close to the surface for extended periods of time (Zaitsev, 1970). Juvenile European sea bass also preferentially use shallow waters during spring and summer, with a high degree of foraging activity particularly in the summer (Cabral & Costa, 2001). For Swansea Marina, this may be relevant as water depth fluctuated between 1-4 m. The water depth at The Prince of Wales Dock is approximately 7.6 m and therefore, the presence of European sea bass under the AFI was driven by an additional factor. Six crustacea species biofouled The Prince of Wales Dock AFI, forming the dominant subphylum (Chapter 3, Figure 3.2). Crustacea are a key prey group for juvenile European sea bass (Cabral & Costa, 2001) and may have attracted the large shoal to the upper surface of the dock during periods of high foraging activity in September.

Fish activity and growth is also dependent on water temperature, salinity and dissolved oxygen availability (Cabral & Costa, 2001; Pörtner, 2001; Vinagre *et al.*, 2012). Under controlled conditions juvenile European sea bass reach their peak growth around 24 °C (Vinagre *et al.*,

2012) as water temperature is positively related to standard and active metabolic rate (Claireaux *et al.*, 2006). During the monitoring period, the highest water temperature in The Prince of Wales Dock was 20.55 ± 1.32 °C (mean ± standard error) in the summer, corresponding with greater MaxN recorded at the three sites. Similarly, peak water temperatures in Swansea Marina reached 20.88 ± 1.51 °C (mean ± standard error) in the summer, aligning with fish activity associated with the AFI, pontoons and unshaded areas. The lack of fish observations during autumn and winter could have been due decreasing water temperatures causing a decline in metabolic rate and the swimming ability (Claireaux *et al.*, 2006) of fish present in at both survey sites. Therefore, season and the corresponding increase in surface temperatures produced favourable conditions for European sea bass to use the AFI for feeding during spring and summer. In addition, stratification of the water column, high nutrient input during the summer and weak currents may have caused oxygen depletion at lower water depths of the dock and marina and produced unfavourable abiotic conditions (Rossignol-Strick, 1985; Elliott & Hemingway, 2002).

There was higher fish activity in the unshaded area in Swansea Marina than The Prince of Wales Dock. This could be due to a lack of other floating structures in The Prince of Wales Dock, encouraging individuals to use sheltered sites more readily. Only 1.08 % of the 10.9 ha dock is covered by pontoons. For comparison, Swansea Marina is 7.01 ha and 32.4 % of its total area is covered with pontoons and recreational boats. The topographic complexity of Swansea Marina could be the key factor influencing the sporadic habitat utilisation of European sea bass and mullet spp., in comparison to fish activity observed in The Prince of Wales Dock.

At The Prince of Wales Dock, adult mullet spp. and European eel were only observed swimming under the pontoon; European eel notably swimming, foraging and resting (Table 3.2). Mullet spp. have a large geographical range, due to their ability to withstand high and low salinity gradients (McDowall, 1989; Cardona, 2006). In high salinity environments fish have exert more energy to osmoregulate and grow (Wootton, 1998; Cardona, 2006). Salinity can also influence the spatial distribution of the species, with juvenile flathead grey mullet (*Mugil cephalus*) preferentially using areas with salinities <15 during laboratory experiments (Cardona, 2000, 2006). This could account for the higher activity of mullet spp. across the three monitored sites in Swansea Marina, that had an average salinity range of 9 – 15.67 (mean) during the monitoring period (Figure 4.9; Appendix 3). For comparison, the salinity in The Prince of Wales Dock ranged from 28 – 32.25 (mean; Appendix 3). Unlike European sea bass, there were no juvenile phase mullet spp. recorded at both sites. Larger piscivorous fish have notably been observed foraging at night-time and migrate to deeper waters during day light hours, in coastal saltmarsh habitats (Colclough *et al.*, 2005). Commuting close to the surface and in shallow waters can leave larger fish vulnerable to predation and potentially cause stranding (Copp & Jurajda, 1993; Paterson & Whitfield, 2000; Colclough *et al.*, 2005). For the adult mullet spp. present in Swansea Marina and Prince of Wales Dock, interacting with the AFI close to the surface may have posed a high predation risk although it is noted that mullet spp. are regularly sited close to the surface in heavily modified coastal water bodies. Mullet spp. as scavengers also preferentially feed on benthic diatoms, algae and detritus and therefore, forage more readily on organic matter associated with the bottom sediment.

European eel is a catadromous species present in the Atlantic and Mediterranean Sea (Laffaille *et al.*, 2005), with a similar distribution to European sea bass and mullet spp. The larvae produced in the Sargasso Sea drift inland metamorphosing into glass eels and in turn elvers, which remain in coastal lagoons and estuaries while they continue to grow. The elvers form yellow eels and finally silver eels, that migrate back to offshore habitats (Tesch & Greenwood, 1977). In The Prince of Wales Dock, the European eel were recorded while in their yellow eel stage interacting with the pontoon. Although not observed in the underwater video footage, elvers also inhabited the matrix material of the AFIs in both The Prince of Wales Dock and Swansea Marina. During these juvenile phases European eel are often recorded in shallow habitats that support macroalgae, providing shelter and foraging sites on the associated invertebrates (Laffaille *et al.*, 2003, 2005). They also exhibit a more sedentary lifestyle while associated with inshore habitats (Laffaille *et al.*, 2005), which may explain why elvers were resting in the matrix material of the AFIs as it supplies shelter from predation and feeding opportunities.

4.6 Conclusion

The presence of an AFI during the deployment phase of the laboratory experiment resulted in a number of fish species altering their vertical distribution to the middle and upper sections of the tank to use the AFI for shelter. Therefore, the first hypothesis of this study was accepted. In the field experiment, the AFIs installed in both The Prince of Wales Dock and Swansea Marina attracted a higher MaxN of juvenile European sea bass than the pontoon and unshaded site and therefore, the second hypothesis was accepted in relation to European sea bass and rejected for mullet spp. and European eel. The installation of AFIs in heavily modified coastal water bodies does have the potential to support nursery sites or EFHs. The AFIs provided shelter and feeding opportunities on biofouling communities that colonised the underside of the structure, providing ecosystem services for fish populations. Water temperature, salinity and the life stage of the individuals were noted as key factors influencing the MaxN of fish in association with the AFI and should be considered during future installations of AFIs in heavily modified coastal water bodies.

Chapter 5: Artificial floating islands as 'bird havens' connecting natural and urbanised environments

Abstract

There are multiple threats to shorebirds in the United Kingdom including urbanisation, anthropogenic disturbance and overexploitation of natural resources, which are causing fragmentation of coastal wetlands and declines in bird populations. Artificial floating islands (AFIs) could be a method of creating new patch habitats, providing vital stop-over sites during migration and aiding the connection of natural and urbanised habitats. This study aimed to test the hypothesis that vegetated AFIs installed in heavily modified coastal water bodies would attract a higher density and species diversity of birds than alternative hard structures within the same survey area. Two AFIs were installed in The Prince of Wales Dock (8 m^2 and 13.2 m^2) and 28 vantage point surveys were conducted from 18th May 2018 – 31st May 2019. An 8 m^2 AFI was installed in Swansea Marina and 23 vantage point surveys were conducted from 23rd May 2018 – 2nd June 2019. At 15 minute intervals during 1.5 – 2 hour surveys the observed species and behaviour of each individual was recorded on the AFI, hard structures and surrounding water environment. Ethograms were also conducted on birds associated with the AFIs. There was a significantly higher density of birds on the AFIs at both locations in comparison to other hard structures and therefore, the hypothesis was accepted. Herring gulls (*Larus argentatus*) were observed pecking the AFIs more frequently than other behaviours recorded in The Prince of Wales Dock; notably on the blue mussels (*Mytilus edulis*) that fouled the underside of the AFI. Species diversity on the AFI was significantly lower than hard structures, with the AFIs predominantly used by large Larids. In Swansea Marina, mallards (*Anas platyrhynchos*) and black-headed gulls (*Chroicocephalus ridibundus*) predominantly used the AFI and had a significantly higher species diversity in comparison to hard structures. For future AFI installations careful consideration should be made on the location and degree of isolation of the AFI and the energetic costs associated, plus the specific requirements of the target species including AFI size and vegetative cover, which can influence nest density and species diversity.

5.1 Introduction

Coastal wetlands are highly productive and biodiverse ecosystems that support several globally threatened bird species (Gibbs, 1993; Green, 1996; Paracuellos & Tellería, 2004). Due to the European Union's (EU) network of Protected Areas known as the Natura 2000 network, birds and their habitats including coastal wetlands are protected under the EU Birds Directive (2009/147/EC) (Ramirez *et al.*, 2017). The directive has resulted in the designation of 275 Special Protection Areas (SPAs) in the United Kingdom (UK), which protect rare and vulnerable birds listed as Annex 1 (JNCC, 2020). There are multiple threats to shorebirds (Charadriiformes) that have been identified including urbanisation (Melles *et al.*, 2003), anthropogenic disturbance (Stillman *et al.*, 2016), sea level rise (Grémillet & Boulinier, 2009) and overexploitation of natural resources (Sutherland *et al.*, 2012). Even with protected site designation common European bird populations are in decline such as the common starling (*Sturnus vulgaris*) (Smith *et al.*, 2012) and house sparrow (*Passer domesticus*) (Laet & Summers-Smith, 2007) that inhabit urbanised areas, highlighting the need for wider scale environmental improvement (Inger *et al.*, 2015). Therefore, it is important that the impact of these threats on natural and urbanised environments are understood (Melles *et al.*, 2003) in relation to the spatial distribution (Yoda *et al.*, 2012), predation vulnerability (Gering & Blair, 1999) and the foraging (Fuirst *et al.*, 2018) and breeding success of birds (Navarro *et al.*, 2017), in order to determine species specific conservation objectives. This chapter will focus on shorebirds associated with coastal habitats.

5.1.1 Gulls

Gulls (Laridae) are highly adaptable (Belant, 1997; Rock, 2005) and opportunistic scavengers, enabling them to adopt a dual foraging strategy between marine and terrestrial food sources (Washburn *et al.*, 2013; Fuirst *et al.*, 2018; Enners *et al.*, 2018b). The necessity to use a generalist feeding strategy is dependent on a number of factors, including the location of the breeding colony, food availability and inter and intraspecific competition (Tiedemann & Nehls, 1997; Enners *et al.*, 2018a). When comparing gull colonies in different locations, individuals adapt their foraging strategy according to the ratio of urbanised and natural habitats (Fuirst *et al.*, 2018). For example, herring gulls (*Larus argentatus*) inhabiting less urbanised areas along the east coast of the United States (US), foraged in a variety of habitats in comparison to individuals based in highly urbanised locations (Fuirst *et al.*, 2018). Additionally, herring gulls and laughing gulls (*Leucophaeus atricilla*) were found to favour marine food sources but alternated between marine, terrestrial and urbanised areas while foraging in New York City (Washburn *et al.*, 2013). This behavioural plasticity allows many gull species (spp.) to successfully coexist in urbanised areas, both on the coast and further inland. The extent of specialization and effective spatial awareness of yellow legged gulls (*Larus michahellis*), can reduce intraspecific competition and control overall population success (Navarro *et al.*, 2017).

Excluding urban nesting populations, the number of herring gull fledged chicks per pair has declined by 31 % from 1986 – 2008 in the UK and are now Red Listed in the Birds of Conservation Concern 4 (Cook & Robinson, 2010). Native breeding populations present across Scandinavia and native resident and non-breeding populations in western Europe are also demonstrating a decreasing population trend (BirdLife International, 2019a). Lesser black-backed gulls (*Larus fuscus*) have declined in population size in the UK and are now Amber listed in the Birds of Conservation Concern 4 (Eaton *et al.*, 2015). Nevertheless, on a global scale lesser black-backed gulls are increasing in abundance covering a greater extent than herring gulls and inhabiting areas of northern Russia to South Africa (BirdLife International, 2019b).

Population success is largely controlled by foraging strategies and food availability during the breeding season (Juvaste *et al.*, 2017). If marine food source availability is constrained by urbanisation, individuals are likely to fly greater distances to feed, increasing the risk of predation on any unprotected chicks (Morris & Black, 1980; Pierotti & Annett, 1991). For example, GPS collared herring gulls breeding close to the mainland of the German North Sea flew a mean total distance of 26.7 km to forage; unlike pairs with access to healthy intertidal habitat on the furthest island, which remained close to the breeding colony (Enners *et al.*, 2018a). Intertidal flats and sandy beaches are also used by gull spp. for roosting and loafing activity. Sandy beaches with coastal armouring in California support fewer gull spp. than beaches without coastal armouring (Dugan *et al.*, 2008). This study highlighted the impact of habitat fragmentation in coastal environments and how increasing isolation of patch habitats can result in declines in spp. richness, in comparison to natural habitats (Wilcox & Murphy, 1985; Andren, 1999).

5.1.2 Waders

Coastal wetlands support a range of breeding populations of wader (Charadrii) and are of national and international importance (Moore & Fuller, 1983; Vickery *et al.*, 1997; Milsom *et al.*, 2000b). For example, the North Kent marshes are home to 13 % of the total lapwing (*Vanellus vanellus*) population and 27 % of redshank (*Truga totanus*) present across England and Wales (Henderson, 1982; Milsom *et al.*, 2000a). Additionally, >50 % of Eurasian curlews (*Numenius arquata*) breed in coastal marsh habitats in the UK (Gregory *et al.*, 2002; Wilson *et al.*, 2004).

Most waders have specialist feeding strategies and occupy foraging grounds on intertidal mudflats exposed during low tide, to feed on benthic invertebrates present in the soft sediment or associated reef feature (Dias *et al.*, 2006). Marine food sources include polychaetes, molluscs and crustacea. During high tide waders commute to high water roost sites often located in the upper saltmarsh, beaches or fields to rest or continue to forage on terrestrial invertebrates (Britton & Johnson, 1987; Rehfisch *et al.*, 1996; Dias *et al.*, 2006). Therefore, it is essential that factors that could limit accessibility to low tide foraging sites and high tide roosting locations are minimised in order to prevent the decline in carrying capacity of estuaries and coastal wetlands (Goss-Custard *et al.*, 2002; West *et al.*, 2005). There a number of definitions for carrying capacity depending on the circumstance it is applied. For example the ‘one in, one out’ principle is often used to define the carry capacity of migratory birds and their breeding territories (Goss-Custard & West, 1997). For non-breeding, overwintering waders carrying capacity is the maximum number of bird-days a site can support during the winter or the maximum number of birds that can survive the winter in good condition at the site (Goss-Custard, 1985; Goss-Custard *et al.*, 2002). In this chapter it will be referred to as the number of birds that can be supported at a specified location and time of year and how that is influenced by the size of the species, competition (Dawson *et al.*, 2011), territory extent and resource availability (Goss-Custard & West, 1997; Duhem *et al.*, 2007; Eason *et al.*, 2012).

Currently, both the pressures of expanding coastal developments and agricultural practices are having a major impact on breeding wader populations in the UK (Milsom *et al.*, 2000a; Wilson *et al.*, 2004). The fertility of coastal wetlands makes them desirable land for arable farming and grazing and has resulted in approximately 50 % of the global wetlands being reclaimed for agricultural and other land uses (Verhoeven & Setter, 2010; Han *et al.*, 2014). Coastal developments and agriculture cause the loss or fragmentation of intertidal mudflats, saltmarshes and beaches increasing the energetic costs for waders that commute greater distances to access foraging and roost sites (Piersma *et al.*, 1993). This can also result in reduced prey availability and therefore, a decline in the carrying capacity of the site (Dugan *et al.*, 2003, 2008). For example, a decline in dunlin (*Calidris alpina*) density was observed at suitable mudflat foraging grounds with increasing distance from the closest roost location, highlighting the impact of increased energetic cost on site utilisation (Dias *et al.*, 2006). This relationship is associated with the term ‘energy landscape’, which was adopted to describe variations in movement driven by environmental parameters such as wind, incline and vegetation (Wilson *et al.*, 2012; Shepard *et al.*, 2013).

Changes to agricultural management strategies such as the use of fertilisers and growth of silage alternatively to hay also effect sward structure (Wilson *et al.*, 2004). The required sward height while breeding and foraging varies between species (Green & Robins, 1993). As a result of these combined impacts on coastal wetlands and low grasslands, black tailed godwit (*Limosa limosa*) are now Red listed (Gregory *et al.*, 2002) and ruff (*Philomachus pugnax*), snipe (*Gallinago gallinago*) lapwing and redshank are all Amber listed under the Birds of Conservation Concern (Wilson *et al.*, 2004).

5.1.3 Artificial Habitat Creation

In addition to the decline of protected shorebird spp., common bird populations are also in decline and there is a necessity to focus conservation efforts in both natural and urbanised environments (Inger *et al.*, 2015). Artificial floating islands (AFIs) are platforms designed to make an immediate change to the local environment, via habitat creation. They can be installed with transplanted vegetation, coir matts, shingle or structures to provide shelter. For both heavily modified coastal water bodies lacking in habitat complexity and coastal wetlands suffering from habitat fragmentation, inundation and changes in vegetation management (Ferrarin *et al.*, 2013), AFIs could provide connectivity via new patch habitats that support roosting, nesting and feeding activity of shorebirds. In natural habitats such as forests, the more structurally complex the habitat, the greater abundance and species richness of birds it can support (Ghadiri Khanaposhtani *et al.*, 2012).

Previously AFIs have assisted in the conservation of California ridgeways rail (*Rallus obsoletus obsoletus*) (Overton *et al.*, 2015). The AFIs were installed with woven palm leaves attached to a frame, to provide shelter and reduce predation risk during inundation of coastal wetlands in Oakland, California (Overton *et al.*, 2015). AFIs have also been installed in Scotland to support breeding territories of black throated loon (*Gavia arctica*), resulting in an increase in chick productivity by a factor of 2.7 where the rafts were deployed (Hancock, 2000). These examples highlight the ecological functioning role AFIs can play for a variety of bird spp. However, it is important to consider the following factors when installing an AFI: the species-area relationship whereby larger islands typically support more species (Higgs, 1981); the habitat complexity of the site proposed for the AFIs installation and connectivity with urban and natural habitats; and the ecological functioning role of the installed AFI, in comparison to other features within the habitat.

5.2 Aims and Objectives

The aim of this study was to determine if AFIs installed in heavily modified coastal water bodies could be used to create new patch habitats for birds between natural and urbanised environments. The testable hypothesis was that vegetated AFIs installed in heavily modified coastal water bodies attract a higher density and species diversity of birds than alternative hard structures within the same survey area. The study tested this hypothesis with the following three key objectives:

1) To assess differences in density and species diversity of birds using the AFIs in comparison to hard structures within the survey area.
2) To determine if there are any behavioural differences between birds using the AFIs, in comparison to hard structures and water habitat in order to assess the ecological functioning role of the installed, vegetated AFIs.
3) To assess if the density of birds on the AFIs varied between the complex habitat (Swansea Marina) and less complex habitat (Prince of Wales Dock) and to determine whether differences in density (if any) were a result of AFI size or location.

5.3 Materials and Methods

5.3.1 Study Site

Several designated SPAs are located in south Wales including Carmarthen Bay (51°39’18.673”N, 4°29’4.679”W) and Bury Inlet SPA (51°39’3.524”N, 4°11’16.671”W). Bury Inlet is part of the Carmarthen Bay and Estuaries Special Area of Conservation (SAC) and features the largest area of intertidal saltmarsh in Wales (Figure 5.1). It was designated as a European wide important site for wintering common scoters (*Melanitta nigra*) (Burton *et al.*, 2010; Bullimore, 2014). Blackpill (51°35’35.862”N, 3°59’11.577”W) and Crymlyn Bog (51°37’8.519”N, 3°51’38.12”W) are Sites of Special Scientific Interest (SSSI) and located along the coast of Swansea Bay (51°35’24.299”N, 3°55’1.398”W). Blackpill was designated due to its importance as a stop-over site and feeding ground for resident and migratory birds such as ringed plover (*Charadrius hiaticula*) and sanderling (*Calidris alba*) (Warbrick *et al.*, 1992). Crymlyn Bog was designated due its fen communities, woodland and presence of slender cotton grass (*Eriophorum gracile*) and hornet robberfly (*Asilus crabroniformis*) (Countryside Council for Wales, 1975). Honeycomb worm (*Sabellaria alveolata*) reefs also present in Swansea Bay support a high diversity of birds such as oyster catcher (*Haematopus longirostris*), redshank and dunlin (*Calidris alpina*) and benthic invertebrate communities. These natural habitats are closely associated with Swansea’s urbanised centre, located less than 500 m from the adjacent SSSIs. As a result, these protected areas are regularly disturbed by anthropogenic activity such as recreational sports, bait digging and dog walking.

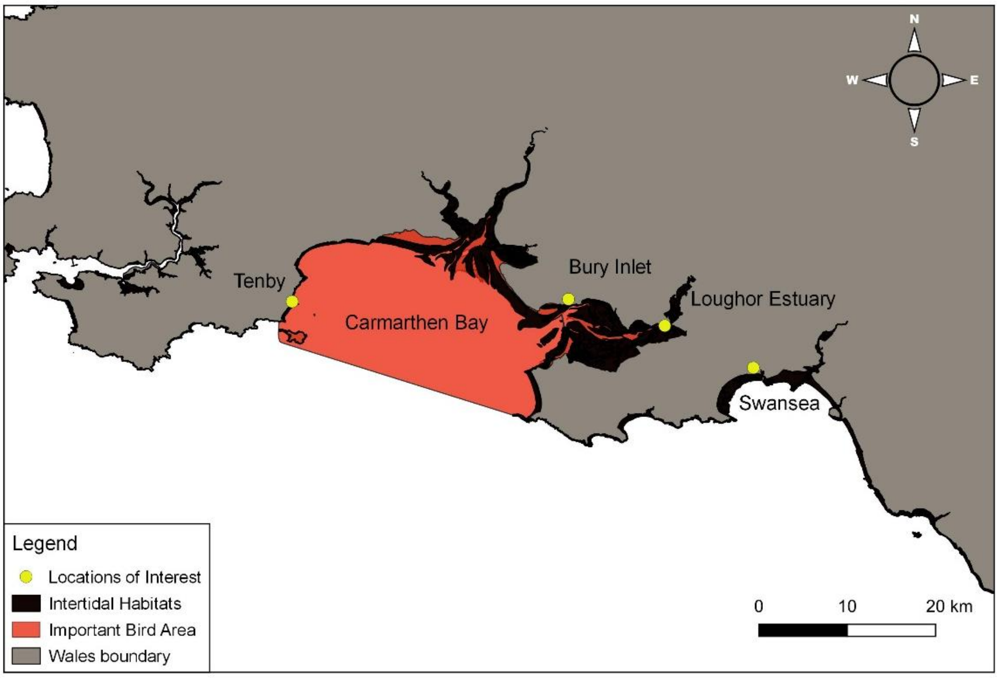


Figure 5.1: Map showing the intertidal habitats of Swansea Bay and Carmarthen Bay, plus the Important Bird Area which includes Carmarthen Bay and Bury Inlet Special Protected Areas (SPA). These are the closest SPAs to Swansea.

5.3.2 Vantage Point Surveys

Two, 8 m² Biohavens® were installed on 28th and 29th September 2017; one located in Swansea Marina (51°36'56.3"N, 3° 56'26.0"W) and one in The Prince of Wales Dock (51°37'10.6"N, 3°55'30.0"W). A 13.2 m² Biohaven® was also installed on 17th May 2018 in The Prince of Wales Dock (51°37'09.8"N, 3°55'29.8"W). At The Prince of Wales Dock, 28 1.5 – 2 hour vantage point surveys were undertaken using RSPB WPG 8 x 32 binoculars from 18th May 2018 – 31st May 2019. A total of 216 bird counts were recorded during 52.75 hours of surveying**.** 23 surveys were completed at Swansea Marina from 23rd May 2018 – 14th May 2019. A total of 169 bird counts were recorded during 42.25 hours of surveying. The surveys were categorised as morning or afternoon surveys based on the start time of the survey. Surveys that started between 06.00 – 11.00 were morning surveys and any from 12.00 – 18.00 were afternoon surveys. Every 15 minutes during the survey the number of individuals of each species, the species behaviour and location were noted. Behaviour was subdivided into foraging, preening, resting, sleeping or swimming. At each vantage point survey location, bird data were recorded across an 180° radius using binoculars (Figure 5.2). In The Prince of Wales Dock and Swansea Marina there is high anthropogenic disturbance from pedestrians, commercial and recreational boating activity and fishermen.

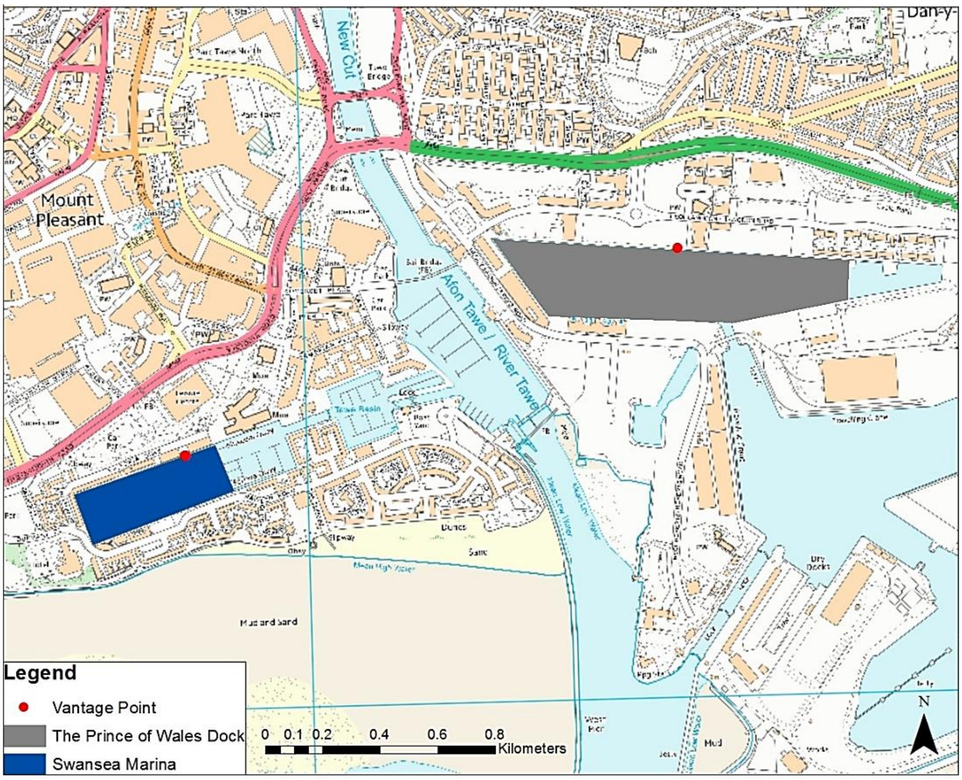


Figure 5.2: The locations of the vantage point bird surveys and the survey extent in The Prince of Wales Dock and Swansea Marina.

5.3.3 Habitat Complexity

In order to assess habitat complexity at The Prince of Wales Dock and Swansea Marina Google satellite images were used to categorise habitat features. Taking the AFIs location as the central point, visual surveys were undertaken of 12, 2500 m² grid squares within a 122500 m² sampling area. The 12 grid squares were randomly selected via a number generator and were assessed to determine the percentage cover of habitat features including buildings, concrete surfaces, pontoons, water, grass and trees. Direct comparisons were then made between the maximum density of birds per survey recorded on the AFIs and hard structures at The Prince of Wales Dock and Swansea Marina, when surveys were conducted on the same day ($n = 18$). This method was used to reduce bias caused by abiotic conditions such as rainfall, air temperature and wind speed. Bird density was calculated on the basis that the total AFI area in The Prince of Wales Dock was 21.2 m², hard structures was 495.91 m² and water habitat was 95446.66 m². In Swansea Marina, the total AFI area was 8 m², hard structures was 9493.83 m² and water habitat was 25789.15 m².

5.3.4 Statistical Analysis

Prior to statistical analysis, the data were tested for normality using the Shapiro Wilk normality test. As the data were non-parametric, Kruskal Wallis and Nemenyi's multiple pairwise comparison testing was used to determine if there was a significant difference in the density of birds on the vegetated AFIs in comparison to the hard structures in each habitat. Shannon-Wiener species diversity index was used to assess variations in species diversity on the AFI, hard structures and water habitats. The number of juvenile *Larus* spp. on each feature was added to either the number of adult herring gulls or lesser black-backed gulls based on the total proportion of the species observed at The Prince of Wales Dock and Swansea Marina. For example, in The Prince of Wales Dock 93.31 % of the *Larus* spp. were herring gulls and therefore, 93.31 % of the total number of juvenile *Larus* spp. recorded on the AFI was added to the total number of herring gulls in order to calculate the Shannon Wiener species diversity index. This analysis was used to test the hypothesis that vegetated AFIs installed in heavily modified coastal water bodies would attract a higher density and species diversity of birds than alternative hard structures within the same survey area.

Additionally, Kruskal Wallis and Nemenyi's multiple pairwise comparison testing was used to determine if there was a significant difference in the behaviour of birds on the AFIs in comparison to the hard structures and water habitat. Chi Squared test of homogeneity was used to assess the behaviours of each species and distribution of time spent conducting a certain behaviour while using the AFIs. This analysis was conducted to assess the ecological functioning role of the AFIs for each bird species recorded. Kruskal Wallis was used to test if there was a significant difference in the percentage cover of buildings, concrete surfaces, pontoons, water, grass and trees in a 122500 m^2 sampling area with the AFIs at The Prince of Wales Dock and Swansea Marina as the central point. Mann-Whitney U Test was used to compare the maximum density of birds using the AFIs and hard structures at The Prince of Wales Dock and Swansea Marina during surveys conducted on the same day. This analysis was used to test if the higher total area of AFIs installed in The Prince of Wales Dock resulted in a higher density of birds using the AFIs or if differences in density were influenced by the percentage cover of habitat features. Data analysis was conducted in R 3.5.1 Statistics Software.

5.4 Results

5.4.1 The Prince of Wales Dock

From 18th May 2018 – 31st May 2019, 1538 individuals of 12 bird species were counted during 28 vantage point surveys conducted at The Prince of Wales Dock; a total of 742 birds of six species were recorded on the water, 526 of eight species on the AFIs and 270 of nine species on the hard structures (Figure 5.3). These included herring gulls, lesser black-backed gulls, black-headed gulls (*Chroicocephalus ridibundus*), mute swans, mallards, great cormorants

(*Phalacrocorax carbo*), carrion crows (*Corvus corone*), turnstones (*Arenaria interpres*)**,** ringed plovers, jackdaws (*Corvus monedula*), magpies (*Pica pica*) and feral pigeons (*Columba livia*). There was a significantly higher density of birds on the AFIs in comparison to the hard structures (Kruskal Wallis = 76.51, df = 2, p <0.001; Nemenyi's multiple comparison, AFI and hard structures, p<0.001; n = 216; Figure 5.3). Hard structures in The Prince of Wales Dock consisted of pontoons, buoys and the pavement surrounding the dock. Additionally, there was a significantly higher density of birds on the AFIs in comparison to the water habitat (Nemenyi's multiple comparison, p <0.001; n = 216).

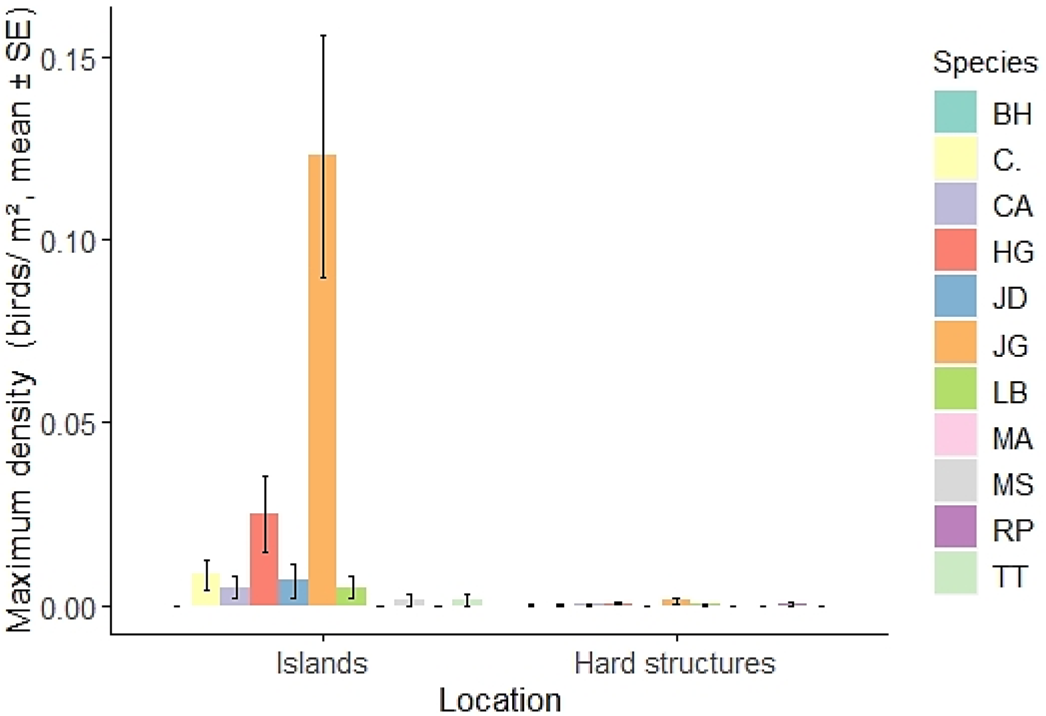


Figure 5.3: The maximum density of birds recorded per survey (birds/m², mean ± standard error) on the artificial floating islands in comparison to hard structures in The Prince of Wales Dock (n = 28). Each species is represented using the British Trust for Ornithology species coding system, excluding juvenile *Larus* spp. which is shown as JG: black-headed gull (BH), carrion crow (C.), cormorant (CA), herring gull (HG), jackdaw (JD), lesser black-backed gull (LB), mallard (MA), mute swan (MS), ringed plover (RP) and turnstone (TT). The graph excludes magpie and feral pigeon as they were not present when maximum density values were recorded per survey.

The highest density of birds was recorded on the 18th May 2018 and 3rd August 2018 (0.52 birds/ m²) associated with 11 juvenile *Larus* spp. using the AFIs (Figure 5.4). The highest density of birds on the hard structures was recorded on the 16th September 2018 (0.02 birds/ m²), associated with 10 juvenile *Larus* spp. The highest density of birds in the water was recorded on the 6th September 2018 (0.02), associated with 29 juvenile *Larus* spp (Figure 5.4).

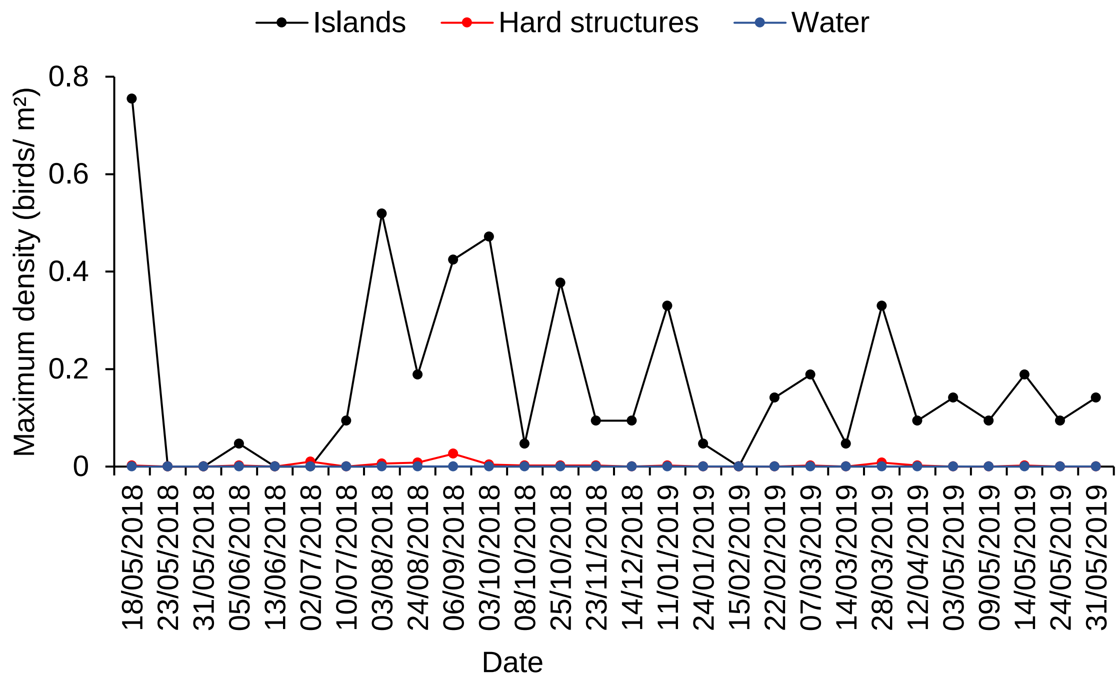


Figure 5.4: The maximum density of birds per survey recorded from 18th May 2018 – 31st May 2019 in The Prince of Wales Dock on the artificial floating islands, hard structures and water habitat (n = 28).

In relation to bird behaviour, there was a significantly higher density of birds foraging on the AFIs in comparison to the hard structures and in the water habitat (Kruskal Wallis = 21.031, df = 2, $p < 0.001$; Nemenyi's multiple comparison, AFIs and hard structures, $p < 0.001$; AFIs and water, p = 0.018; n = 28; Figure 5.5). Birds also used the AFIs, hard structures and water habitat for resting and preening. Overall the species diversity at The Prince of Wales Dock was low, with the highest species diversity recorded on the hard structures (Shannon Wiener species diversity index = 1.028; n = 28), followed by the water habitat (0.909) and the lowest species diversity on the AFIs (0.816).

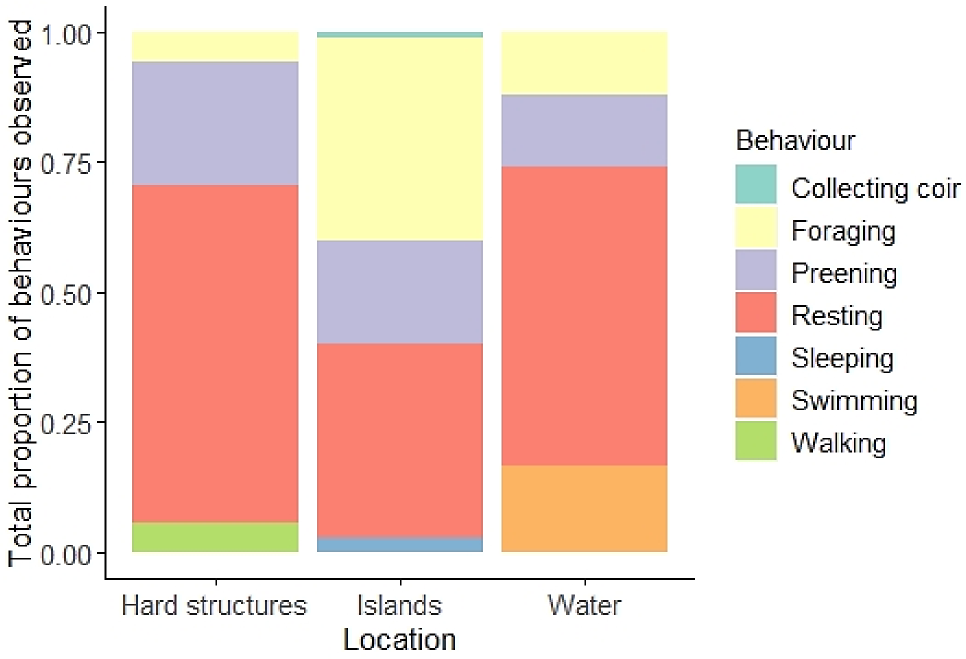


Figure 5.5: The total proportion of birds recorded as collecting coir, foraging, preening, resting, sleeping, swimming and walking on the hard structures, artificial floating islands and water habitat during the survey period from 18th May 2018 – 31st May 2019 in The Prince of Wales Dock (n = 216).

Of the 30 possible behaviours considered prior to conducting the ethograms (see Appendix 4), 26 behaviours were recorded for herring gulls, lesser black-backed gulls and cormorants in The Prince of Wales Dock. The distribution of behaviours exhibited by four of the herring gulls were significantly different than expected (Table 5.1), as each individual was observed pecking the AFI more often than other behaviours recorded (p = 0.005, 0.003, 0.014 and <0.001). In addition to pecking the AFI, three herring gulls also walked, stood alert and called more frequently than other behaviours noted for each individual.

A juvenile lesser black-backed gull also had a significantly different distribution of behaviours than expected, as it pecked the AFI and stood alert more frequently than the seven other behaviours observed ($\chi^2 = 87.902$, df = 6, $p < 0.001$). An adult lesser black-backed gull and juvenile herring gull exhibited no significant differences in behaviour. Both individuals were observed for a limited time as they commuted out of sight during the survey. In contrast to *Larus* spp., the great cormorant had a significantly different distribution of behaviours than expected, due to frequently preening, standing alert and shaking its head ($\chi^2 = 21.667$, df = 8, $p = 0.006$). A mute swan also had a significantly different distribution of behaviours than expected, notably preening and pecking the AFI more frequently than other behaviours observed ($\chi^2 = 173.86$, df = 12, $p < 0.001$; Table 5.1).

Table 5.1 Chi square test of homogeneity of the ethograms undertaken at The Prince of Wales Dock on herring gull (*Larus argentatus*), lesser black-backed gull (*Larus fuscus*), great cormorant (*Phalacrocorax carbo*) and mute swan (*Cygnus olor*) when using the artificial floating islands, in order to assess significantly dominant behaviours of each individual. The length of each ethogram (in minutes), degrees of freedom (df), p value and the dominant behaviour of each bird are also provided.

Species	Length (minutes)	Chi (χ^2)	df	p value	Dominant behaviour
Herring gull	6	23.4	9	0.005	Pecking island and walking.
Herring gull	7	21.778	7	0.003	Pecking island.
Herring gull	23	20.636	9	0.014	Standing alert, walking and pecking island.
Herring gull	60	69.140	21	<0.001	Pecking island and calling.
Herring gull (juvenile)	3	7.268	5	0.201	Pecking island.
Lesser black-backed gull	2	5.429	7	0.607	Standing alert.
Lesser black-backed gull (juvenile)	26	87.902	6	<0.001	Pecking island and standing alert.
Great cormorant	20	21.667	8	0.006	Preening, head shaking and standing alert.
Mute swan	50	173.86	12	<0.001	Head raised alert, preening and pecking island.

5.4.2 Swansea Marina

From 23rd May 2018 – 2nd June 2019, 1429 birds of 10 species were recorded during 23 vantage point surveys conducted at Swansea Marina; 1072 of eight species were recorded on hard structures, 278 of seven species on the water and 79 of six species on the AFI. These included herring gulls, lesser black-backed gulls, black-headed gulls, mute swans, mallards, great cormorants, carrion crows, jackdaws, feral pigeons and a grey wagtail (*Motacilla cinerea*). There was a significantly higher density of birds on the AFI in comparison to the hard structures (Kruskal Wallis = 70.693, df = 2, p <0.001; Nemenyi's multiple comparison, AFI and hard structures, p < 0.001; Figure 5.6). This was due to the presence of mallard, black-headed gulls and juvenile *Larus* spp. Hard structures in Swansea Marina consisted of the pontoons, pilings, poles, buoys and the pavement.

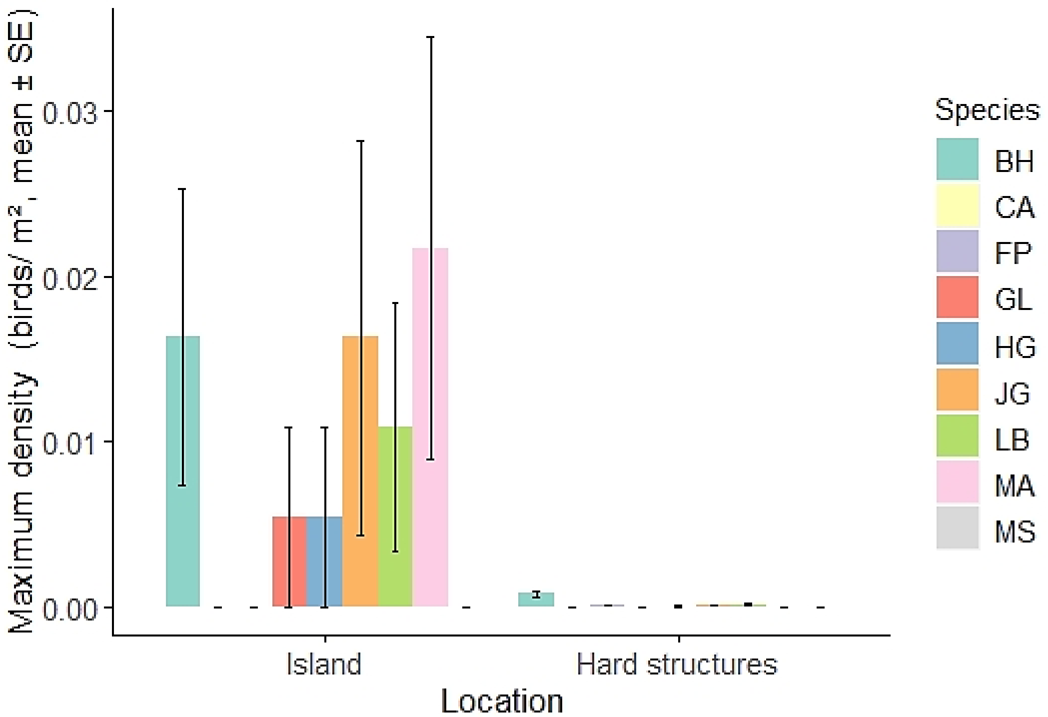


Figure 5.6: The maximum density of birds recorded per survey (birds/m², mean ± standard error) on the artificial floating islands in comparison to hard structures in Swansea Marina (n = 23). Each species is represented using the British Trust for Ornithology species coding system, excluding juvenile Larus spp. which is shown as JG: black-headed gull (BH), cormorant (CA), feral pigeon (FP), grey wagtail (GL), herring gull (HG), jackdaw (JD), lesser black-backed gull (LB), mallard (MA) and mute swan (MS). The graph excludes carrion crow, jackdaw and magpie as they were not present when maximum density values were recorded.

The highest density of birds recorded during the surveys was on the 10th October 2018 when two juvenile *Larus* spp. were using the AFI and on the 2nd June 2019 when two mallards were using the AFI (0.25 birds/ m²; Figure 5.7). The highest density of birds recorded on the hard structures was on the 22nd February 2019 associated with 35 black-headed gulls (0.004 birds/ m²). The highest density of birds recorded on the water was on the 15th February 2019 associated with 15 black-headed gulls (<0.001 birds/ m²; Figure 5.7). The AFI had the highest species diversity in comparison to the water habitat and hard structures (Shannon Wiener species diversity index, AFI = 1.599; water = 1.491; hard structures = 0.902; n = 23).

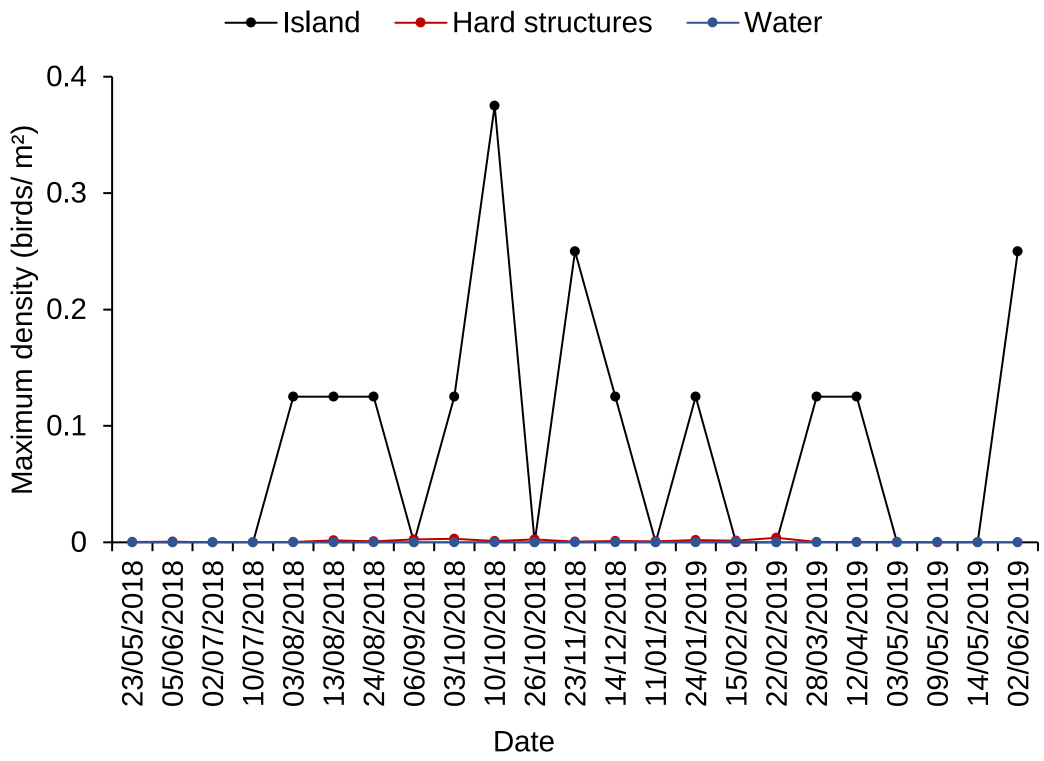


Figure 5.7: The maximum density of birds per survey recorded from the 23rd May 2018 – 2nd June 2019 in Swansea Marina on the artificial floating island, hard structures and water habitat (n = 23).

Each feature was used for preening, resting and foraging by birds in Swansea Marina, however, there was a significantly higher density of birds preening on the AFI and hard structures in comparison to the water habitat (Kruskal Wallis = 13.856, df = 2, $p < 0.001$; Nemenyi's multiple comparison, AFI and water, $p = 0.039$; hard structures and water, $p = 0.001$; n = 23; Figure 5.8). In addition, a significantly higher density of birds were resting on the AFI in comparison to the hard structures (Kruskal Wallis = 13.991, df = 2, $p < 0.001$; Nemenyi's multiple comparison, AFI and hard structures, $p = 0.038$; n = 23). There was no significant difference in the density of birds foraging on the AFI, hard structures and water habitat (Kruskal Wallis = 1.097, df = 2, $p = 0.578$; n = 23).

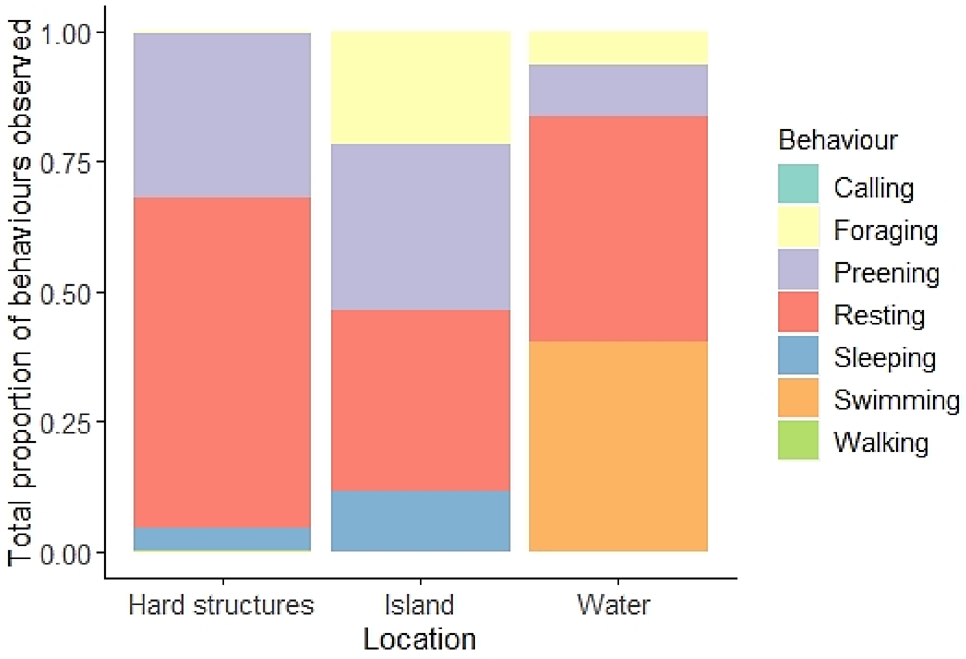


Figure 5.8: The total proportion of birds recorded as calling, foraging, preening, resting, sleeping, swimming and walking on the hard structures, artificial floating island and water habitat during the survey period from the 23rd May 2018 – 2nd June 2019 in Swansea Marina (n = 169).

In addition to vantage point surveys, ethograms were produced for one mallard that used the AFI for 60 minutes on the 28th March 2019 and 2nd June 2019. During the survey in March, the mallard was sitting down and resting with its head tucked under its wing and raising its head in an alert manor significantly more often than the 14 other behaviours observed (χ^2 = 39.60, df = 13, p <0.001). In June, the mallard's distribution of behaviours was not significantly different than expected. The individual exhibited 9 behaviours including sitting resting, sitting resting with head tucked under wing and head raised alert (χ^2 = 8.64, df = 8, p = 0.373; Appendix 4).

5.4.3 Habitat Complexity

The percentage cover of buildings in the sample area at Swansea Marina was significantly higher than at The Prince of Wales Dock, with an average cover of 22.92 ± 8.91 in Swansea Marina and 2.92 ± 2.92 in The Prince of Wales Dock (mean ± standard error; Kruskal Wallis = 4.965, df = 1, p = 0.026; Figure 5.9).

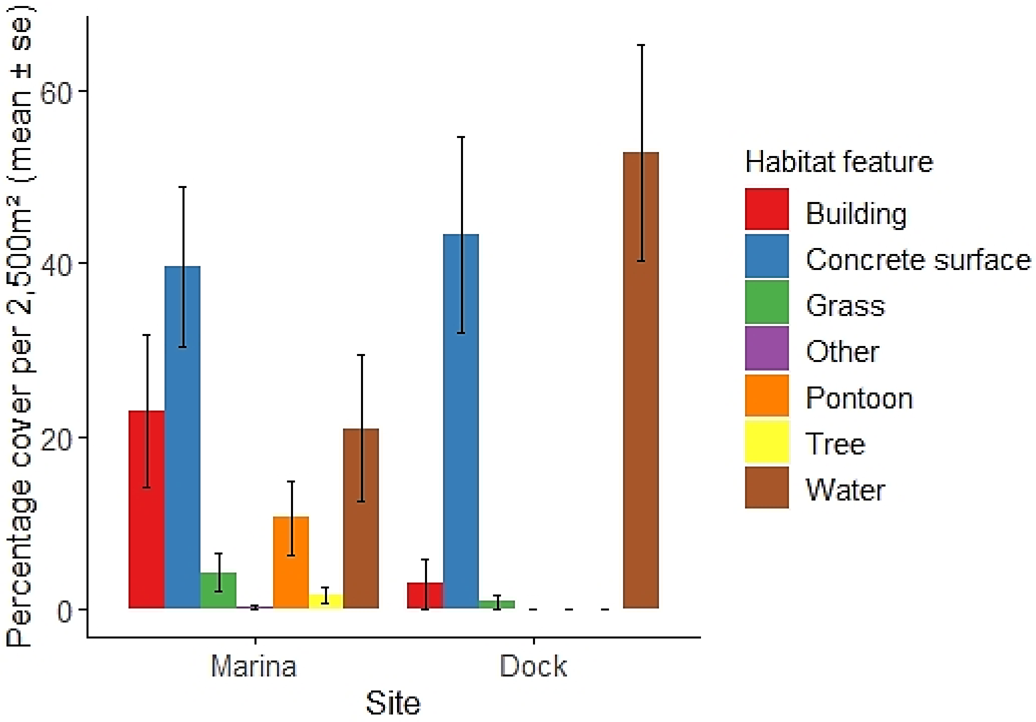


Figure 5.9: The percentage cover (mean ± standard error) of buildings, concrete surfaces, pontoons, water, grass, trees and other non-categorised habitat features in randomly selected 2500 m² grids covering Swansea Marina and The Prince of Wales Dock (n = 12).

In addition, the percentage cover of pontoons in the sample area at Swansea Marina was significantly higher than The Prince of Wales Dock, with an average cover of 10.58 ± 4.33 in Swansea Marina and no cover recorded in The Prince of Wales Dock (mean ± standard error; Kruskal Wallis = 4.964, df = 1, p = 0.006; n = 12). This corresponded with a significantly higher density of birds on the AFIs in The Prince of Wales Dock in comparison to Swansea Marina when including both AFIs (Mann-Whitney U Test = 90.5, p = 0.02; n = 18). However, when comparing the density of birds on the 8 m² in The Prince of Wales Dock and Swansea Marina, bird density was not significantly different (Mann-Whitney U Test = 142.5, p = 0.491; n = 18) and therefore, it may be a function of AFI size rather than habitat complexity. There was also no significant difference in the density of birds using the hard structures in Swansea Marina in comparison to The Prince of Wales Dock (Mann-Whitney U Test = 115, p = 0.836, n = 18).

5.5 Discussion

At both The Prince of Wales Dock and Swansea Marina, bird density was significantly higher on the artificial floating islands (AFIs) in comparison to the hard structures (Figure 5.3 and 5.6). The species diversity of birds on the AFI in Swansea Marina was also significantly higher than the hard structures and water environment. Unlike the hard structures at both sites, the

AFI was planted with halophytes growing into the matrix unit, mimicking the aesthetic features of a natural wetland. The sward height remained under 30 cm for the duration of the installation. Based on previous coastal AFI projects and greater understanding of coastal bird ecology, sparsely vegetated habitats largely attract a high species richness of terns and gulls (Burgess & Hirons, 1992; Milsom *et al.*, 2000b; Wilson *et al.*, 2004; Shealer *et al.*, 2006). Greater sward height and cover by saltmarsh vegetation can attract large numbers of small waders such as the grey plover (*Pluvialis squatarola*) which used the upper saltmarsh for protection from anthropogenic disturbance in the Tagus estuary, Portugal (Rosa *et al.*, 2003). However, during comparison of the species richness and abundance of birds at a natural smooth cordgrass (*Spartina alterniflora*) saltmarsh and an artificially created saltmarsh, gulls and terns were the primary inhabitants at the artificial site and a greater diversity of waders and wildfowl used the natural site (Melvin & Webb, 1998). Similarly, in this study, high densities of large Larids were recorded on the AFIs in The Prince of Wales Dock resulting in the lowest species diversity in comparison with the hard structures and water habitat. This could be due to interspecific competition for resources provided by the AFI such as feeding opportunities on the fouling invertebrates and as a resting site.

In addition to the sward height of the AFIs influencing utilisation by birds, the AFIs were also installed in heavily modified coastal water bodies. Artificial habitats created inland tend to support fewer species (Burgess & Hirons, 1992), which could account for the low density of waders on the AFIs and low bird densities on the hard structures in both The Prince of Wales Dock and Swansea Marina. The location of the AFIs in heavily modified coastal water bodies exposed them to anthropogenic disturbance from passing pedestrians and recreational boating activity; an impact that can control bird community structure in wetland habitats (Malavasi *et al.*, 2009; Scarton & Montanari, 2015). In addition, there is a lack of exposed sediment limiting foraging opportunities for waders with specialist feeding strategies. Sediment type, exposure time based on the tidal cycle and presence of saltmarsh habitat have been highlighted as key factors influencing the spatial distribution of shorebirds (Yates, 1993; Rosa *et al.*, 2003; Kalejta & Hockey, 2008). Therefore, AFIs providing new patch habitat in heavily modified coastal water bodies could be used in the absence of natural high tide roost sites for resting and shelter.

Larus spp. in The Prince of Wales Dock predominantly foraged on the AFIs. Food availability for all taxa plays a key role in breeding success, spatial and temporal distribution, population stability, health and survival (Martin, 1987, 1995; Pons & Migot, 1995; Camphuysen & Gronert, 2012). Excluding urban populations, herring gulls typically forage on bivalves and crustaceans present in intertidal habitats, relatively close to the breeding colony (Camphuysen & Gronert, 2012; Washburn *et al.*, 2013; Enners *et al.*, 2018b). As omnivorous, opportunistic scavengers, coastal urbanised populations have adopted a dual foraging strategy, commuting frequently between marine and terrestrial environments depending on food availability (Ditchkoff *et al.*, 2006; Bartumeus *et al.*, 2010; Yoda *et al.*, 2012). Herring gulls for example have been observed commuting between intertidal habitats and terrestrial habitats based on the tidal height and exposure of soft sediment (Enners *et al.*, 2018b). On the AFI present in The Prince of Wales Dock, blue mussels (*Mytilus edulis*) were the most abundant fouling organism and formed a complex secondary reef on the underside of the AFI. The presence of empty blue mussel shells on the upper surface of the AFI confirmed that gull spp. were feeding on the blue mussels. Previous literature has also confirmed that herring gulls alongside oystercatchers and eiders (*Somateria*) are the main consumers of blue mussels, accounting for 42 % of a herring gulls diet during summer periods in Spiekeroog, off the coast of Germany (Hilgerloh *et al.*, 1997; Spaans, 2002). Therefore, the presence of blue mussels as a food source could have been the key factor attracting herring gulls to the AFI.

In contrast, great cormorants and mute swans largely used the AFI for preening while remaining alert of their surroundings (Table 5.1). These results demonstrated a difference in habitat utilisation due to varying species ecology. For piscivorous diving species such as the great cormorant (Kirby *et al.*, 1996), the AFIs do not provide additional feeding opportunities, although they may attract fish close to the surface. Great cormorants exclusively forage during daylight hours on locally abundant fish and are known to adapt their foraging strategy based on season and location (Kirby *et al.*, 1996; Randall *et al.*, 2002). During this study, great cormorants were observed foraging in the dock and using the AFI for resting, preening and drying of their wings. Mute swans are one of the largest omnivorous wildfowl species (Guillaume *et al.*, 2014) that have gained attention due to their potential negative impact on the abundance of vegetation in wetland habitats (Gayet *et al.*, 2011; Wood *et al.*, 2012, 2013; Guillaume *et al.*, 2014). On the upper elevation of the AFI, mute swans can damage and destroy vegetation via feeding, trampling, faecal deposition, transporting seeds to the site and causing pH fluctuations in the substratum (Wood *et al.*, 2012, 2013). Mute swans in The Prince of Wales Dock were observed pecking the vegetation and the coir used as substratum on the AFI potentially collecting it as nesting material. The visits were sporadic throughout the year, suggesting that the adult pair were territorial individuals unlike non-territorial individuals that have been observed in water bodies only during the spring and summer months (Holm, 2002; Gayet *et al.*, 2011; Wood *et al.*, 2013). Jackdaws were also observed collecting the coir matting for nesting material at The Prince of Wales Dock.

In Swansea Marina there were high densities of mallard and black-headed gulls on the AFI. Mallards are omnivores and referred to as ecological generalists (Sauter *et al.*, 2012). They are the most numerous dabbling duck, able to adapt to a wide variety of habitats across the northern hemisphere (Bengtsson *et al.*, 2014). In the UK, mallards have both resident breeding and migratory populations and use a diversity of water habitats (Sauter *et al.*, 2012). During ethograms conducted on two mallards using the AFI, they were predominantly resting. Mallards on Öland island in the southern Baltic sea tend to rest during daylight hours and forage at night in flooded, wetland locations (Bengtsson *et al.*, 2014). When birds are migrating in particular, stop-over sites to forage and rest at high tide roosts is vitally important and becoming less abundant due to anthropogenic developments causing habitat fragmentation (Melles *et al.*, 2003; Stein *et al.*, 2014; Bengtsson *et al.*, 2014). In this case, the male mallard is resident to Swansea Marina and is often fed by recreational boat users within the facility.

Black-headed gulls are omnivores and scavengers that use both intertidal and terrestrial habitats to forage, with populations regularly observed in urbanised areas (Kubetzki & Garthe, 2003). More specifically, black-headed gulls have frequently been associated with flat blocks and green spaces, with the latter acting as an ecological corridor (Fernández-Juricic & Jokimäki, 2001; Maciusik *et al.*, 2010). In Swansea Marina, the percentage cover of buildings in a 122500 m^2 area was significantly higher than The Prince of Wales Dock (Figure 5.9). The combination of more buildings, pontoons and buoys in Swansea Marina could be the key factor attracting more black-headed gulls, that may have used the hard structures as an energy saving mechanism, roosting site or shelter from onshore winds (Maciusik *et al.*, 2010; Shepard *et al.*, 2013). Additionally, 78.6 % of the adult *Larus* spp. recorded in Swansea Marina were lesser black-backed gulls; in The Prince of Wales Dock adult lesser black-backed gulls only accounted for 6.69 % of the total *Larus* spp. recorded. On refuse sites lesser black-backed gulls have shown avoidance behaviour when herring gulls approached the same food source. Adult herring gulls are larger than lesser black-backed gulls, with a wing span ranging from 123 – 148 cm in comparison to 117 – 134 cm for lesser black-backed gulls and also display aggressive behaviour to protect their prey (Verbeek, 1977; Svensson *et al.*, 2011). This may explain the differences in the proportion of herring gulls and lesser black-backed gulls across the two survey sites.

There was a significant difference in bird density when comparing the total area of AFIs in The Prince of Wales Dock to the 8 m^2 AFI in Swansea Marina and no significant difference when comparing bird density of the same sized AFIs. Therefore, the larger the AFI installed the greater the bird density with no influence of habitat complexity apparent from bird density data collected during this study. Previously, the relationship between island size and number of breeding birds and nests has been varied (Eason *et al.*, 2012). For example, in south-eastern Alberta smaller islands further offshore with greater vegetative cover had a higher density of wildfowl nests than larger islands closer inshore (Giroux, 1981). In contrast, larger islands supported higher density of yellow legged gull nests off the French Mediterranean coast

(Duhem *et al.*, 2007). These contradicting relationships are a result of numerous factors including species-specific requirements, predation risk and degree of isolation and disturbance (Eason *et al.*, 2012).

5.6 Conclusion

Bird density was higher on the vegetated AFIs in comparison to local hard structures, therefore, the hypothesis of this study was accepted. In The Prince of Wales Dock, the higher salinity facilitated the fouling of blue mussels on the underside of the AFI creating biogenic reefs that provided feeding opportunities for *Larus* spp. The coir matting and vegetation also provided resting and preening sites suitable for mallard, mute swan, great cormorant and black-headed gulls. The size of the AFI was highlighted as a key factor influencing bird density, rather than localised habitat complexity. The study highlighted the importance of understanding the ecology of target species during the planning stages of a habitat creation project, which will aid decision making processes on the appropriate size, location and vegetation cover required.

Chapter 6: Public perception of coastal habitat loss and habitat creation using artificial floating islands in the United Kingdom

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Abstract

Ecological engineering and the installation of green infrastructure such as artificial floating islands (AFIs), are novel techniques used to support biodiversity. Research specifically on AFIs in marine environments has largely focused on their ecological functioning role and engineering outcomes, with little consideration for the social benefits or concerns. The aim of this study was to gain an understanding of public perception of coastal habitat loss in the United Kingdom (UK) and AFIs as a method of habitat creation in coastal environments. The testable hypothesis was that the majority of the respondents will be aware of the ecological functioning role of AFIs and would support their installation in coastal environments. This was achieved via a survey, consisting of six closed and two open questions. Of the 200 respondents, 94.5 % were concerned about the loss of coastal habitats in the UK, but less than a third were aware of habitat restoration or creation projects in their area of residence. There was a positive correlation between proximity of residency to the coast and knowledge of habitat restoration or creation projects. The majority of the respondents understood the ecological functioning role of AFIs and 62 % would preferably want successful plant growth and birds using the AFI. 90.9 % of the respondents supported the installation of AFIs and therefore, the hypothesis was accepted. Nearly a third of the respondents had concerns about AFI installations, such as the degradation of the plastic matrix, long term maintenance and disturbance of native species which must be addressed during the planning stages of any habitat creation project.

6.1 Introduction

By 2025, more than 75 % of the human population is estimated to live within 100 km of the coast (Bulleri & Chapman, 2010; Sekovski *et al.*, 2012; Mercader *et al.*, 2017b). Currently, 14 of largest cities occupy coastal regions (Sekovski *et al.*, 2012), associated with extensive infrastructure to support commercial, residential and recreational developments (Chapman & Underwood, 2011; Firth *et al.*, 2013, 2016; Evans *et al.*, 2019). Due to the risk of flooding and erosion caused by rising sea levels and severe storms, densely populated areas require protection via coastal defences such as sea walls, groynes and revetments (Bader *et al.*, 2011; Neumann *et al.*, 2015; Firth *et al.*, 2016; Mercader *et al.*, 2017a). The combined impact of coastal 'armouring' and marine urban sprawl has caused increasing spatial disconnection of coastal habitats, habitat degradation and alterations to natural community assemblages (Bulleri & Chapman, 2004, 2010; Chapman & Blockley, 2009; Bishop *et al.*, 2017). Coastal wetlands for example, are considered one of the most threatened ecosystems, with up to 50 % of global saltmarsh recorded as either lost or degraded (Lotze *et al.*, 2006; Worm *et al.*, 2006; Halpern & Walbridge, 2008; Barbier *et al.*, 2011). Birds are reliant on coastal habitats for nesting, foraging and roosting and are increasingly under threat, due to rising sea levels and proposed coastal infrastructure (Chu-Agor *et al.*, 2011). Fish larvae dispersal and recruitment can also be disrupted by coastal infrastructure, as their construction causes fluctuations in current patterns and sediment loading (Roberts, 1997; Bouchoucha *et al.*, 2016). Therefore, the United Kingdom (UK) Post–2010 Biodiversity Framework intends to prevent any further loss of biodiversity and ecosystem services, utilising biodiversity enhancement methods where appropriate (JNCC & Defra, 2019).

Ecological engineering (eco-engineering) refers to the modification of planned or existing structures integrating ecological theory into structural design to influence physcio-chemical processes (Type A), or direct engineering of biota via replanting or restocking (Type B) (Elliott *et al.*, 2016; Morris *et al.*, 2016; Dafforn, 2017). In heavily modified marine ecosystems such as marinas and docks, eco-engineering offers a means of enhancing existing or planned structures to benefit local biodiversity, while maintaining the integral anthropogenic function of the structure (Martins *et al.*, 2010; Browne & Chapman, 2011; Naylor *et al.*, 2012a). AFIs, also referred to as floating treatment wetlands, Biohavens® and floating ecosystem modules, offer an alternative eco-engineering method (Connell, 2000, 2001). In the UK, they are commercially sold by companies that provide eco-engineering solutions for silt management, plastic pollution, wastewater treatment and habitat creation. They broadly consist of a buoyant mat, planting media and emergent vegetation (Yeh *et al.*, 2015; Frog Environmental, 2016b; Chen *et al.*, 2016; Pavlineri *et al.*, 2017). The design referred to in this study (Figure 6.1, *top left*), consists of a non-woven recycled plastic matrix, an integrated connection grid providing structure and closed cell polyurethane foam for buoyancy (Burzaco & Frog Environmental, 2016; Frog Environmental, 2019). With established plants grown on coir matting, AFIs support a localized ecological community in the submerged roots and on the surface of the structure itself; these include algal communities, macroinvertebrates and epibiotic species (Kato *et al.*, 2009; Yeh *et al.*, 2015). The deployment of an AFI seaward of mean high water springs in Wales requires the issue of a marine licence under the Marine and Coastal Access Act, 2009 by the Licensing Authority, Natural Resources Wales.

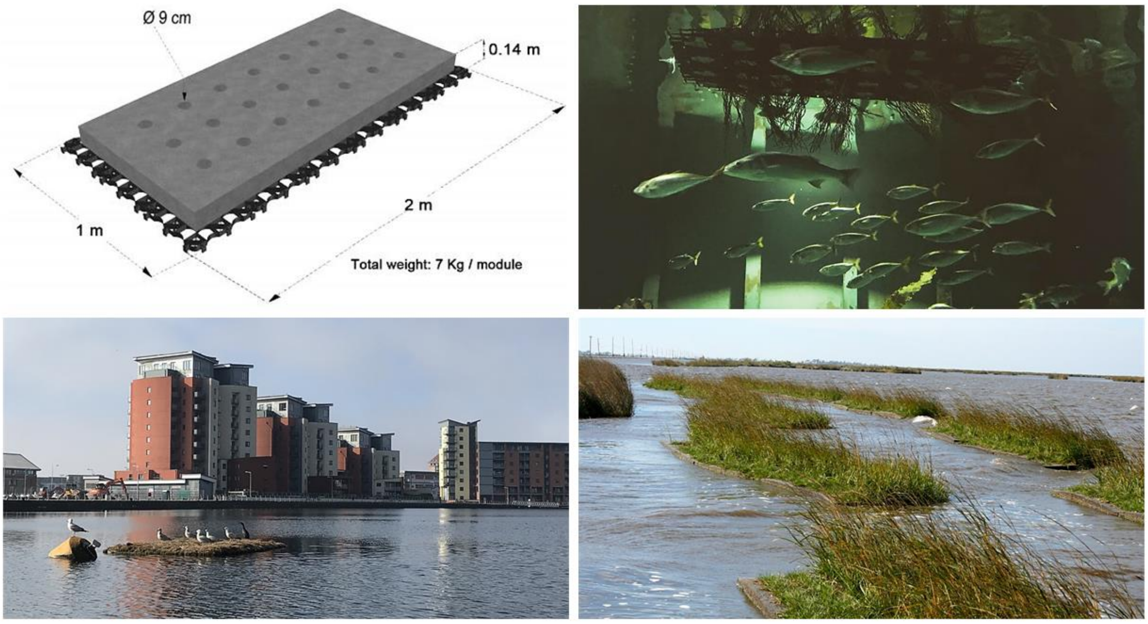


Figure 6.1: Artificial floating island (AFI) unit and existing installations and research. *Top left* – Schematic diagram of a 2m² matrix unit, commercially sold as Biohavens®. These AFIs consist of a non-woven plastic matrix, integrated connection grid and polyurethane foam (Burzaco & Frog Environmental, 2016); *top right* – AFI installed in a controlled experiment at Bristol Aquarium, with 13 native fish; *bottom left* – AFI installed in a saline dock in Swansea known as Prince of Wales Dock; and *bottom right* – Linear arrangement of AFIs used on the coast of Louisiana, USA, for wave absorption and to reduce coastal erosion (Frog Environmental, 2016a).

Over 300 AFIs have been deployed by the Royal Society for the Protection of Birds (RSPB) to provide breeding grounds and roosting sites for divers, gulls, terns, waders and wildfowl species, within coastal wetlands in the UK (Burgess & Hirons, 1992). Floating structures also promote the formation of biofouling communities (Connell, 2000; Perkol-Finkel *et al.*, 2006a; Nall *et al.*, 2017), increasing productivity and nutrient availability via deposition of organic matter into the local environment. This can attract higher trophic species such as fish, elevating the local species diversity (Pardue, 1973; Perkol-Finkel *et al.*, 2006a; Neal & Lloyd, 2018). However, there currently is a lack of understanding of the public perception of AFIs, which could impact on the success of future installation projects (Morris *et al.*, 2016; Evans *et al.*, 2017; Kienker *et al.*, 2018; Strain *et al.*, 2019).

Public awareness and perception of both national and international scale environmental concerns is important, as it influences acceptance of environmental policies and positive behavioural change in society (von Borgstede *et al.*, 2013; Shi *et al.*, 2015). Understanding the relationship the public currently have with marine ecosystems will enable the identification of any misconceptions of environmental issues and highlight the issues of concern (Gelcich *et al.*, 2014). With a better understanding of successful and failed processes of scientific communication, future environmental management and policy strategies can be improved, encouraging public support. Incorporating public awareness and citizen science campaigns into environmental conservation can positively contribute to the success of achieving new, conservation objectives (Horwich & Lyon, 2007; Jefferson *et al.*, 2015; Hawkins *et al.*, 2016). Previously, the importance of stakeholder engagement has been highlighted during the installation of artificial reefs off the west coast of Scotland and southern Portugal (Sayer & Wilding, 2002; Ramos *et al.*, 2007). In a number of studies, the majority of the respondents supported eco-engineering initiatives that enhanced the conservation of biodiversity (Morris *et al.*, 2016; Kienker *et al.*, 2018; Strain *et al.*, 2019). However, awareness and knowledge of eco-engineering initiatives tends to be lower in Europe compared to America and Australia (Strain *et al.*, 2019).

In the UK, public perception research has focused on the general marine environment and its protection from global concerns such as climate change (Fletcher *et al.*, 2009; Chilvers *et al.*, 2014; Jefferson *et al.*, 2014; Hawkins *et al.*, 2016), managed realignment (Myatt-bell *et al.*, 2002; Myatt *et al.*, 2003), beach aesthetic and selection (Tudor & Williams, 2006) and offshore wind farms (Haggett, 2008). It is important that similar information is gained on the public perception of eco-engineering methods, such as AFIs.

6.2 Aims and Objectives

The aim of this study was to gain an understanding of the perceived importance of coastal habitat loss in the UK, in comparison to other environmental issues. Further, the study aimed to obtain information on the public understanding of AFIs and any concerns related to AFI installations. The testable hypothesis was that the majority of the respondents will be aware of the ecological functioning role of AFIs and would support their installation in coastal environments. The four objectives of the survey were to assess whether the public were:

1) Concerned about the loss of coastal habitats in the UK.

2) Aware of local habitat restoration or creation projects.
3) Aware of the ecological functioning role of AFIs.
4) Supportive of AFI initiatives as a method of habitat creation within coastal environments.

The results of this study will help inform stakeholders planning on installing AFIs in coastal environments on public opinion and best practice before and during the AFI installation.

6.3 Materials and Methods

6.3.1 Survey Design

The survey consisted of eight questions, subdivided into two themes: coastal habitats and AFIs (Table 6.1). The survey included questions with 5-point Likert scale answers, binary and multiple choice. It was restricted to six closed questions and two open questions, with an average completion time of three minutes, thus maximising participation. No background information was provided prior to the respondent completing the survey. Question 1 was limited to five factors for simplicity and the factors selected were all environmental issues prevalent in the UK. In terms of personal information, only distance that the respondent lived from the coast was determined. Other demographic information was not collected in this survey, such as age and occupation, as these details were not required to meet the studies objectives. However, more detail about the location of residency was inferred from Question 3, addressing awareness of habitat restoration initiatives and assuming that participants had greater knowledge of projects in their local area. Question 5 addressed a common issue associated with high numbers of wildfowl and maintaining plant growth on AFIs. Additionally, AFIs can be specifically installed without vegetation to attract certain species that require only substratum for breeding (Burgess & Hirons, 1992; Hancock, 2000).

Table 6.1: The complete survey consisting of 8 questions.

Section 1: Coastal habitats

1. Which of the following factors do you think are negatively impacting on the health of coasts in the UK? Rank each factor by importance. Urbanisation/ Coastal Developments, Flooding, Invasive species, Plastic pollution and Habitat loss

Possible answers: Very important, Fairly important, Important, Slightly important or Not at all important.

2. Are you concerned about the loss of coastal habitats in the UK, such as beaches, coastal wetlands and saltmarsh?

Possible answers: Yes, No or Not sure.

3. Are you aware of any habitat restoration or creation projects in your area like artificial floating islands or wildflower planting? If yes, any further details of the type of project and in what location can be added here.

Possible answers: Yes or No.

Section 2: Artificial floating islands

Artificial floating islands consist of a recycled plastic matrix and growing medium, that plants are able to grow roots through. They are often installed in lakes and rivers.

4. What do you think artificial floating islands are installed for? Tick any answers that you think are correct.

Possible answers: Aesthetic, To create habitat and support biodiversity, To support boating activity, To improve water quality, To collect litter or Other.

5. On some occasions it is difficult to maintain both plant growth and bird use. Which of the following scenarios would you prefer if an island were installed in your local area?

Possible answers: Bird activity and no plants, Plants and fencing with roots growing through the island for fish, Plant growth but not fully covering the island and bird activity or Not sure.

6. Would you have any concerns about the installation of an artificial floating island?

Open question

7. Would you support future installations of artificial floating islands or other habitat creation projects along the coast?

Possible answers: Yes, No or Not sure.

8. How far from the coast do you live?

Possible answers: 1 mile, 5 miles, 10 miles or 20 miles +.

6.3.2 Survey Collections

The target demographic was members of the public living in the UK, aged 18 or above. One respondent living in the Netherlands completed the survey and was included in the analysis. The survey was self-administrated using the survey tool 'Survey Monkey' (*https://www.surveymonkey.com*) and it went live on the 27th January 2019. The survey was live for 68 days, until the 5th April 2019. The survey was circulated on social media platforms such as Facebook and Twitter and members of the public were approached in Bristol Aquarium and Swansea. The survey was also circulated via community forums such as such as 'Maritime Quarter Residents Association' and 'Uplands and Brynmill community forum', to gain information on the opinion of local residents, who may have observed the AFIs in Swansea. A total of 200 surveys were collected during the 68 days that the survey was live (online, n = 170; in person, n = 30). The information provided during the online surveys and in person was the same, minimising any bias results. Swansea University ethics committee approved research conducted in this study (SU-Ethics-Student-030719/1106).

6.3.3 Statistical Analysis

Descriptive statistics were used to summarise the results from each question of the survey. Chi squared test of independence was used to assess whether there was a relationship between the distance the respondent lived from the coast and their (1) concern of coastal habitat loss; (2) awareness of habitat restoration and creation projects; (3) awareness of AFIs and their ecological functioning role; and (4) concerns related to AFIs being installed. This analysis was used to test the hypothesis that the majority of the respondents will be aware of the ecological functioning role of AFIs and would support their installation in coastal environments. Comments that addressed concerns about AFI installations (Question 6; Table 1) were organised into categories appropriately. Statistical tests were completed using R 3.6.0 statistics software.

6.4 Results

Of the 200 respondents, 29.5 % (n = 59) lived within 1 mile of the coast, 23 % (n = 46) within 5 miles, 17.5 % (n = 35) within 10 miles and 30 % (n = 60) greater than 20 plus miles.

6.4.1 Coastal Habitats

Most respondents considered plastic pollution (77.8 %, n = 154) and habitat loss (70.9 %, n = 139) to be very important factors affecting the health of coasts in the UK (Figure 6.2). There was no significant relationship between perceived importance of coastal habitat loss and proximity of residence to the coast ($\chi^2 = 2.86$, d.f. = 3, p = 0.41, n = 200). Less than a third of the respondents considered flooding (28.4 %, n = 55) and invasive species (24.2 %, n = 47) to be very important factors affecting the health of coasts in the UK. Three of the factors were perceived as not important at all. These were invasive species (5 %, n = 9), flooding (4 %, n = 7) and urbanisation/coastal developments (1 %, n = 2).

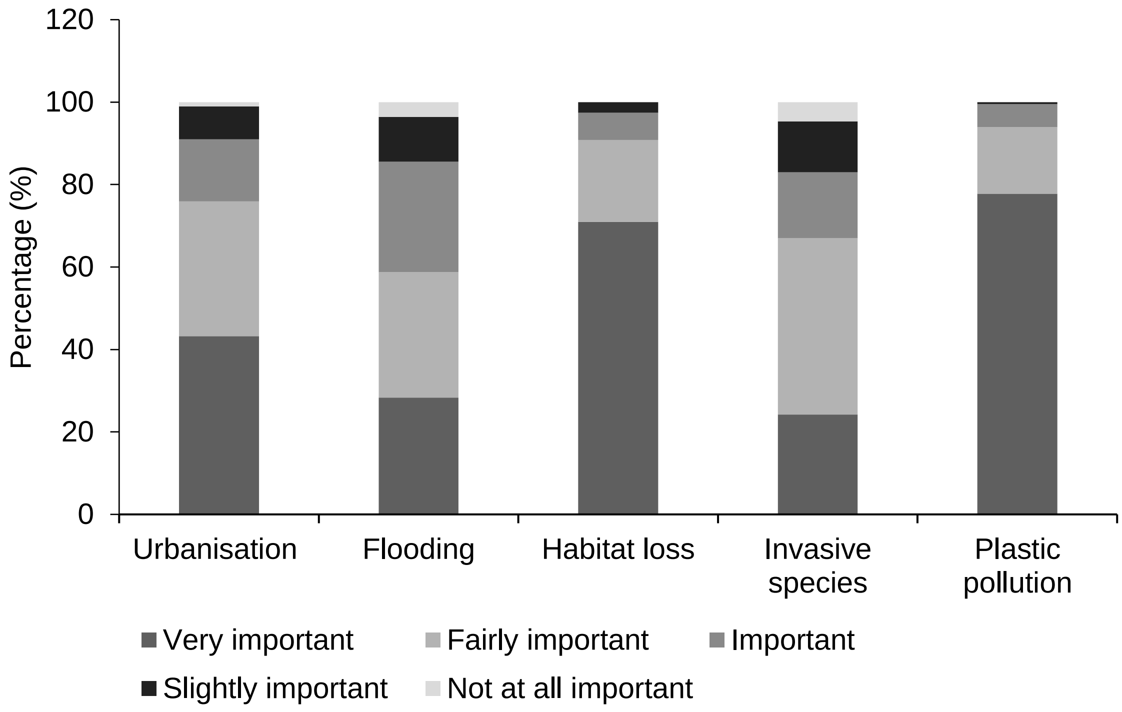


Figure 6.2: The perceived importance of factors negatively impacting on the health of UK coasts (n = 200).

Most respondents were concerned about the loss of coastal habitats in the UK (94.5 % n = 189). Under a third of the respondents (28.5 %, n = 57) were aware of habitat restoration or creation projects in their area of residence and this was dominated by respondents living within 1 – 5 miles of the coast (70 %). There was a significant relationship between the respondents' awareness of habitat restoration and creation projects and the proximity of residence from the coast ($\chi^2 = 8.95$, d.f. = 3, p = 0.02, n = 200). The respondents that provided further detail to Question 3 (n = 34) mentioned projects located in south Wales and England (Figure 6.3) and 52 % of the schemes were related to marine environments, rather than terrestrial or freshwater habitats.

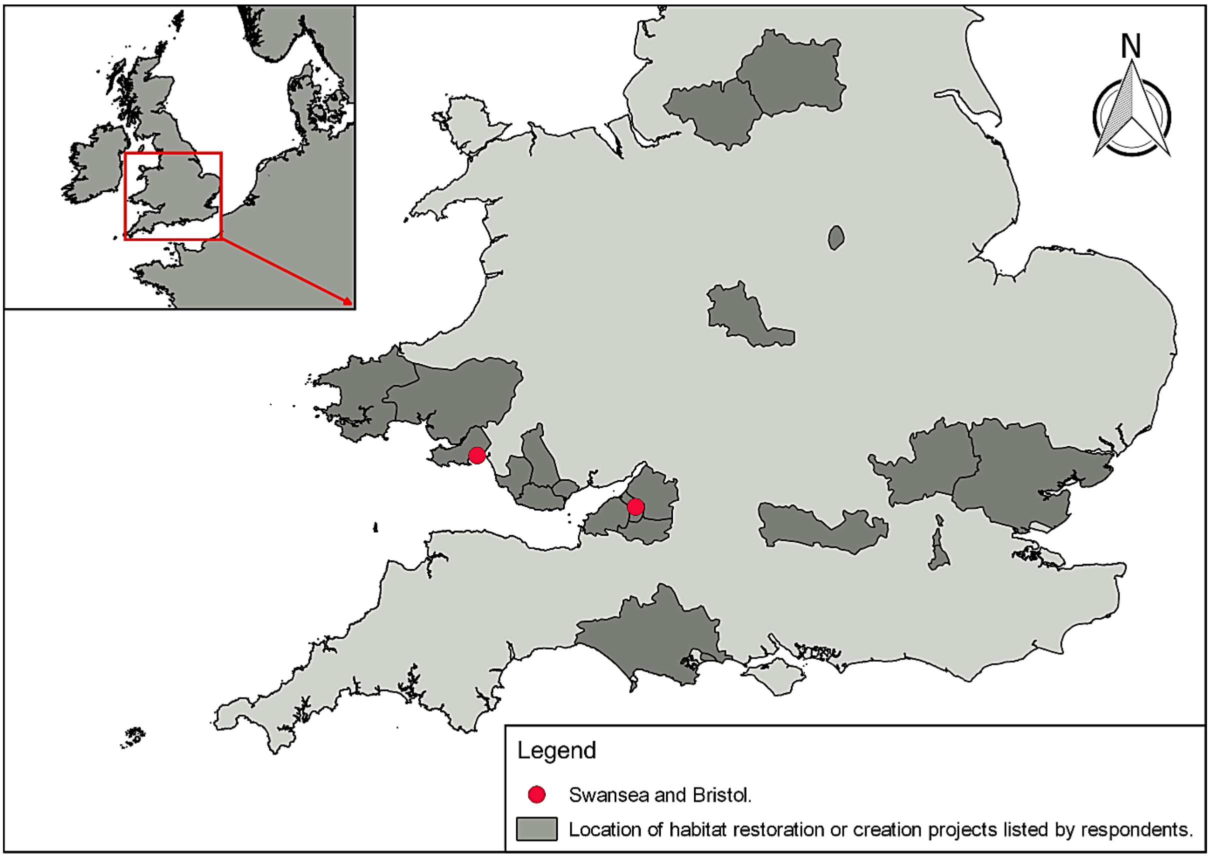


Figure 6.3: The location of habitat restoration or creation projects listed by the respondents of the survey (n = 34). The projects mentioned by respondents were located in 23 counties in England and Wales. Each project is represented by county it was located in (UK Postcode, 2012; European Environment Agency, 2019).

6.4.2 Artificial Floating Islands

As the respondents could give multiple answers on the perceived purpose of installing an AFI (Table 6.1, Question 4), there were 385 responses; 306 understood the ecological functioning role of AFIs ('to create habitat and support biodiversity' n = 196; 'to improve water quality' n = 110). There was no significant relationship in public awareness of the ecological functioning role of AFIs between the four proximity categories ($\chi^2 = 3.64$, d.f. = 3, $p = 0.30$, n = 200).

The majority of the public surveyed preferred to have both successful plant growth and birds using an AFI (62 %, n = 125, Figure 6.4). One third of the respondents preferred the installation of an AFI with successful plant growth, maintained by the inclusion of fencing (33 %, n = 67). High levels of bird activity with no plants growing was the least popular response (4 %, n = 9).

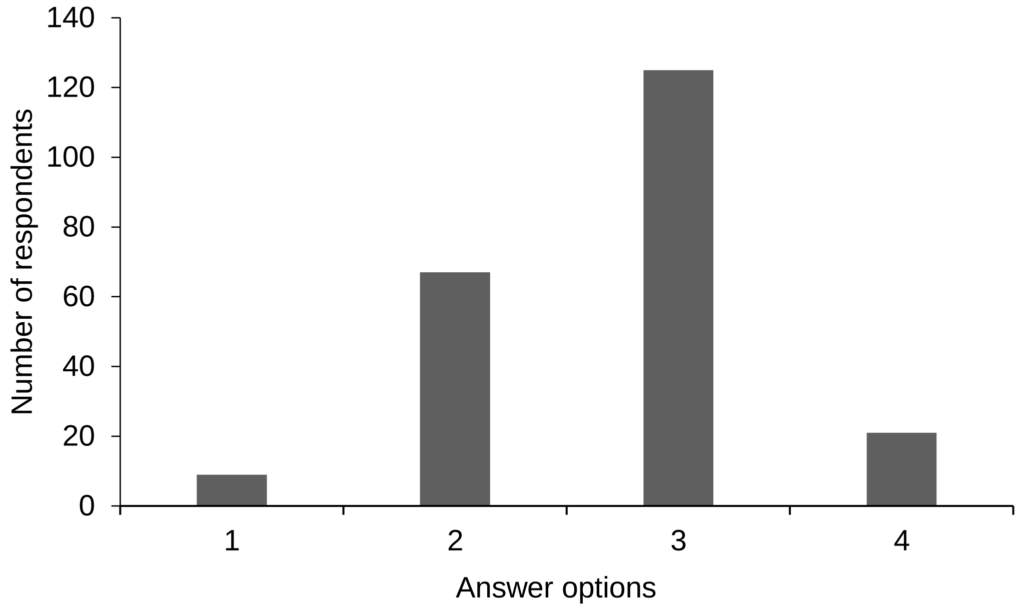


Figure 6.4: The respondents' preference of an installed artificial floating island in their local area based on five scenarios (n = 200). (1) Bird activity and no plants; (2) Plants and fencing, with roots growing through the island for fish; (3) Plant growth, but not fully covering the island and bird activity; and (4) Not sure.

Question 6 of the survey allowed the respondents to voice any concerns regarding AFI installations on the coast; 33 % of the 200 (n = 66) chose to comment on their concerns. These were broadly categorised into maintenance, recreation, aesthetic, plastic pollution, disturbance and invasive species concerns (Figure 6.5). The definition of each term based on the respondents' answers are outlined in Table 6.2.

Table 6.2: Definition of the six concerns listed by respondents in Question 6 of the survey.

Concern	Definition
Maintenance	Damage or detachment of the island during severe weather or as a result of vandalism.
Recreation	Disrupt boating, kayaking or surfing activity on the coast.
Aesthetic	It is unnatural and a potential eyesore.
Plastic pollution	Degradation of the plastic matrix into the water body.
Disturbance	Noise pollution during installation and impact on natural processes.
Invasive species	Encourage the presence or spread of a non-indigenous species that could cause damage to the ecosystem.

Plastic pollution (n = 33) and the long-term maintenance (n = 26) of an installed AFI were the key areas of concern by the respondents of the survey (Figure 6.5). The majority of the respondents would support the future installation of AFIs along the coast (90.9 %, n = 181), with the remaining respondents either unsure or against the method of habitat creation.

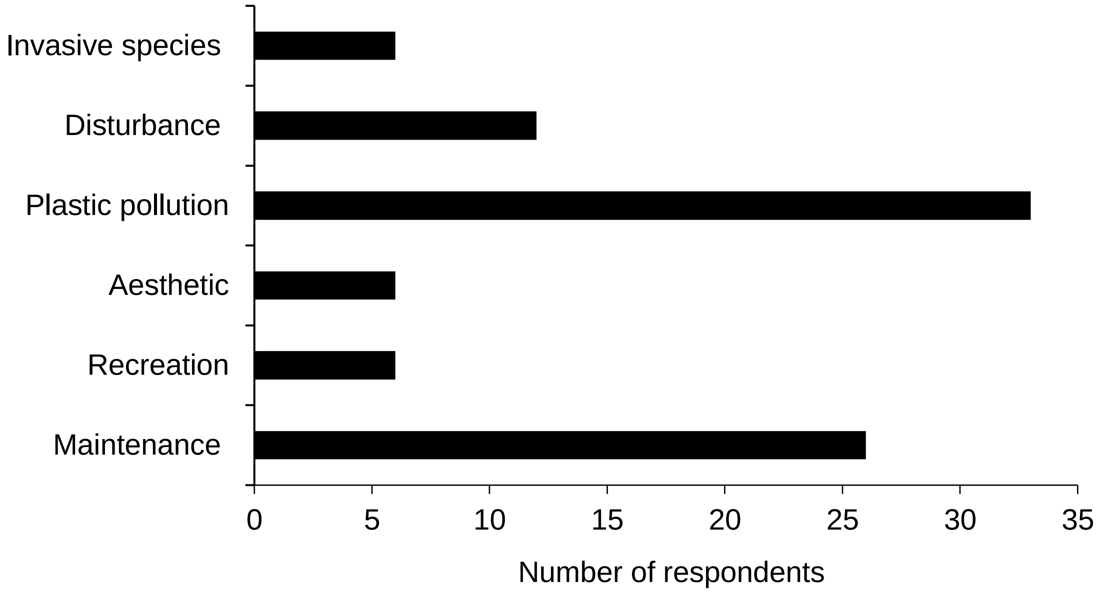


Figure 6.5: The number of concerns raised by respondents. These have been categorised into maintenance, recreation, aesthetic, plastic pollution, disturbance and invasive species (n = 200).

6.5 Discussion

6.5.1 Coastal Habitats

Artificial structures are proliferating in marine environments in the form of coastal defences (Nicholls & Cazenave, 2010; Bader *et al.*, 2011; Neumann *et al.*, 2015) and infrastructure to support shipping, transport, commercial, recreational and residential developments (Chapman & Underwood, 2011; Firth *et al.*, 2013, 2016; Evans *et al.*, 2019). Current legislation including the UK Post-2010 Biodiversity Framework address that novel techniques such as eco-engineering have a role to play to prevent any further loss of biodiversity and ecosystem services caused by anthropogenic activities (European Commission, 2011; Naylor *et al.*, 2012b; Strain *et al.*, 2018b; JNCC & Defra, 2019). Alongside meeting legislative targets, it is also important to engage with the public on environmental issues and conservation approaches that could be introduced. Without public engagement, the awareness and public support of future projects cannot be guaranteed. This study aimed to gain an understanding of the public perception of coastal habitat loss and AFIs as a habitat creation method.

The majority of participants of this survey were concerned about the loss of coastal habitats in the UK and consider plastic pollution and habitat loss as very important factors negatively impacting on the coast (Figure 6.2). Due to the release of documentaries such as ‘A Plastic Ocean’ in 2016 and ‘Blue Planet II’ in 2017, public awareness has increased substantially on the impacts of litter and specifically, non-biodegradable material in ocean ecosystems. The UK public also demonstrated an understanding of the deterioration of marine environments in when 95.8% of respondents to a survey considered marine habitats to be of ‘fair to poor’ health (Jefferson *et al.*, 2014; Hawkins *et al.*, 2016). Pollution and climate change are consistently mentioned as the most concerning environmental issues for members of the public, in the UK and abroad (Fletcher *et al.*, 2009; Morris *et al.*, 2016; Ruiz-Orejon *et al.*, 2016). In this survey, coastal urbanisation, flooding and invasive species were perceived as less important factors (Figure 6.2). This could be due to a lack of understanding of secondary impacts of developments, such as light and noise pollution and fluctuating hydrodynamics that can result in flooding. The importance of flooding to the respondent can also be governed by personal experience (Drosou *et al.*, 2019). The individuals socio-economic status linked to education and occupation and their specific motivations and interests, have also been identified as factors that drive awareness of environmental issues (Steel *et al.*, 2005; Fletcher *et al.*, 2009). These details were not included as part of this survey, as the information was not required to meet the studies research objectives. However, this does limit comparisons to other public surveys.

The majority of the public desire greater protection and conservation of the UK marine environment, from fishing and other damaging, exploitative practices (Hawkins *et al.*, 2016). However, as part of this survey under a third of the respondents were aware of habitat restoration or creation projects in their area. The respondents that did mention restoration and/or creation projects mostly lived within 1 – 5 miles of the coast and 52 % of the schemes were related to marine environments, rather than terrestrial or freshwater habitats. Examples of schemes mentioned across all habitat types included: dune slack management in Kenfig National Nature Reserve, Bridgend, to promote early succession of orchids; creating habitats for common kingfisher (*Alcedo atthis*) populations in the Lee Valley, Essex, via river management and; habitat restoration at Saltwells Local Nature Reserve, Dudley (Figure 6.3). The focus on marine conservation and policy could be a direct result of greater national awareness, personal interest based on residential location or occupation. The correlation between proximity to the coast, marine conservation and policy knowledge was discovered during a large-scale survey in the United States (Steel *et al.*, 2005; Fletcher *et al.*, 2009). However, this outcome could also be a result of the marine focus of the survey. To reduce potential bias towards marine projects, wildflower planting was also mentioned as a terrestrial habitat restoration and/or creation method in Question 3. The respondent was also asked to mention projects within their local area (Table 6.1).

For future research, more detailed demographic information would be desirable to gain deeper insight into relationships between social and economic background with views on marine conservation awareness and AFIs.

6.5.2 Artificial Floating Islands

In this survey, the majority of respondents showed an understanding of the ecological functioning role of AFIs. This could be linked to a positive shift in perception in the UK of the importance of wetland biodiversity and support towards wetland restoration (Rispoli & Hambler, 1999). Overall, the survey confirmed that the public preferred a vegetated AFI used by birds (Fig 6.4). In urban environments, green landscapes play a significant role in health and mediating the stresses of daily life (van den Berg *et al.*, 2010; Zhang *et al.*, 2013). This could have contributed to the respondents positive association with vegetation growth on the AFIs. Water quality of natural wetlands, the presence of emergent vegetation and trees and habitat value to local wildlife, were factors viewed as important in assessing wetland health in Australia (Dobbie & Green, 2013). There is however, evidence that a lack of understanding of ecological values is linked to a negative view of wetlands (Nassauer, 2004; Gobster *et al.*, 2007; Dobbie & Green, 2013).

Public and stakeholder perception studies of artificially created habitats have largely focused on benthic habitats including artificial reefs, concrete flowerpots used in the intertidal zone and coastal defence structures (Gray *et al.*, 2017; Kienker *et al.*, 2018; Strain *et al.*, 2019); therefore, limiting comparisons of the results from this study. Stakeholders including engineering and ecological consultants, academics and statutory bodies unanimously supported the installation of multi-functional artificial structures, which prioritised ecological benefits within coastal environments (Evans *et al.*, 2017).'Education and outreach' was one of the lowest assigned considerations by stakeholders, while a greater evidence base of the ecological benefits was seen as desirable. This illustrated the importance of accessible research and a strong evidence base for stakeholders (Evans *et al.*, 2019). It also demonstrated the lack of importance placed on public engagement by stakeholders, which could be limiting future public support of eco-engineering and artificial habitat creation projects.

Nearly a third of the respondents had concerns about the installation of AFIs in the marine environment. These concerns largely focused on the future degradation of the AFI matrix and potential for the islands to become plastic pollution (Figure 6.5). Additionally, the public were concerned about the long term maintenance and aesthetic features of the island; 'would it look unnatural and therefore un-aesthetic?', 'how will they be maintained?', and 'would the plastic in the matrix enter the food chain?'. Other comments were related to the potential disturbance of commercial and recreational boating, surfers and native wildlife. During the planning stages of an AFI installation, it is important that research and monitoring is undertaken by the individual or company responsible, on the environmental conditions of a proposed AFI location, in order to determine the size required to achieve ecological benefits and to assess the degree of exposure. The former includes abiotic parameters such as nutrient concentrations, pollutant levels and biotic variables such as species presence. The latter includes wind speed, water velocity, tidal height (if applicable) and salinity as certain metals are susceptible to corrosion. This information will aid decisions on the appropriate size, configuration and method of installation of an AFI, that minimises disruption of native fauna, ensures it is securely installed and does not become an eyesore. Research and open communication with potential stakeholders and members of the public, will also ensure that no recreational activities are disrupted by the installed AFI.

AFIs have an approximate life span of 20 years and this varies depending on its location (Frog Environmental, 2016b). As most AFIs are installed in ponds, reservoirs and rivers, case studies of islands exposed to waves, tides, marine biofouling and saline conditions are limited. This is due to the current design of AFIs commercially sold in the UK not being able to withstand the harsh conditions of exposed marine environments. An AFI designed with a stronger integrated connection grid would also cost more to produce and may not be a commercially viable option. Laboratory experiments have demonstrated that the size and configuration of the AFI determines the force (kilonewton, kn) exerted on the islands structure. Prior to the installation of an AFI, a maintenance and potential disposal plan should be established and made publicly accessible. This will ensure the long-term success of an AFI and reassure local residents that the island will be maintained and disposed of appropriately, to prevent potential degradation of the plastic matrix.

6.6 Conclusion

There was a positive correlation between proximity of residency to the coast and knowledge of habitat restoration or creation projects. The majority of the respondents were aware of the ecological functioning role of AFIs and would support their installation along the coast, therefore, the hypothesis was accepted. The successful establishment of plants and positive benefits to local wildlife, were equally important factors valued by respondents. There were concerns regarding the longevity of an artificially created habitat, which must be rectified with thorough strategic planning and appropriate aims, based on the location of the proposed AFI installation. Further research is required on socio-economic factors that could be influencing public awareness of habitat loss and artificially created habitats within urban ecosystems.

Chapter 7: General Discussion

The broad aim of this thesis was to assess AFIs installed in heavily modified coastal water bodies to answer the overarching question 'Can artificial floating islands be used as a restoration tool in heavily modified coastal water bodies to increase their ecological potential?'. Based on the findings of this study, AFIs can be used to increase ecological potential, although site specific considerations must be made prior to installation including size and therefore, carrying capacity of the AFI, location and degree of isolation, disturbance, exposure and presence of biota. Figure 7.1 summarises the biota that interacted with the installed AFIs in Swansea Marina and The Prince of Wales Dock.

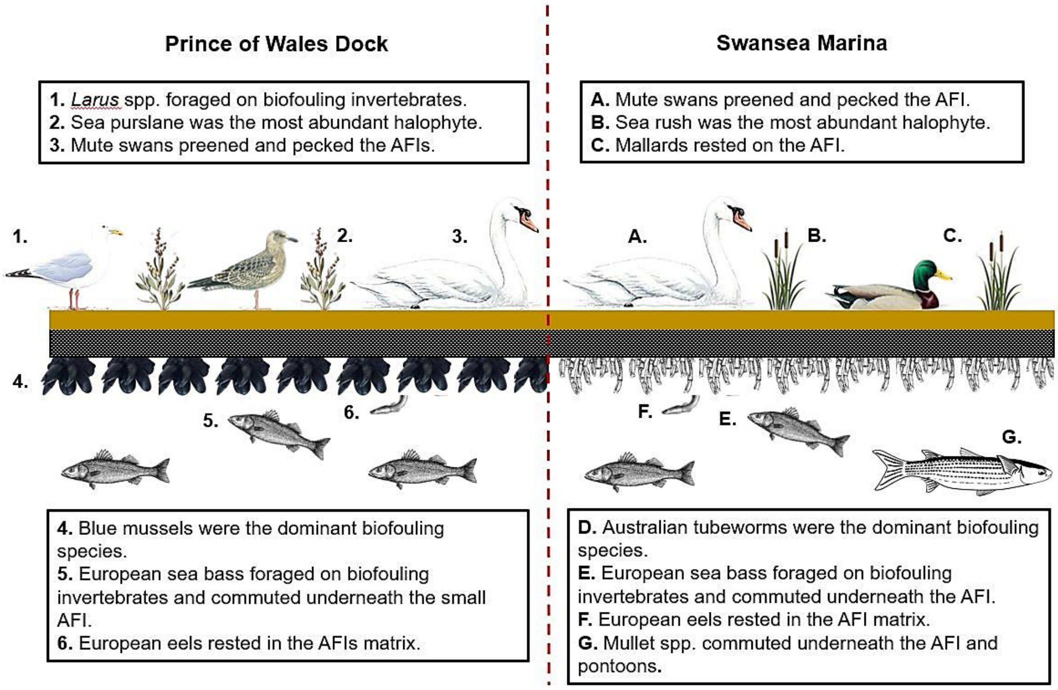


Figure 7.1: Schematic diagram summarising the terrestrial and aquatic biota that interacted with the artificial floating islands (AFIs) installed in Swansea Marina and The Prince of Wales Dock during their deployment and the halophytes that successfully grew until the AFIs were removed (Image references: Avramenko, 2000; RSPB, 2000a, 2000b, 2000c; Hulme, 2007; Ilbusca, 2011; European Commission, 2016; Extreme Environments, 2017; IFCA, 2020; RocknReef Inc, 2020).

The European Water Framework Directive (WFD; 2000/60/EC and as amended by Directives 2008/105/EC, 2013/39/EU and 2014/101/EU) is a legislative framework established to protect inland, transitional and coastal waters, groundwater and improve heavily modified water bodies, with the aim of achieving good ecological status or good ecological potential (Borja & Elliott, 2007; Temino-Boes et al., 2018). The directive assesses a combination of abiotic and biotic factors to classify a water bodies overall status including benthic communities, fish, hydromorphological and physcio-chemical characteristics (Borja & Elliott, 2007). Both Swansea Marina and The Prince of Wales Dock are of moderate ecological status and therefore, their ecological potential must be improved in order to meet the WFD objectives and to protect adjacent natural habitats (Temino-Boes *et al.*, 2018). Ecological-engineering (eco-engineering) methods such as AFIs should be considered as restoration tools in heavily modified water bodies, with the aim of reaching the maximum ecological potential of the site.

7.1 Recommendations

7.1.1 Research

The size of the AFIs installed in this study were experimental and determined based on the requirements to test the hypotheses and due to feasibility and available funding. Based on the island biogeography theory, the size of an island is a key factor impacting on species composition, known as the species-area relationships (Losos & Ricklefs, 2009) and the carrying capacity of plants, benthic invertebrates, fish and birds depending on the individual species ecology (Eason *et al.*, 2012). The concept of size and carrying capacity was supported in Chapter 5, as bird density was significantly higher in The Prince of Wales Dock when comparing the total area of the two AFIs (21.2 m^2) in comparison to the single AFI (8 m^2) in Swansea Marina. In contrast, the large AFI did not attract a higher relative abundance of fish in comparison to the small AFI in The Prince of Wales Dock. Differences in deployment time of the 8 m^2 and 13.2 m^2 AFIs and limited data collection may have contributed to the lack of fish recorded under the large AFI during its deployment. Therefore, research is required on small and large AFIs in the same water body with more replicates, in order to determine if size has an impact on species composition and carrying capacity. The lack of AFI replicates at each survey site was a key limitation in this study. In addition, research comparing one large AFI to multiple small AFI installations in the same water body would also contribute to the current understanding of island biogeography theory (Higgs, 1981) and allow comparison of the ecological benefits of both scenarios for future reserve design.

The size of a vegetated AFI and plant composition also impact on the retention of pollutant loads and water quality improvement within a system (Carleton *et al.*, 2001), as plants differ in nutrient assimilation capacity (Klomjek & Nitisoravut, 2005). In natural systems approximately 0.1 – 1 % of the watershed should be converted to wetland in order to detect tangible water quality improvement (Ham *et al.*, 2010). Based on the results of this study, future research should investigate replicate AFIs of different sizes transplanted with a total cover of sea purslane (*Halimione portulacoides*) and common glasswort (*Salicornia* *europaea*), to compare the successful growth of the halophytes, nutrient assimilation capacity and bird activity. In Lafri and Karatza Lagoons, north Greece yellow-legged gulls (*Larus michahellis*) showed preference for halophytic vegetation; notably sea purslane and glaucous glasswort (*Arthrocnemum fruticosum*) (Goutner, 1992). This could be due to the dense cover provided by the two halophytes reducing predation risk and exposure to harsh environmental conditions (Blokpoel *et al.*, 1978; Burger & Lesser, 1978; Becker & Erdelen, 1986). On Clarks Island, Massachusetts herring gulls (*Larus argentatus*) hatched more eggs and increased chick survival rates in nests sheltered by vegetation than unsheltered nests (Parsons & Chao, 1983). In addition to protecting nests, vegetation can also provide a recognition cue for a breeding partner returning to a nest site (Goutner, 1992). As yellow-legged gulls are phenotypically similar to herring gulls, the latter may have similar breeding phenology and show preference towards halophytic vegetation like sea purslane while nesting (Pons *et al.*, 2004).

The structure of the AFI module used in this study is currently only suitable for long term installation in non-tidal locations, not exposed to harsh currents and high winds. For the AFIs to be considered as a restoration or compensation tool for future large-scale renewable energy developments, research is required on a more robust and structurally sound design that will be able to withstand full tidal exposure. The high degree of biofouling on the AFI deployed in The Prince of Wales Dock also impacted on the buoyancy of the structure, as it exerted downward stress on the installation chain and plastic matrix. In this study, differences in salinity and orientation controlled biofouling species assemblages in Swansea Marina and The Prince of Wales Dock. Also, the biofouling communities on other floating hard structures present in the survey sites would be indicative of the climax communities. More research is required to compare the biofouling community assemblages on other floating hard structures and AFIs in the same survey area. Preliminary biofouling invertebrate samples were taken from pontoons as part of this study however, due to the lack of replicates and limitations of sampling from a small tender, the sampling methodology was not consistent with the scrape samples collected as part of Chapter 3 and was not included in any analysis.

In addition, the location and degree of isolation of an AFI will influence species composition and is referred to as the species-isolation relationship (Losos & Ricklefs, 2009). The AFIs in this study were deployed in relatively small heavily modified water bodies subject to high levels of disturbance from pedestrians and boat users, which may have impacted on the species richness of fish and birds observed interacting with the AFIs. The energetic costs associated with using patch habitats in urbanised areas further inland do not favour waders that commute between low tide foraging grounds and high tide roost sites (Piersma *et al.*, 1993; Dias *et al.*, 2006). Therefore, future research on AFI installations should consider the energetic costs of commuting between natural and urbanised environments and how the ‘energy landscape’

(Wilson *et al.*, 2012; Shepard *et al.*, 2013) may influence biota using the artificial habitat. With a robust and structurally sound design more research is required on AFIs installed in natural habitats such as coastal wetlands, to assess if species diversity of birds and fish varies in comparison to heavily modified coastal water bodies. Research on day and night-time bird and fish activity in association with the AFIs would also provide information on temporal variations. This could be achieved by the deployment of ARIS sonar cameras to monitor fish and infrared cameras to monitor birds.

7.1.2 Management

When considering the installation of AFIs as a habitat creation method in heavily modified coastal water bodies, the location and size of the AFI must be carefully considered in order to prevent disruption of boating activity and to ensure it is accessible for maintenance. The deployment of an AFI seaward of mean high water springs in Wales requires the issue of a marine licence under the Marine and Coastal Access Act, 2009 by the Licensing Authority, Natural Resources Wales and the production of a Biosecurity Risk Assessment. Early communication to inform relevant stakeholders such as the Maritime and Coastguard Agency, Crown Estate and local communities about the proposed works and maintenance plans will aid determination of the marine licence and gaining public approval. Although the public acknowledge that coastal habitat loss is a key environmental concern and support future installations of AFIs on the coast (Chapter 6), local concerns should be addressed during the planning stages of a development.

The ecological benefits sought by the AFI installation should also be determined during the planning stages of a project in relation to local biota. Pre-deployment benthic invertebrate, fish and bird surveys will provide information on species presence and current use of the site including abundance, species richness and behaviour plus anthropogenic disturbance levels. In addition, gaining information on the current physcio-chemical conditions of the site may aid discussions on the potential impact of installing an AFI on ecohydrology (Elliott *et al.*, 2016) and water quality. Sediment grab or scrape samples from other hard floating structures near the proposed installation, will provide details on the biofouling communities including: the presence or absence of non-indigenous species to ensure the AFI does not facilitate further spread; the primary ecosystem engineers (if present); and the predicted climax community. Data collected should aid decision making on the AFIs installation design and deployment date based on the spawning season of the primary ecosystem engineer and anticipated processes of settlement and recruitment. If chain and concrete weights are used to anchor the AFI, biofouling on the chain by mussels and ascidians will add a substantial amount of weight. The installation chain will require regular cleaning as part of a long term management plan for the AFI, to prevent the downward pull on the matrix (Table 7.1). Nevertheless, the presence of filter feeders and algae enhances localised nutrient cycling (Keene, 1980), dampens wave action on the AFI and adjacent habitats (Coombes *et al.*, 2013, 2015; O'Shaughnessy *et al.*, 2020) and via the formation of biodiverse invertebrate communities, supports essential fish habitats as feeding sites in heavily modified coastal water bodies (Chapter 4; Table 7.1).

Table 7.1: The ecological and social pros and cons of installing artificial floating islands in heavily modified coastal water bodies**.**

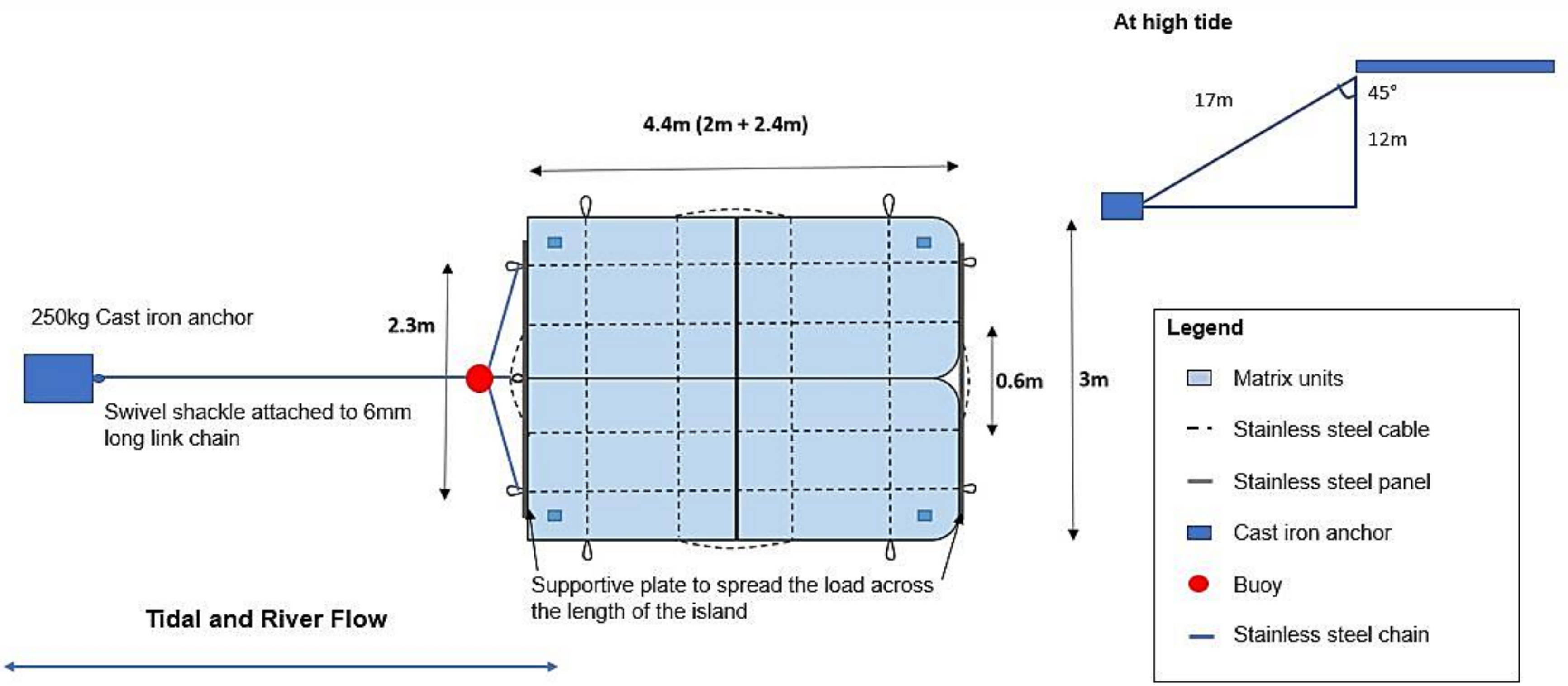
Pros	**Cons**
Ecological • Provide a surface for epibenthic invertebrates to colonise and form secondary reefs, that can dampen waves and have a 'bioprotective effect' on the adjacent habitat. • Improve water quality via phytoremediation and high density of filter feeders. • Provide feeding and sheltering opportunities for fish populations associated with essential fish habitat. • Provide feeding and resting sites for resistant and migratory birds.	**Ecological** • Short term installation with 20 year life span. • Attracts non-indigenous species and could act as a propagule for their dispersal. • Only suitable for enclosed or low velocity water bodies due composition and design of commercially sold AFIs. • Made out of plastic and can accumulate plastic on the upper surface.
Social • Aesthetic benefits associated with green infrastructure and observing wildlife.	**Social** • Short term installation with 20 year life span. • In highly productive environments AFIs require regular cleaning of the installation chain. • High densities of wildfowl can remove vegetation growth and reduce aesthetic benefits of AFI installations.

If the AFI is being installed to provide a feeding and resting site for a specific species of conservation concern, the AFI can be designed to meet the ecological needs of that species. For example, little terns (*Sterna albifrons*) will nest on sand, sand-mud and shell material with higher nest densities typically found on shell substratum (Goutner, 1990). Alternatively, vegetation cover is a primary factor influencing habitat selection during the breeding season for ground nesting species such as gulls (Wilson *et al.*, 2004; Shealer *et al.*, 2006; Overton *et al.*, 2015). During selection of suitable halophytes for transplantation on a proposed AFI installation, it is important to consider the interaction with biota as well as the species growth potential. For optimum plant growth in a saline environment the halophytes should be watered with saline solution and deployed in late spring – early summer. Pre-growing the halophytes from late winter – early spring in a greenhouse will also allow the plants to establish roots through the matrix before deployment. However, in highly saline and productive environments like The Prince of Wales Dock, the degree of biofouling prevented the penetration of roots through the matrix. In Yundang Lagoon, China root biomass negatively correlated with the abundance and biomass of black-striped mussels (*Mytilopsis sallei*) on installed AFIs (Xie *et al.*, 2019b). The impact of heavily colonised artificial structures on ecohydrology, should also be considered (Elliott *et al.*, 2016). If vegetation cover is not required to support a bird species, soft substratum such as coir matting can be added to the AFI; a quicker process than pre-establishing plant growth through the AFI matrix. The topographic complexity lost by the lack of root growth is gained by biofouling communities that establish within three to six months of the deployment depending on the season of the installation.

7.2 Conclusion

AFIs can be used as a habitat creation method in heavily modified coastal water bodies to increase the ecological potential of the site. AFIs can provide a variety of ecosystem services including phytoremediation, wave absorption and provision of nesting, feeding, and resting opportunities. The necessity for root growth through the matrix to add complexity is not required in highly productive environments where artificial structures are heavily biofouled however, this may conflict with species specific requirements or water quality improvement needs. The AFIs size, design and location should be determined based on the degree of exposure, conservation objectives and desired ecological functioning role plus social considerations, such as recreational and commercial boating activity. Future research requirements and knowledge gaps still remain, which include: the phytoremediation capacity of halophytes hydroponically grown in saline systems and the influence of AFI size; differences in ‘climax community’ formation between AFIs and other localised floating hard structures; the impact of AFIs on ecohydrology; temporal fluctuations in species activity associated with AFIs; the potential for AFI installations in tidal environments with a stronger design; the movement ecology of mobile species between natural and urbanised habitats; and to determine if public awareness and support of AFI installations is influenced by socio-economic status, linked to education and occupation and their specific motivations and interests. Gaining an understanding of these fundamental relationships between AFIs and natural ecosystems, and AFIs and society will be key for the future success or failure of restoration projects using eco-engineering methods in coastal and marine environments.

Appendix 1: Schematic diagram of proposed 13.2 m² tidal island with stainless steel cable fed through the length and width of each unit and stainless steel panels (2.3 m) installed across the two 3 m lengths.



Tidal Lagoon Power who initially funded this project were interested in AFI installations, as a potential habitat creation method that could provide shelter to adult and juvenile salmonoids within enclosed, heavily modified, tidal environments. Therefore, a 13.2 m² AFI was proposed for installation below the primary and secondary weirs of Swansea Barrage. In order to withstand the complex hydrodynamics of this proposed location, the AFI was modified to strengthen its internal structure. Each unit had four 19 mm plastic conduits running at right angles along its length and width, which can be inserted with cable for installation. It was important that the AFI pivoted and moved flexibly, with the longest length of the AFI sitting parallel with the changing water direction. Therefore, it was installed with a 250 kg cast iron anchor which created one anchor point. The plastic tubing closest to the anchor was reinforced with 18 mm stainless steel tubing. 10 mm stainless steel cable was inserted and crimped through the reinforced tubing, forming eight connecting points to aid installation. A swivel shackle was attached to the cast iron anchor, allowing the 6 mm stainless steel long link chain to freely move while the AFI was installed. The chain was connected to the AFI at three points along its shortest length (3 m), to spread the load of the AFI across the four integrated cables. Stainless steel plates were attached to reinforce the two, 3 m elevation of the AFI and prevent tearing of the matrix caused by drag forces exerted on the chain and cable. An A2 buoy fender was attached to the installation chain to add buoyancy and prevent the downward bending of the AFI caused by drag forces, especially during high tide when the chain sits at the steepest angle. The most suitable location for the AFI to be installed was determined by the maximum spatial extent at low tide and ensuring that it was safely positioned away from boat traffic. Based on the 10.5 – 12 m tidal range in Swansea Bay (Waters & Aggidis, 2016) and the 45° angle of the chain during high tide, the chain used for installation was 17 m. Accounting for the chain length (17m) and length of the AFI (4.4 m) it was estimated to cover an area of 379 m² and an 11 m radius. The installation took place on 30th April 2018, however, the AFI was dragged downstream two days later endangering boat traffic using Swansea Marina. Therefore, due to resource restrictions the AFI was installed in The Prince of Wales Dock on 17th May 2018. Although this deployment attempt failed, the installation design still has merit for future tidal island projects with a larger cast iron anchor.

Appendix 2: Fish species abundance in the Bristol Aquarium native tank experiment and the total number of recordings of each species in the lower section during the reference (LR) and deployment (LD) phase, middle section during the reference (MR) and deployment (MD) phase and the upper section during the reference (UR) and deployment (UD) phase. The percentage difference between the reference phase and deployment phase is also provided.

Species	Abundance	LR	LD	% diff	MR	MD	% diff	UR	UD	+/-
Dicentrarchus labrax	26	173	224	+29.48	509	778	+52.85	547	868	+58.68
Trachurus trachurus	18	208	98	-52.88	384	572	+48.96	138	287	+107.97
Pollachius pollachius	12	112	136	+21.43	2	34	+1,600	1	4	+300
Labrus bergylta	9	26	28	+7.69	7	4	-42.86	0	1	+100
Scyliorhinus spp.	12	13	18	+38.46	2	7	+250	5	5	0
Pleuronectiformes spp.	6	42	5	-88.10	3	12	+300	7	13	+85.71
Spondyliosoma cantharus	5	97	87	-10.31	66	142	+115.15	69	76	+10.14
Sparus aurata	4	95	40	-57.89	81	175	+116.05	1	6	+500
Mustelus spp.	3	16	2	-87.5	3	4	+33.33	7	16	+128.57

Trisopterus luscus	2	9	9	0	1	0	-100	0	0	0
Raja brachyura	1	4	5	+25	3	2	-33.33	16	14	+12.5

Appendix 3: Meteorological and water chemistry data collected at Prince of Wales Dock (POWD; small AFI) and Swansea Marina (mean ± standard error). *only one survey day for spring 2019 in The Prince of Wales Dock.

POWD	Air temperature (°C)	Humidity (%)	Wind speed (m/s)	Illumination (lux)	Water temperature (°C)	Salinity	pH	Redox potential (Eh)
Spring 2018	21.63 ± 3.07	52.03 ± 2.85	2.77 ± 1.42	32,450 ± 6257.86	18 ± 2.67	30 ± 0	8.38 ± 0.44	155 ± 4.51
Summer 2018	20.93 ± 0.68	57.8 ± 9.43	2.58 ± 0.39	26,728 ± 9,527.05	20.55 ± 1.32	32.25 ± 0.85	7.50 ± >0.01	139.25 ± 14.43
Autumn 2018	13.1 ± 0.75	55.67 ± 4.65	1.77 ± 0.43	4,829.33 ± 854.34	10.7 ± 1.16	30.33 ± 0.33	8.27 ± 0.10	146.67 ± 7.69
Winter 2019	11.75 ± 0.55	68.6 ± 1.8	0 ± 0	9,258.5 ± 2,841.5	8.65 ± 1.25	28 ± 0	8.66 ± 0.16	150.5 ± 1.5
Spring 2019	13.7*	57.3*	3*	17,600*	13.9*	28*	8.09*	116*
Swansea Marina								
Spring 2018	18.23 ± 1.42	60.57 ± 7.28	1.7 ± 1.28	22,500 ± 4,633.17	17.07 ± 2.24	15.67 ± 0.67	8.42 ± 0.47	154 ± 15.14

Summer 2018	19.6 ± 1.72	54.68 ± 7.08	1.25 ± 0.46	35,703 ± 11,1118.2	20.88 ± 1.51	13.5 ± 1.44	7.93 ± 0.43	149.25 ± 13.11
Autumn 2018	12.15 ± 0.35	49.1 ± 6.1	1.25 ± 1.25	8,237.5 ± 1,962.5	6.55 ± 0.55	11 ± 1	8.47 ± 0.24	138.5 ± 7.5
Winter 2019	13.35 ± 2.25	66.9 ± 9.8	0.8 ± 0.8	16,507 ± 8,407	7.95 ± 2.45	11 ± 1	8.61 ± 0.05	171 ± 21
Spring 2019	13.7 ± 2	55.95 ± 0.45	2.45 ± 0.95	18,500 ± 1,600	12.1 ± 1.9	9 ± 1	8.21 ± 0.33	6.5 ± 0.5

Appendix 4: Description of the behaviours used as part of the ethogram.

Resting Behaviours	Description
Standing alert	The bird is standing and stationary, turning its head frequently to examine the surroundings.
Standing resting	The bird is standing and stationary, noticeably relaxed with infrequent head movements. The individual may close its eyes for short periods of time.
Standing resting *head tucked	The bird is standing and stationary with its head turned and bill tucked under one wing.
Standing resting *head tucked on one leg	The bird is standing on one leg and stationery with its head turned and bill tucked under one wing.
Sitting alert	The bird is sitting with both feet tucked under its body and turning its head frequently to examine the surroundings.
Sitting resting	The bird is sitting with both feet tucked under its body noticeably relaxed with infrequent head movements. The individual may close its eyes for short periods of time.
Sitting resting*head tucked	The bird is sitting with both feet tucked under its body, its head turned and bill tucked under one wing.
Head raised alert	The bird is either standing or sitting down and suddenly flexes its neck muscles to raise its head and examine the surroundings. This is typically due disturbance from another individual, a load noise or as a break from another behaviour such as eating or preening.
Maintenance behaviours	

Preening	The bird uses its bill to smooth and clean feathers repeatedly on its own wings, throat, breast or region around the legs. To reach the throat, the neck is extended backwards and head bends downwards leading with the bill. For the breast the individual bends its head downwards. While preening the wings the individual will tilt head sideways and at times stretch the wing to aid cleaning.
Drying wings	The bird is standing with both wings outstretched, flexing wing muscles to move them backwards and forwards in small motions.
Stretching	The bird is standing and flexes muscles in the neck, wings and/or legs extending the feature for several seconds and returns back to a stationary position.
Tail movement	The bird is standing and flexes muscles in their tail to move their tail back and fore.
Wing movement	The bird is standing or sitting and flexes wing muscles to adjust the position of their wings.
Head shake	The bird is standing or sitting and flexes muscles in their neck to move their head back and fore.
Scratching	The bird extends its leg upwards, flexing muscles in the leg to scratch a part of its body using its toe. The head is typically lowered to allow the individual to conduct the movement.
Locomotion behaviours	
Walking	The bird is standing, simultaneously flexing muscles at the ankle, knee and hip joint extending each leg alternatively to move forwards.
Swimming	The bird is floating on a water body, extending each leg alternatively to move forwards.

Entered water	The bird is at the edge of the artificial floating island and initiates the movement by leaning forwards into the water body using power directed from its feet.
Flying	The bird extends both wings, flexing their wing muscles up and down in synchrony, gaining momentum to take flight.
Island Interaction	
Pecking island	The bird bends it head downwards and uses force in its bill to peck the artificial floating island including the coir, matrix, plants or associated fouling invertebrates.
On island	The bird lands on the island, absorbing force of the landing with its extended feet or lifts itself onto the island from the water body.
Social Behaviour	
Displaces juvenile	While landing on the artificial floating island, the individual spooks and displaces a juvenile *Larus* spp. already on the island.
Calling	The bird flexes its neck muscles to extend its neck and raise its head while vocalising.
Pecking juvenile	The bird flexes its neck muscles to extend its neck and head to peck the body of a juvenile *Larus* spp. within proximity.
Opened bill	The bird flexes muscles of the bill to close it.

Ingesting/excretory behaviour	
Eating	The bird extends its neck and head downwards and ingests plant matter or invertebrates that have fouled on the artificial floating island by pecking and swallowing the food source.
Drinking	The bird flexes its neck muscles to extend the neck and head downwards and ingest water.
Defecated	The bird fouls on the island.
Pecking buoy	The bird is sat in the water, pauses by the buoy and pecking algae or invertebrates from the buoys surface.
Other	
On buoy	The bird is standing and stationary on a buoy.

Appendix 5: Abiotic conditions recorded at the beginning of the vantage point surveys at The Prince of Wales Dock.

Date	Time	Air temperature (°C)	Wind speed (mph)	Wind direction	Humidity (%)	Cloud cover (%)	Precipitation (mm)
18/05/2018	08.15	11	6	SSE	68	40	0
23/05/2018	13.00	16	9	SSW	75	0	0
31/05/2018	06.00	16	6	ESE	95	90	0
05/06/2018	08.30	15	7	ENE	82	80	0
13/06/2018	17.00	16	11	WSW	75	95	0
02/07/2018	16.00	27	9	ENE	41	60	0
10/07/2018	11.00	20	8	SSE	66	50	0
03/08/2018	14.30	22	9	WNW	78	60	0
24/08/2018	08.00	12	15	W	70	100	2
06/09/2018	18.00	16	10	NNW	85	100	2
03/10/2018	08.30	13	3	NW	90	10	0
08/10/2018	11.15	14	13	WSW	81	90	1
25/10/2018	14.00	12	9	NNW	74	100	0

23/11/2018	08.45	6	11	ENE	92	80	0
14/12/2018	15.15	5	17	ESE	71	80	0
11/01/2019	10.00	7	4	NW	88	80	0
24/01/2019	14.30	4	2	N	96	100	1
15/02/2019	08.30	7	5	SE	78	20	0
22/02/2019	14.30	11	11	ESE	75	20	0
07/03/2019	07.15	7	11	W	85	25	0
14/03/2019	09.30	7	18	W	76	50	0
28/03/2019	13.00	11	7	S	71	0	0
12/04/2019	08.45	5	8	NE	82	100	0
03/05/2019	14.00	11	9	W	79	90	0
09/05/2019	05.45	10	7	NW	94	100	0
14/05/2019	14.15	17	11	SSE	49	15	0
24/05/2019	12.30	15	11	W	85	100	0
31/05/2019	13.45	17	13	WSW	72	80	0

Appendix 6: Abiotic conditions recorded at the beginning of the vantage point surveys at Swansea Marina.

Date	Time	Air temperature (°C)	Wind speed (mph)	Wind direction	Humidity (%)	Cloud cover (%)	Precipitation (mm)
23/05/2018	10.30	16	7	SW	75	0	0
05/06/2018	10.30	16	7	ENE	82	80	0
02/07/2018	14.00	26	13	ENE	45	50	0
10/07/2018	13.00	20	8	SSE	66	50	0
03/08/2018	17.00	22	9	WNW	78	60	0
13/08/2018	15.00	19	12	NNW	76	50	0
24/08/2018	10.30	15	15	W	57	80	0
06/09/2018	15.45	16	10	NNW	85	100	2
03/10/2018	11.00	15	5	W	87	10	0
10/10/2018	09.30	15	7	ESE	86	0	0
26/10/2018	16.00	14	11	NNW	76	60	0
23/11/2018	10.00	7	9	E	93	100	1
14/12/2018	13.30	5	20	ESE	72	100	0

11/01/2019	14.00	9	5	NW	83	100	0
24/01/2019	09.00	4	2	N	96	100	1
15/02/2019	11.00	7	5	SE	78	0	0
22/02/2019	16.15	11	11	ESE	75	20	0
28/03/2019	09.00	7	6	SE	80	0	0
12/04/2019	15.45	10	12	SSE	53	15	0
03/05/2019	10.00	8	2	WNW	91	90	0
09/05/2019	10.30	10	13	NW	83	100	0
14/05/2019	17.15	18	8	SE	45	15	0
02/06/2019	15.45	14	15	WSW	91	25	0

Appendix 7: Publication of Chapter 6 in PLOS ONE.

PLOS ONE

RESEARCH ARTICLE

Public perception of coastal habitat loss and habitat creation using artificial floating islands in the UK

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Abstract

Eco-engineering and the installation of green infrastructure such as artificial floating islands (AFIs), are novel techniques used to support biodiversity. The European Convention on Biological Diversity highlighted the development of green infrastructure as a key method of enhancement in degraded habitats. Research specifically on AFIs in marine environments has largely focused on their ecological functioning role and engineering outcomes, with little consideration for the social benefits or concerns. The aim of this study was to gain an understanding of public perception of coastal habitat loss in the UK and AFIs as a method of habitat creation in coastal environments. This was achieved via a survey, consisting of six closed and two open questions. Of the 200 respondents, 94.5% were concerned about the loss of coastal habitats in the UK, but less than a third were aware of habitat restoration or creation projects in their area of residence. There was a positive correlation between proximity of residency to the coast and knowledge of habitat restoration or creation projects. The majority of the respondents understood the ecological functioning role of AFIs and 62% would preferably want successful plant growth and avian species utilising the AFI. Nearly a third of the respondents had concerns about AFI installations, such as the degradation of the plastic matrix, long term maintenance and disturbance of native species. Despite 90.9% of the respondents supporting the installation of AFIs, the concerns of the public must be addressed during the planning stages of any habitat creation project.

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Introduction

By 2025, more than 75% of the human population is estimated to live within 100km of the coast [1–6]. Currently, 14 of the World's largest cities occupy coastal regions [4], associated with extensive infrastructure to support commercial, residential and recreational developments [1,7–12]. Due to the risk of flooding and erosion caused by rising sea levels and severe storms, densely populated areas require protection via coastal defences such as sea walls, groynes and revetments [1,7,13–17]. The combined impact of coastal 'armouring' and marine urban sprawl has caused increasing spatial disconnection of coastal habitats, habitat degradation and alterations to natural community assemblages [1,18–22]. Coastal wetlands for example, are considered one of the most threatened ecosystems, with up to 50% of global saltmarsh recorded as either lost or degraded [23–26]. Avian species are reliant on coastal habitats for nesting, foraging and roosting and are increasingly under threat, due to rising sea levels and proposed coastal infrastructure [27]. Fish larvae dispersal and recruitment can also be disrupted by coastal infrastructure, which causes fluctuations in current patterns and sediment loading [28,29]. The European Convention on Biological Diversity aims to prevent any further loss of biodiversity and ecosystem services in Europe by 2020, with the support of novel techniques such as eco-engineering and green infrastructure [30–32]. The United Kingdom (UK) Post–2010 Biodiversity Framework intends to meet these international obligations, utilising biodiversity enhancement methods where appropriate [33].

Eco-engineering refers to the modification of planned or existing structures to become multifunctional [8,34–36]. The process integrates ecological theory with the design of a proposed structure, either during the construction or post construction phase [37]. For example texture can be added to a sea wall via small indents, larger pits or water holding features, such as flower pots [32,38–40]. In highly modified marine ecosystems such as marinas and docks, eco-engineering offers a means of enhancing existing or planned structures to benefit local biodiversity, while maintaining the integral anthropogenic function of the structure [41–43].

AFIs, also referred to as floating treatment wetlands, biohavens and floating ecosystem modules, offer an alternative eco-engineering method [44,45]. These small-scale floating structures should not be confused with the larger land reclamation activities occurring around the world and proposals for floating cities to support population growth and climate migration [46,47]. In the UK, they are commercially sold by companies that provide eco-engineering solutions for silt management, plastic pollution, wastewater treatment and habitat creation. They broadly consist of a buoyant mat, planting media and emergent vegetation [48–51]. The design referred to in this study (Fig 1, *top left*), consists of a non-woven recycled plastic matrix, an integrated connection grid providing structure and closed cell polyurethane foam for buoyancy [52,53]. With established plants grown on coir matting, AFIs support a localized ecological community within the submerged roots and on the surface of the structure itself; these include algal communities, macroinvertebrates and epibiotic species [49,54]. They have largely been installed in deteriorated and over-modified freshwater habitats to improve water quality, via the removal of suspended solids and organic matter, and biosynthesis of nutrients, effectively purifying the surrounding water body [48,49,55–58]. However, interest in the use of AFIs in coastal environments has increased and is the key focus of this study [59].

Over 300 AFIs have been utilised by the Royal Society for the Protection of Birds (RSPB) to provide breeding grounds and roosting sites for divers, gulls, terns, waders and wildfowl species, within coastal wetlands in the UK [61]. Their use extends to conservation projects in San Leandro Bay Oakland, California, to provide tidal refuge habitat for the California Ridgeway's rail (*Rallus obsoletus obsoletus*) during inundation periods of the natural wetland habitat [57]. Floating structures also promote the formation of biofouling communities [44,62,63], increasing productivity and nutrient availability via deposition of organic matter within the local environment. This can attract higher trophic species such as fish, elevating the local species diversity [63–65]. For example juvenile common two-banded sea bream (*Diplodus vulgaris*) have been associated with artificial structures in high abundances, utilising installed 'biohuts' that add complexity to the localised habitat [29]. In Swansea, three AFIs have been installed in inshore marine habitats to assess the successful establishment of vegetation and their utilisation by birds, fish and invertebrates (Fig 1, *bottom left*). However, there currently is a lack of understanding of the public perception of AFIs, which could impact on the success of future installation projects [35,66–68].

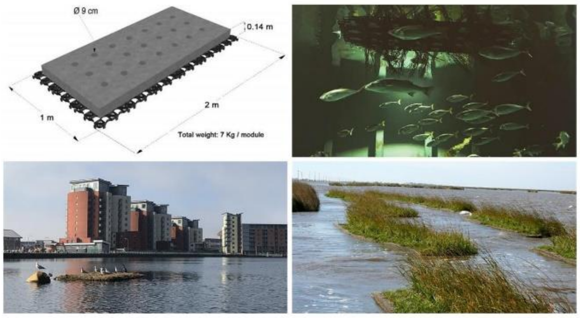

Fig 1. Artificial floating island (AFI) unit and existing installations and research. *Top left*–Schematic diagram of a $2m^2$ matrix unit, commercially sold as 'biohavens'. These AFIs consist of a non-woven plastic matrix, integrated connection grid and polyurethane foam [53]; *top right*–AFI installed in a controlled experiment at Bristol Aquarium, with 13 native, marine vertebrates; *bottom left*–AFI installed in a saline dock in Swansea known as Prince of Wales Dock; and *bottom right*–Linear arrangement of AFIs used on the coast of Louisiana, USA, for wave absorption and to reduce coastal erosion [60].

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Public awareness and perception of both national and international scale environmental concerns is important, as it influences acceptance of environmental policies and positive behavioural change within society [69,70]. Understanding the relationship the public currently have with marine ecosystems will enable the identification of any misconceptions of environmental issues and highlight the issues of concern [71]. With a better understanding of successful and failed processes of scientific communication, future environmental management and policy strategies can be improved, encouraging public support. Incorporating public awareness and citizen science campaigns into environmental conservation can positively contribute to the success of achieving new, conservation objectives [72–74]. Previously, the importance of stakeholder engagement has been highlighted during the installation of artificial reefs off the west coast of Scotland and southern Portugal [75,76]. In a number of studies worldwide, the majority of the respondents supported eco-engineering initiatives that enhanced the conservation of biodiversity [35,67,68]. However, awareness and knowledge of eco-engineering initiatives tends to be lower in Europe compared to America and Australia [67].

In the UK, public perception research has focused on the general marine environment and its protection from global concerns such as climate change [72,77–79], managed realignment [80,81], beach aesthetic and selection [82] and offshore wind farms [83]. It is important that similar information is gained on the public perception of eco-engineering methods, such as AFIs.

This study aimed to gain an understanding of the perceived importance of coastal habitat loss in the UK, in comparison to other environmental issues. Further, the study aimed to obtain information on the public's understanding of AFIs and any concerns related to AFI installations. The objectives of the survey were to assess whether the public were: (1) concerned about the loss of coastal habitats in the UK; (2) aware of local habitat restoration or creation projects; (3) aware of the ecological functioning role of AFIs; and (4) supportive of AFI initiatives as a method of habitat creation within coastal environments. Further, the study aimed to assess whether public awareness correlated with proximity of residency from the coast. The results of this study will help inform stakeholders planning on installing AFIs in UK coastal environments on public opinion and best practice before and during the AFI installation.

Methods

Survey design

The survey consisted of eight questions, subdivided into two themes: coastal habitats and AFIs (Table 1). The survey included questions with 5-point Likert scale answers, binary and multiple choice. It was restricted to six closed questions and two open questions, with an average completion time of 3 minutes, thus maximising participation. No background information was provided prior to the respondent completing the survey. Question 1 was limited to five

Table 1. The complete survey consisting of 8 questions.

Section 1: Coastal habitats	
Questions	**Possible answers**
1. Which of the following factors do you think are negatively impacting on the health of coasts in the UK? Rank each factor by importance. Urbanisation/ Coastal Developments, Flooding, Invasive species, Plastic pollution and Habitat loss.	Very important, Fairly important, Important, Slightly important or Not at all important.
2. Are you concerned about the loss of coastal habitats in the UK, such as beaches, coastal wetlands and saltmarsh?	Yes, No or Not sure.
3. Are you aware of any habitat restoration or creation projects in your area like artificial floating islands or wildflower planting? If yes, any further details of the type of project and in what location can be added here.	Yes or No.
Section 2: Artificial floating islands	
Questions	**Possible answers**
4. Artificial floating islands consist of a recycled plastic matrix and growing medium, that plants are able to grow roots through. They are often installed in lakes and rivers. What do you think artificial floating islands are installed for? Tick any answers that you think are correct.	Aesthetic, To create habitat and support biodiversity, To support boating activity, To improve water quality, To collect litter or Other.
5. On some occasions it is difficult to maintain both plant growth and bird use. Which of the following scenarios would you prefer if an island were installed in your local area?	Bird activity and no plants, Plants and fencing with roots growing through the island for fish, Plant growth but not fully covering the island and bird activity or Not sure.
6. Would you have any concerns about the installation of an artificial floating island?	Open question.
7. Would you support future installations of artificial floating islands or other habitat creation projects along the coast?	Yes, No or Not sure.
8. How far from the coast to do live?	1 mile, 5 miles, 10 miles or 20 miles +.

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factors for simplicity and the factors selected were all environmental concerns prevalent in the UK. In terms of personal information, only distance that the respondent lived from the coast was determined. Other demographic information was not collected in this survey, such as age and occupation, as these details were not required to meet the study objectives. However, more detail about the location of residency was inferred from Question 3, addressing awareness of habitat restoration initiatives and assuming that participants had greater knowledge of projects in their local area. Question 5 addressed a common issue associated with high numbers of wildfowl and maintaining plant growth on AFIs. Additionally, AFIs can be specifically installed without vegetation to attract certain avian species that require only substrate for breeding [61,84].

Survey collections

The target demographic was members of the public living in the UK, aged 18 or above. One respondent living in the Netherlands completed the survey and was included in the analysis. The survey was self-administrated using the survey tool 'Survey Monkey' (https://www.surveymonkey.com) and went live on 27th January 2019. The survey was live for 68 days, until 5th April 2019. The survey was circulated on social media platforms such as Facebook and Twitter and members of the public were approached in Bristol Aquarium and Swansea. The survey was also circulated via community forums such as such as 'Maritime Quarter Residents Association' and 'Uplands and Brynmill community forum', to gain information on the opinion of local residents, who may have observed the AFIs in Swansea. A total of 200 surveys were collected during the 68 days that the survey was live (online, n = 170; in person, n = 30). The information provided during the online surveys and in person was the same, minimising any bias results. Swansea University ethics committee approved research conducted in this study (SU-Ethics-Student-030719/1106).

Data analysis

Descriptive statistics were used to summarise results from each question of the survey. Chi squared tests were used to assess whether there was a relationship between the distance the respondent lived from the coast and their (1) concern of coastal habitat loss; (2) awareness of habitat restoration and creation projects; (3) awareness of AFIs and their ecological functioning role; and (4) concerns related to AFIs being installed. Comments that addressed concerns about AFI installations (Question 6; Table 1) were organised into categories appropriately. Statistical tests were completed using R 3.6.0 statistics software.

Results

Of the 200 respondents, 29.5% (n = 59) lived within 1 mile of the coast, 23% (n = 46) within 5 miles, 17.5% (n = 35) within 10 miles and 30% (n = 60) greater than 20 miles.

Coastal habitats

The majority of respondents considered plastic pollution (77.8%, n = 154) and habitat loss (70.9%, n = 139) to be very important factors affecting the health of coasts in the UK (Fig 2). Urbanisation was also considered to be a very important factor by 43.2% of the respondents (n = 86). There was no significant relationship between perceived importance of coastal habitat loss and proximity of residence to the coast ($\chi^2 = 2.86$, d.f. = 3, $p = 0.41$, n = 200). Less than a third of the respondents considered flooding (28.4%, n = 55) and invasive species (24.2%, n = 47) to be very important factors affecting the health of coasts in the UK. Three of the

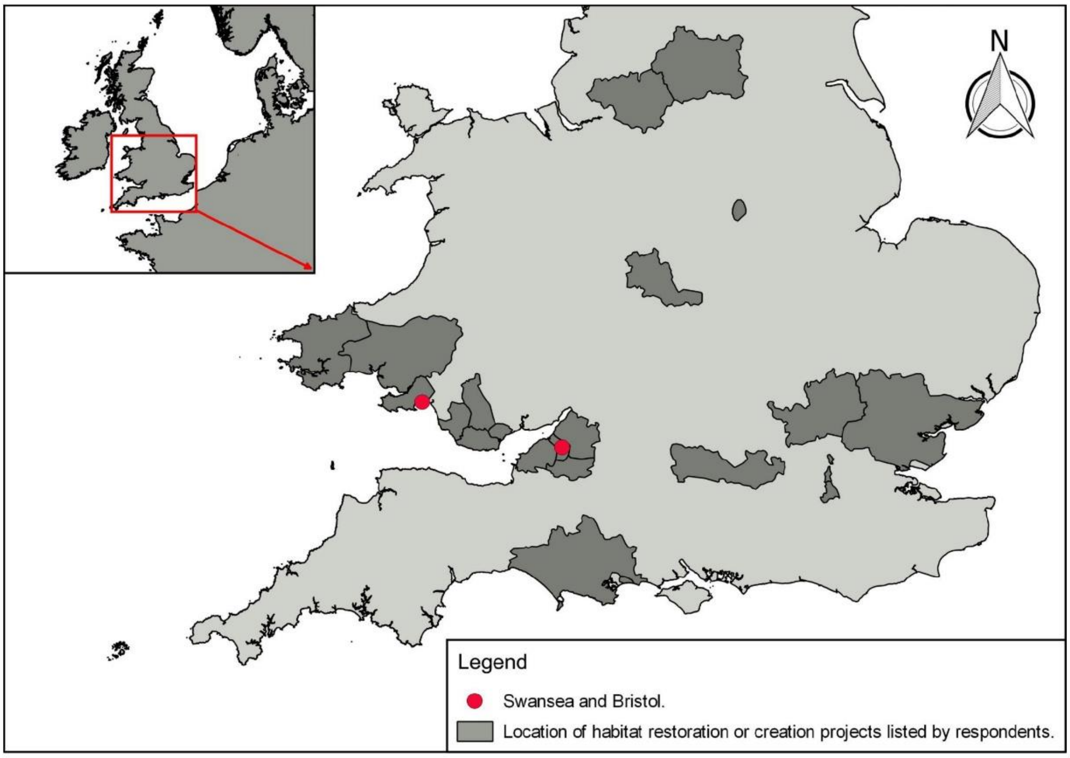

Fig 2. The perceived importance of factors negatively impacting on the health of UK coasts.

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factors were perceived as not important at all. These were invasive species (5%, n = 9), flooding (4%, n = 7) and urbanisation/coastal developments (1%, n = 2).

The majority of respondents were concerned about the loss of coastal habitats in the UK (94.5% n = 189). Under a third of the respondents (28.5%, n = 57) were aware of habitat restoration or creation projects in their area of residence and this was dominated by respondents living within 1–5 miles of the coast (70%). There was a significant relationship between the respondents' awareness of habitat restoration and creation projects and the proximity of residence from the coast ($\chi^2 = 8.95$, d.f. = 3, $p = 0.02$, n = 200). The respondents that provided further detail to Question 3 (n = 34) mentioned projects located in South Wales and England (Fig 3) and 52% of the schemes were related to marine environments, rather than terrestrial or freshwater habitats.

Artificial floating islands

As the respondents could give multiple answers on the perceived purpose of installing an AFI (Table 1, Question 4), there were 385 responses; 306 understood the ecological functioning role of AFIs ('to create habitat and support biodiversity' n = 196; 'to improve water quality'

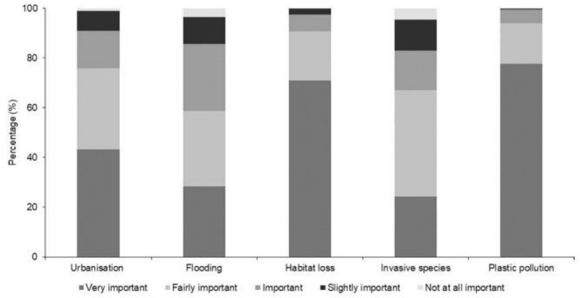

Fig 3. The location of habitat restoration or creation projects listed by the respondents of the survey (n = 34). The projects mentioned by respondents were located in 23 counties in England and Wales. Each project is represented by county it is located in [85,86].

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n = 110). There was no significant relationship in public awareness of the ecological functioning role of AFIs between the four proximity categories ($\chi^2 = 3.64$, d.f. = 3, p = 0.30, n = 200).

The majority of the public surveyed preferred to have both successful plant growth and birds utilising an AFI (62%, n = 125, Fig 4). One third of the respondents preferred the installation of an island with successful plant growth, maintained by the inclusion of fencing (33%, n = 67). High levels of bird activity with no plants growing was the least popular response (4%, n = 9).

Question 6 of the survey allowed the respondents to voice any concerns regarding AFI installations on the coast; 33% of the 200 (n = 66) chose to comment on their concerns. These were broadly categorised into maintenance, recreation, aesthetic, plastic pollution, disturbance and invasive species concerns (Fig 5). The definition of each term based on the respondents' answers are outlined in Table 2.

Plastic pollution (n = 33) and the long-term maintenance (n = 26) of an installed AFI were the key areas of concern by the respondents of the survey (Fig 5). The majority of the respondents would support the future installation of AFIs along the coast (90.9%, n = 181), with the remaining respondents either unsure or against the method of habitat creation.

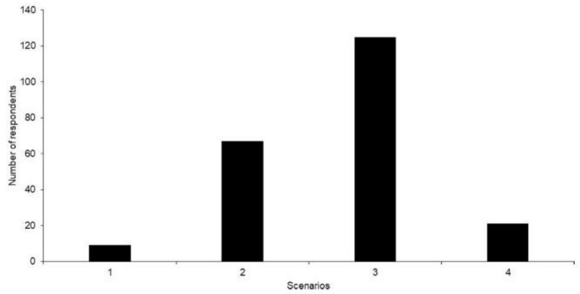

Fig 4. The respondents' preference of an installed artificial floating island in their local area based on five scenarios. (1) Bird activity and no plants; (2) Plants and fencing, with roots growing through the island for fish; (3) Plant growth, but not fully covering the island and bird activity; and (4) Not sure.

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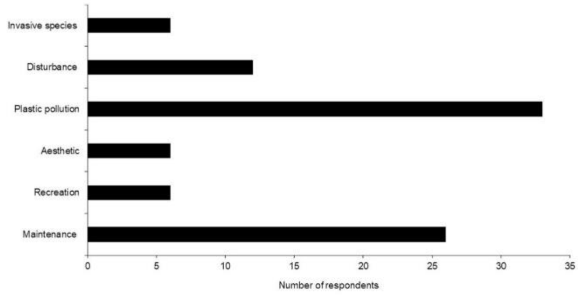

Fig 5. The number of concerns raised by respondents. These have been categorically organised into maintenance, recreation, aesthetic, plastic pollution, disturbance and invasive species.

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Discussion

Artificial structures are proliferating in marine environments in the form of coastal defences [13,14,17] and infrastructure to support shipping, transport, commercial, recreational and residential developments [1,7–12]. Current legislation including the European Convention on Biological Diversity and the UK Post-2010 Biodiversity Framework, address that novel techniques such as eco-engineering have a role to play to prevent any further loss of biodiversity and ecosystem services caused by anthropogenic activities [30–32,87]. Alongside meeting legislative targets, it is also important to engage with the public on environmental issues and conservation approaches that could be introduced. Without public engagement, the awareness and public support of future projects cannot be guaranteed. This study aimed to gain an understanding of the public's perception of coastal habitat loss and AFIs as a habitat creation method.

The majority of participants of this survey were concerned about the loss of coastal habitats in the UK and consider plastic pollution, habitat loss and urbanisation as very important factors negatively impacting on the coast (Fig 2). Due to the release of documentaries such as 'A Plastic Ocean' in 2016 and 'Blue Planet II' in 2017, public awareness has increased substantially on the impacts of litter and specifically, non-biodegradable material in ocean ecosystems. The UK public also demonstrated an understanding of the deterioration of marine environments in a previous study, where 95.8% of respondents considered marine habitats to be of 'fair to poor' health [72,78]. Pollution and climate change are consistently mentioned as the most concerning environmental issues for members of the public, in the UK and abroad [35,77,88]. In this survey, coastal urbanisation, flooding and invasive species were perceived as less important factors by some respondents (Fig 2). This could be due to a lack of understanding of

Table 2. Definition of the six concerns listed by respondents in Question 6 of the survey.

Concern	Definition
Maintenance	Damage or detachment of the island during severe weather or as a result of vandalism.
Recreation	Disrupt boating, kayaking or surfing activity on the coast.
Aesthetic	It is unnatural and a potential eyesore.
Plastic pollution	Degradation of the plastic matrix into the water body.
Disturbance	Noise pollution during installation and impact on natural processes.
Invasive species	Encourage the presence or spread of a non-native species that could cause damage to the ecosystem.

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secondary impacts of developments, such as light and noise pollution and fluctuating hydrodynamics that can result in flooding. The importance of flooding to the respondent can also be governed by personal experience [89]. The individuals' socio-economic status linked to education and occupation and their specific motivations and interests, have also been identified as factors that drive awareness of environmental issues [77,90]. These details were not included as part of this survey, as the information was not required to meet the study research objectives. However, this does limit comparisons to other public surveys.

The majority of the public desire greater protection and conservation of the UK marine environment, from fishing and other damaging, exploitative practices [72]. However, as part of this survey under a third of the respondents were aware of habitat restoration or creation projects in their area. The respondents that did mention restoration and/or creation projects mostly lived within 1–5 miles of the coast and 52% of the schemes were related to marine environments, rather than terrestrial or freshwater habitats. Examples of schemes mentioned across all habitat types included: dune slack management in Kenfig National Nature Reserve, Bridgend, to promote early succession of orchids; creating habitats for common kingfisher (*Alcedo atthis*) populations in the Lee Valley, Essex, via river management and; habitat restoration at Saltwells Local Nature Reserve, Dudley (Fig 3). The focus on marine conservation and policy could be a direct result of greater national awareness, personal interest based on residential location or occupation. The correlation between proximity to the coast, marine conservation and policy knowledge was discovered during a large scale survey in the United States [77,90]. However, this outcome could also be a result of the marine focus of the survey. To reduce potential bias towards marine projects, wildflower planting was also mentioned as a terrestrial habitat restoration and/or creation method in Question 3. The respondent was also asked to mention projects within their local area (Table 1).

For future research, more detailed demographic information would be desirable to gain deeper insight into relationships between social and economic background with views on marine conservation awareness and AFIs.

In this survey, the majority of respondents showed an understanding of the ecological functioning role of AFIs. This could be linked to a positive shift in perception in the UK of the importance of wetland biodiversity and support towards wetland restoration [91]. Overall, the survey confirmed that the public preferred a vegetated island utilised by birds (Fig 4). Within urban environments green landscapes play a significant role in health and mediating the stresses of daily life [92,93]. This could have contributed to the respondents' positive association with vegetation growth on the AFIs. Water quality of natural wetlands, the presence of emergent vegetation and trees and habitat value to local wildlife, were factors viewed as important in assessing wetland health in Australia [94]. There is however, evidence that a lack of understanding of ecological values is linked to a negative view of wetlands [94–96].

Public and stakeholder perception studies of artificially created habitats have largely focused on benthic habitats including artificial reefs, concrete flowerpots used in the intertidal zone and coastal defence structures [35,67,68,75,76,97]; therefore, limiting comparisons of the results from this study. In a preliminary study, stakeholders including engineering and ecological consultants, academics and statutory bodies unanimously supported the implementation of multi-functional artificial structures, which prioritised ecological benefits within coastal environments [66]. The study also highlighted that 'education and outreach' was one of the lowest assigned considerations by stakeholders, while a greater evidence base of the ecological benefits was seen as desirable. This illustrated the importance of accessible research and a strong evidence base for stakeholders [12]. It also demonstrated the lack of importance placed on public engagement by stakeholders, which could be limiting future public support of the implementation of eco-engineering and artificial habitat creation projects.

Nearly a third of the respondents had concerns about the installation of AFIs in the marine environment. These concerns largely focused on the future degradation of the AFI matrix and potential for the islands to become plastic pollution (Fig 5). Additionally, the public were concerned about the long-term maintenance and aesthetic of the island; 'would it look unnatural and therefore un-aesthetic?', 'how will they be maintained?', and 'would the plastic in the matrix enter the food chain?'. Other comments were related to the potential disturbance of commercial and recreational boating, surfers and native wildlife. During the planning stages of an AFI installation, it is important that research and monitoring is undertaken by the individual or company responsible, on the environmental conditions of a proposed island location. This includes factors such as average wind speed, water velocity and tidal height (if applicable). In addition, salinity and pH should be assessed as certain metals are susceptible to corrosion, based on the surrounding water chemistry. This information will aid decisions on the appropriate size, configuration and method of installation of an AFI, that minimises disruption of native fauna and ensures it is securely installed. Research and open communication with potential stakeholders and members of the public, will also ensure that no recreational activities are disrupted by the installed AFI.

AFIs have an approximate life span of 20 years and this varies depending on its location [50]. As most AFIs are installed in ponds, reservoirs and rivers, case studies of islands exposed to waves, tides, marine biofouling and saline conditions are limited. Laboratory experiments have demonstrated that the size and configuration of the AFI determines the force (kilonewton, kn) exerted on the islands structure. Prior to the installation of an AFI, a maintenance and potential disposal plan should be established and made publicly accessible. This will ensure the long-term success of an AFI and reassure local residents that the island will be maintained and disposed of appropriately, to prevent potential degradation of the plastic matrix. If the AFI is installed where invasive species are present, the island should not be translocated to prevent the potential spread of invasive species.

In conclusion, the majority of the respondents would support the installation of AFIs along the coast, as they recognised coastal habitat loss as an important environmental issue. The successful establishment of plants and positive benefits to local wildlife, were equally important factors valued by respondents. There were concerns regarding the longevity of an artificially created habitat, which must be rectified with thorough strategic planning and appropriate aims, based on the location of the proposed AFI installation. Further research is required on socio-economic factors that could be influencing public awareness of habitat loss and artificially created habitats within urban ecosystems.

Supporting information

S1 Appendix. Dataset of the survey. Includes the 200 respondents' answers to the eight questions of the survey that was open from 27th January– 5th April 2019.
(XLSX)

S2 Appendix. Survey questions. The eight survey questions answered by respondents.
(PDF)

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Methodology: Jessica Ware, Ruth Callaway.

Project administration: Jessica Ware.

Supervision: Ruth Callaway.

Visualization: Jessica Ware.

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Writing – review & editing: Ruth Callaway.

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