

Interaction and impact of cultivated macroalgae and growing structures on faunal community assemblages and surrounding ecosystems.

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Cover Photograph: Two-spotted goby (*Gobiusculus flavescens*) hiding in cultivated *Saccharina latissima*, Dingle Bay Seaweed Ltd. Licensed seaweed farm in Ventry Harbour, Ireland. © Dr. Jose M. Fariñas-Franco

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¹ All photographs in this report by the author Dr. Jose M. Fariñas-Franco unless stated otherwise

Summary

GENIALG aimed to study the possibilities for upscaling and streamlining seaweed production as a way not only to satisfy growing demand from industry but also to promote local economic activity and develop seaweed aquaculture in an efficient and environmentally sustainable way. In GENIALG, work package 6.1 focused on providing an assessment of the impact of seaweed farms on coastal ecosystems using a medium sized farm (1.8 ha) in the southwest coast of Ireland (Ventry Harbour, Co. Kerry) and The Netherlands (Eastern Scheldt) as case studies.

No negative impacts were detected on abiotic seafloor condition other than seasonal variability in organic matter linked to primary production and within site spatial variability in both mean grain size, kurtosis, and skewness. Observational data from both test farms indicates that seaweed farming does not exert, at the scales and under the conditions studied, a negative influence on its surrounding environment but rather provide positive ecosystem interactions.

After two full years of monitoring of the test farm we concluded that *Zostera marina* under and near the farm biomass depended largely on location, with deeper areas to the north and west, more exposed to bottom swell and thus higher turbidity, yielding the lowest seagrass biomass away from the farm. More importantly, biomass increased towards the farm, where more sheltered conditions are found and were highest under it. The shallower locations under and away from the farm to the east and south yielded the highest seagrass biomass values. Spatial variability could not be directly linked to the effect of the farm but to natural variability influenced by co-related environmental parameters, likely light penetration, depth, and turbidity. There was nonetheless seasonal variability in biomass, to be expected in a benthic primary producer, with highest standing biomass values recorded in summer and autumn.

Reduced water flow and turbidity within the seaweed farm was likely responsible for higher photosynthetically active irradiance values recorded under the longlines compared to distant controls and were linked to significantly higher seagrass shoot abundance and coverage under the farm. The role of seaweed farms coupled with their de-facto role as exclusion zones providing protection against physical disturbance from anchoring or bottom trawling is evidence of their positive effect on the environment, facilitating the conservation of benthic habitats of high conservation value such as seagrass.

Biodiversity within cultivated seaweed was relatively high and similar (e.g. species numbers) to that recorded in natural kelp beds. The results showed no significant differences in the number of taxa and

their abundances between the two cultivated kelp species studied. Seeded droppers yielded significantly higher abundance of macroinvertebrates, but similar number of taxa, compared to unseeded rope droppers used as controls, attracting a community dominated by amphipods *Jassa* spp. and the non-native Japanese ghost shrimp *Caprella mutica* in both the Irish and Dutch sites. The latter also host other non-native invasive species such as Japanese knotweed *Sargassum muticum* and the sea squirt *Styela clava*. Mussel spat were abundant during the seaweed growing season and took over the assemblage after the summer, following the degradation of the kelp fronds. Large number of medium sized mussels and the remnant seaweed holdfast created a complex habitat that attracted velvet swimming crabs and brown crabs, both species of commercial interest, which used the site as a nursery.

The surveys conducted at the test farms in Ireland and The Netherlands provide, in agreement with predictions from desk based studies, confirmation of the significant positive effect of seaweed farms for biodiversity and their role as nursery habitats for fish and shellfish. Some of the fish species recorded in Ireland included commercially important numbers of juvenile pollack *Pollachius pollachius* and near-threaten thornback ray *Raja clavata* highlighting the importance of the activity to preserve natural habitats and populations of species of conservation concern. However, although small farms of the scale currently in Europe, do not represent a threat to the environment, it is possible that larger farms (in the tens of squared kilometres) or farms next to each other could have a cumulative effect, especially on habitats of high conservation value.

The findings represent evidence of the role of seaweed farming in the sustainable development of coastal economies and communities, maintaining healthy ecosystems which in turn provide valuable ecosystem services. This role aligns with Aichi Target 11 on marine biodiversity protection and UN Sustainable Development Goals (SDG) 2 food security and 14 on oceans as well (Le Gouvello et al. 2017), potentially helping to achieve EU targets for Good Environmental Status (GES; Fariñas-Franco et al. 2014; Hasselström et al. 2018) and could play a role in the European Green Deal Strategy boosting the efficient use of resources by moving to a clean, circular economy, restore biodiversity and cut pollution (European Commission 2019). Maximising this role and ecosystem services should be incorporated in management strategies and be considered in the site selection and licensing process.

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1 Introduction

The interest in novel natural resources has increased as a response to food demands by a growing world population. Seaweed has been identified as being one of these resources, resulting in an increased interest in cultivating seaweeds that may deliver a broad range of food but also non-food products (Holdt and Kraan 2011; van den Burg et al. 2016; FAO 2020). Seaweed cultivation is also becoming increasingly recognised as a more environmentally friendly alternative to wild harvesting, and it is currently the main source of raw algal product (Rebours et al. 2014; Moffitt and Cajas-Cano 2014). Seaweed aquaculture predominantly occur in China and other countries of the Asian region, dominating the sector with over 90% of the global production (Nayar and Bott 2014; Chung et al. 2017; Ferdouse et al. 2018; Campbell et al. 2019). In Europe, the seaweed aquaculture sector is still in its infancy but has increased in importance in the last decade (Rebours et al. 2014; Buck and Langan 2017; Stévant et al. 2017). Kelp species of the order Laminariales in particular (e.g. *Saccharina latissima*, *Laminaria digitata*, *Alaria esculenta*) are best suited for cultivation in temperate waters in northern Europe (Reid et al. 2013; Kerrison et al. 2015).

While the main research focus in kelp aquaculture has generally been on improving the efficiency and scale of the production of seaweed to satisfy demand (Buck and Langan 2017; Araújo et al. 2021), strain selection and domestication and product diversification (Kim et al. 2019), there is a need to identifying the ecosystem services provided by seaweed aquaculture and the prevention of negative effects (Stévant et al. 2017). As a new but rapidly expanding sector, we are in a good position to learn from the experiences in other aquaculture activities, to align the licensing and site selection process with European policy (Buschmann et al. 2017; Walls et al. 2017a). Despite the existence of a number of desktop studies that have reviewed the potential environmental interactions of seaweed aquaculture (Buschmann et al. 2014; Wood et al. 2017; Hasselström et al. 2018; Campbell et al. 2019), there is a dearth of observational studies from the field and test farms identifying and quantifying these effects. Most studies are from Asia and eastern Africa (Ólafsson et al. 1995; Eklöf et al. 2006a; Zhang et al. 2009; Liu et al. 2016; Zheng et al. 2019), and in European temperate waters knowledge of seaweed cultivation impact on natural habitats is scarce and site specific. There is work conducted in small to medium sized farms in Ireland by Walls et al. (2016, 2017b, a) and in Sweden more recently by Visch et al. (2020) suggesting low intensity impacts largely compensated by beneficial effects. However, nutrient limitation models predict significant reductions in phytoplankton biomass, depending on farming intensity and size of the farm (Aldridge et al. 2012).

Identifying potential environmental effects of offshore seaweed aquaculture

A key issue to address in an ecosystem-based approach to inform licensing seaweed aquaculture projects, especially when upscaling, is their effect on the surrounding coastal ecosystems. The work of Wood et al. (2017) looked at some of these key environmental considerations concluding that, in a UK context, there was no need to draw up a new regulatory framework but rather update existing marine licensing legislation to include environmental and social considerations. Potential environmental effects include changes in hydrodynamic conditions, sediment transport patterns in and around the farm and potential changes in siltation rates, seafloor organic enrichment rates and light penetration that could affect benthic communities and primary producers. Wood et al. (2017) estimated no-major population level changes or impacts in the ecosystem but identified numerous knowledge gaps that needed to be covered by observational studies, from effects on the physical environment to nutrient dynamics and marine life. Campbell et al. (2019) conducted a similar systematic review to identify the factors that need to be considered to minimise any potential negative effects from seaweed aquaculture with a view to inform site selection and seaweed cultivation methods used. The elements covered were a close match to those identified by Wood et al. (2017) including absorption of light, nutrients, carbon, and wave energy; the addition of artificial material (increasing the potential for pollution and entanglement of megafauna) and noise; the release of dissolved and particulate organic matter; provision of habitat for diseases, parasites and non-native species; the release of reproductive material leading to competition and hybridisation between domesticated strains and wild populations; and habitat creation for plankton, benthos and epi and megafauna. In agreement with Wood et al. (2017), Campbell et al. (2019) identified knowledge gaps while providing environmental risk assessment matrices and decision-making flow diagrams to identify impact thresholds and subsequently design monitoring programs that do not burden the industry unnecessarily.

The factors and potential effects, negative and positive, that relate to offshore seaweed aquaculture, and which could be considered as the foundation to inform the environmental monitoring and evidence gathering followed by GENIALG (this report) can be summarised as follows (see also Figure 1):

1. Potential effects on the physical environment

1. Nutrient and pollutant removal:

- Ameliorate eutrophication or

- Competition with primary producers, reducing phytoplankton and/or macroalgae and seagrass biomass leading altering trophic fluxes.
- 2. Wave and current attenuation: coastal protection or erosion
- 3. Sedimentation: both from alteration in local current patterns, enhancing water clarity, as farms might act as sediment traps (Wood et al. 2017), or by increasing the amount of dissolved and particulate organic matter, which could in turn increase the provision of food to opportunistic seabed species from shedding. A potential increase in organic content would raise biological oxygen demand through an increase in respiration processes and the reduction of oxygen levels (Aldridge et al. 2012).

2. Potential positive effects on local biodiversity

1. Cultivated seaweed as keystone species:
 - Essential habitat for fish and shellfish (spawning grounds or settlement surfaces).
 - Habitat and feeding opportunities for other fauna.
 - Support of new food webs by presence of cultivated seaweed (see Ratcliff et al. 2016 and references therein).
2. Longlines and ancillary structures as artificial habitats:
 - Provision of settlement substrate.
 - Refuge/aggregation devices.
 - Foraging opportunities for fish, birds, and other predators (e.g. crabs).

3. Potential negative effects on local biodiversity

1. Cultivated seaweed:
 - Increased deposition of organic matter leading to change in community composition and biotope type.
 - Shading limiting benthic primary producers (kelp, seagrass).
 - Nutrient competition with primary producers, e.g. phytoplankton.
 - Steppingstones for non-native and invasive species.
2. Ancillary structures:
 - Abrasion and smothering.

- Scouring
- Steppingstones for non-native/invasive species.

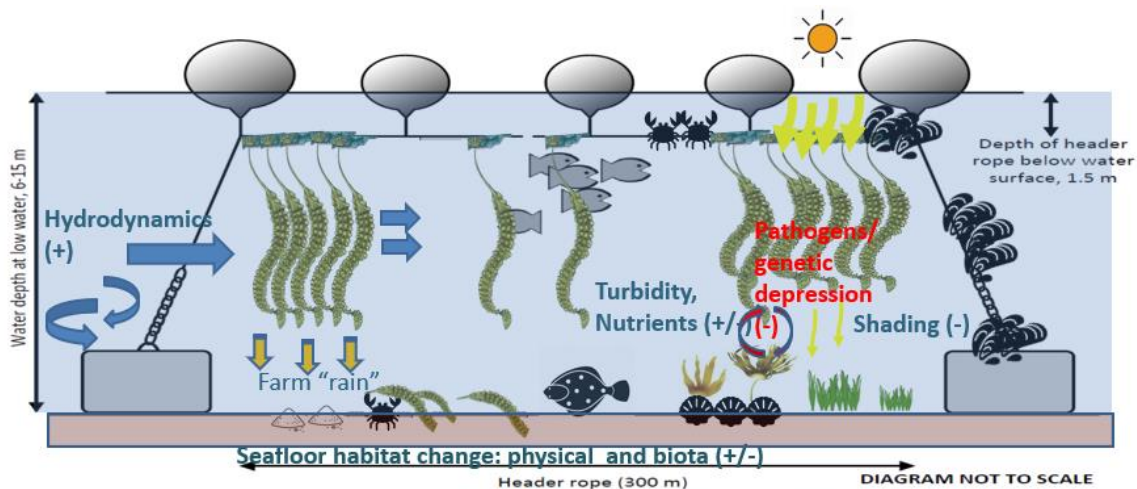


Figure 1 Schema of possible interactions between seaweed aquaculture sites and environment considered in the monitoring conducted at the test farms studied by GENIALG.

Rationale of GENIALG Task 6.1: Potential impacts of seaweed farming on local ecosystems

Seaweed-dominated ecosystems, and a seaweed farm can be regarded as one, perform a number of functions, including provision and maintenance of biodiversity, primary and secondary productivity, carbon and nutrient cycling and water flow attenuation, which are translated in important ecosystem services. Besides providing valuable biomass, seaweed mariculture, in monoculture or integrated multi-trophic aquaculture (IMTA) systems, may also provide additional ecosystem functions. As seaweed cultivation is a novel form of utilizing the marine ecosystem it should be approached with care, despite its expected low environmental impact (Folke et al. 1998; Roberts and Upham 2012).

Whereas the abiotic footprint (e.g. nutrient removal, CO₂ fixation) has been investigated to some extent, the biological consequences of large-scale farming initiatives are far less foreseeable and will likely require scrutiny. The attitude of the public to such developments is also not well known. Thus, GENIALG placed emphasis on monitoring the ecological and social footprints of seaweed cultivation, with respect to the scale up of production. These multidisciplinary data were fed into models to determine the carrying capacity of farms and quantify the benefits of cultivation using an ecosystem approach (see Broch et al., 2021). The knowledge created will be used to develop best practices to improve management to maximise economic benefit, biosecurity, and social acceptability.

Work package 6 of GENIALG was subdivided into four main tasks. Specific Objective 1 of work package 6 was to “Assess and monitor the potential biotic and abiotic impacts of farms on surrounding ecosystems” addressed by Task 6.1 “Potential impacts of seaweed farming on local ecosystem”.

Task 6.1 was itself subdivided into the following subtasks:

- 6.1.1 Measure and monitor potential biotic and abiotic impacts of seaweed cultivation on local ecosystems.
- 6.1.2 Assess the importance of gene flow between farms and wild populations.
- 6.1.3 Measure and monitor abiotic factors to validate models and evaluation of farm footprint. Links to 6.1.1.

The bulk of the work presented in this report relates to subtasks 6.1.1 and 6.1.3 and places emphasis on faunal communities and abiotic interactions recorded at tests farm in Ireland and The Netherlands for the duration of the project. The data collated as part of subtask 6.1.3 was used to identify potential physical changes in the environment from cultivated seaweed during the studies, parametrise faunal community and biodiversity data, and validate ecosystem models being developed by other partners (e.g. Sintef/WUR in Task 6.2 see Broch et al. 2021).

There were two deliverables for which UOG was responsible as a partner in the GENIALG project:

- **D6.1:** Faunal biodiversity datasets, uploaded to European biodiversity database, e.g. through biodiversity information system for Europe (Month 42).
- **D6.2** Report or manuscript submitted to peer-reviewed journal describing the interactions and impact of cultivated algae and growing structure on faunal community assemblages (Month 36).

Deliverable 6.1 included biotic and abiotic datasets already uploaded to a public repository via the EMODNET data ingestor while this report constitutes Deliverable 6.2.

Aims and objectives of this report

Macrophytes, including kelps, as keystone and foundation species positively contribute to biodiversity (Norderhaug et al. 2005; Christie et al. 2009). Cultivated seaweed along with ancillary support structures form a three-dimensional habitat that may enhance local biodiversity by generating a substrate for sessile organisms and a sheltered habitat for mobile species, including fish (Bartsch et al. 2008; Christie et al. 2009; Wood et al. 2017; Visch et al. 2020a) while the holdfast acts as a habitat for

multiple organisms such as crustaceans (Thiel and Vásquez 2000; Bergman et al. 2001). It should however be taken into account that age, morphology, and location in the water column of a cultivation unit do differ from natural kelp forests, altering species composition and richness and rather than expanding the natural habitat it forms a novel habitat for seaweed affiliated species (Walls et al. 2016). In addition, the uptake and excretion of nutrients and organic compounds by cultivated seaweeds can alter the phytoplankton composition as well as the benthic community (Perini and Bracken 2014; Wood et al. 2017; Moschonas et al. 2017). An increase in biodiversity is also observed in similar forms of low trophic aquaculture, like shellfish cultivation affecting biodiversity on multiple levels and overall enhancing it (McKindsey et al. 2011; Callier et al. 2018). Although general principles on the interactions between seaweed cultivation and biodiversity are easy to visualize, and have been well outlined in numerous studies/reviews (e.g. Roberts and Upham 2012; Hughes et al. 2013; Wood et al. 2017; Campbell et al. 2019), it remains unclear to what extent these processes contribute to ecosystem functioning due to a lack of quantitative data.

While the broad aims of Task 6.1 in GENIALG were to identify potential impacts (negative or positive) caused by the cultivated green and brown algae and the ancillary growing structures on the surrounding environment, the present report focuses on one of those species, the sugar kelp *Saccharina latissima*, and two study sites, Ventry Harbour (Ireland) and Schelphoek (The Netherlands), as case studies to provide evidence. In GENIALG, we devised a scientifically sound survey plan to evaluate and quantify the condition of the benthic habitats, including seagrass beds, located under the seaweed culture operations, and described the biodiversity and faunal communities associated with the cultivated kelp. Monitoring programs were therefore established to gather a broad range of abiotic and biological data at two test sites, in Ireland, operated by Dingle Bay Seaweeds Ltd., and a second one in the Netherlands, operated by Seaweed Harvest Holland. The aim of these monitoring programs was to monitor biodiversity in and around the reference farms, to assess effects of this new habitat and increase of seaweed biomass, on diversity of other species, including commercial fish and unwanted invasive species. Parallel to these studies, seaweed pathogens and other pests, as well as the evolution of genetic diversity of wild seaweed populations neighbouring farms were conducted but are not covered by this report. Relevant parameters were monitored and used to validate the carrying capacity models developed as part of Task 6.2 (see Broch et al., 2021).

Therefore, in this report we present quantitative data on biodiversity obtained during sampling campaigns at two test sites in Ireland and The Netherlands collated using a comprehensive suite of surveys and measuring equipment (see Figure 2). The approach was grounded on solid scientific principles, following Before-After-Control-Impact (BACI) (Underwood 1992) accounting for temporal

and spatial variability that could detect how much of the variability observed can be attributed to the aquaculture activities. The approach is replicable and was used to inform surveys conducted at a second test farm in the Eastern Schelde and incorporated into the manual on best practices for seaweed farming developed by GENIALG as part of Deliverable 6.9 (Tett et al. 2021). The data presented comprises a broad variety of measurements, biotic and abiotic and represents an ambitious piece of work that aims to support evidence-based licensing of new seaweed aquaculture sites and the management of existing ones using an ecosystem-based approach. Ultimately, this study broadens our understanding of the ecosystem impact and potential benefits this novel form of aquaculture could have under the current climate and biodiversity crisis.

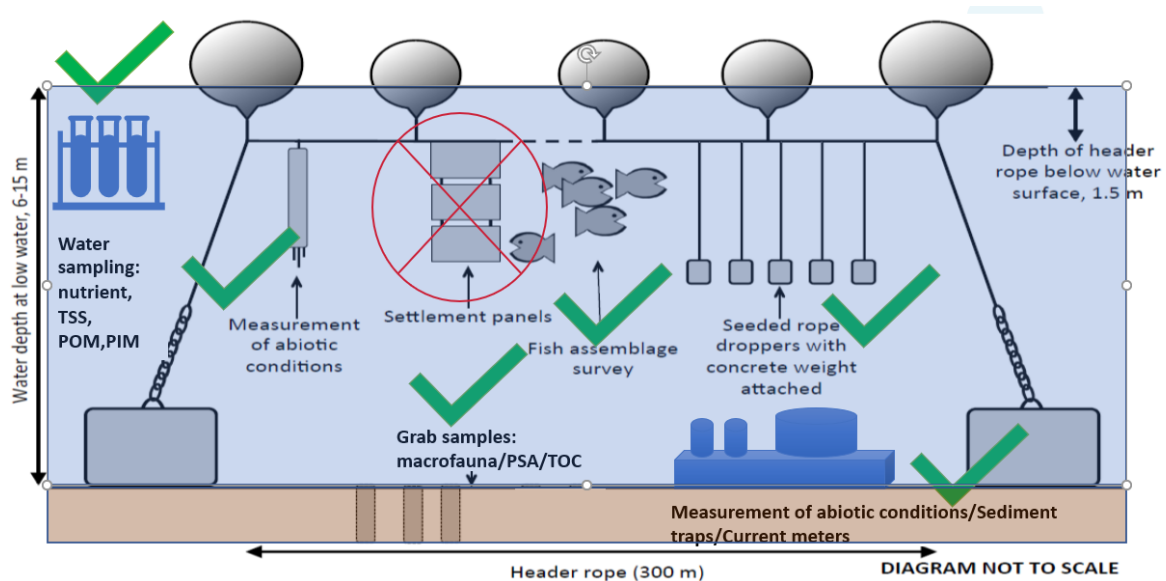


Figure 2 Schematic representation of the instrumentation used at the test sites to measure environmental effects of offshore aquaculture. Green tick marks represent studies conducted during the GENIALG project for which results are presented in this report. Deployment of settlement panels was not conducted.

2 Materials and methods

Study kelp species

The two farmed kelp species considered in this study were *Saccharina latissima* and *Alaria esculenta*, representative photographs of both species are displayed in Figure 3.

Saccharina latissima (L.) is a brown alga of 2-5 m length with a <60 cm stipe and an undivided, “elongated frilly-edged, crinkled fronds” (Morrissey et al. 2001). Present in polar and temperate waters in the north Pacific and Atlantic oceans, *S. latissima* is native to Europe (Peteiro and Freire 2012), occurring in a broad range of habitats, under a wide spectrum of light and temperature conditions (Heinrich et al. 2012). The species prefers sheltered rocky habitats and is usually absent from very wave-exposed areas (Morrissey et al. 2001). Also known as ‘sugar kelp’, *S. latissima* is currently used for direct human consumption (Peteiro and Freire 2013; Lüning and Mortensen 2015), as animal feed for aquaculture (Troell et al. 2006), and a species in integrated multi-trophic aquaculture (Freitas et al. 2016). Wild harvesting of *S. latissima* occurs in several European countries (e.g. Rodríguez González and Tasende 2003). However, natural stocks are limited and populations around Atlantic European coasts have declined drastically in recent years (Pehlke and Bartsch 2008; Bekkby and Moy 2011; Christie et al. 2019; Sogn Andersen et al. 2019).

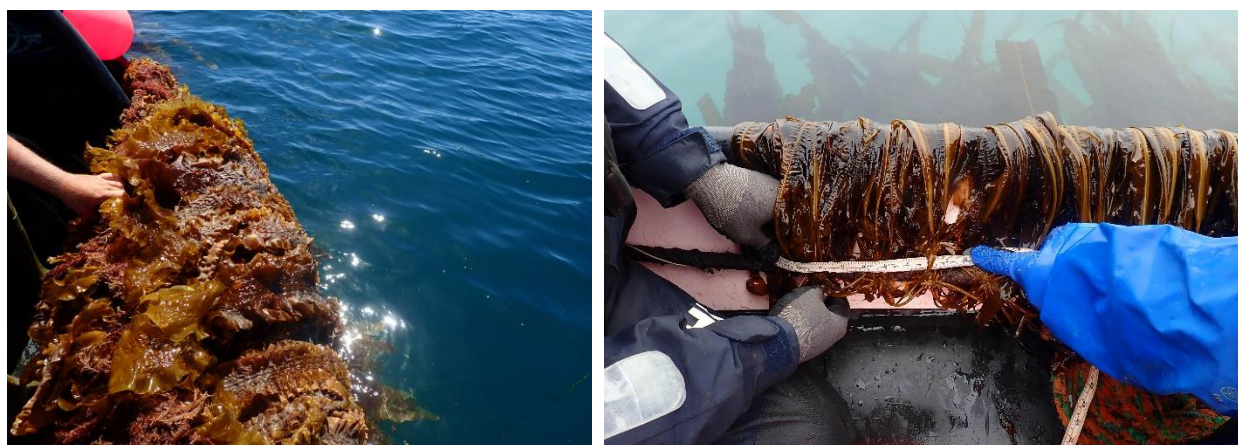


Figure 3 Cultivated sugar kelp *Saccharina latissima* (left) and Irish wakame *Alaria esculenta* (right). Photographs taken at the test farm located in Ventry Harbour (Ireland) between 2017 and 2019.

Alaria esculenta (L.) Greville is a 1-4 m length brown alga with an upright and short stipe, and a long single blade with a yellow visible midrib which helps to distinguish it from *S. latissima* juveniles (Fredersdorf et al. 2009). The distribution of *A. esculenta* spans sublittoral zones in the Northern Hemisphere, in exposed, rocky coasts in both the Pacific and Atlantic oceans (Hession et al. 1998;

Kraan et al. 2001). The distributional range is largely limited by the species water temperature mortality threshold of 16°C (Hiscock et al. 2000; Fredersdorf et al. 2009). Offshore cultivation of *A. esculenta* using longlines is usually preferred to harvesting of wild populations because of their ability to attach well to the ropes and ease of harvest (Morrissey et al. 2001).

Seaweed farms used as test sites

2.1.1 Ventry Harbour

The aim of the study was to evaluate the impact that a test farm has on the environment by collating in-situ observational data (biotic and abiotic) for the duration of the project, that is at least two growing seasons, i.e. 2017/2018 and 2018/2019. This study was carried out in a seaweed farm owned and operated by Dingle Bay Seaweed Ltd. The site is located off the south-west coast of Ireland, in Ventry Harbour, Co. Kerry (52° 06' 49.45" N, -10° 21' 20.17" W). The 900 x 200 m farm is located on the south westerly side of the harbour, laying on a north-west to south-east orientation and is accessible from Cuan Pier (Figure 4; Figure 6).

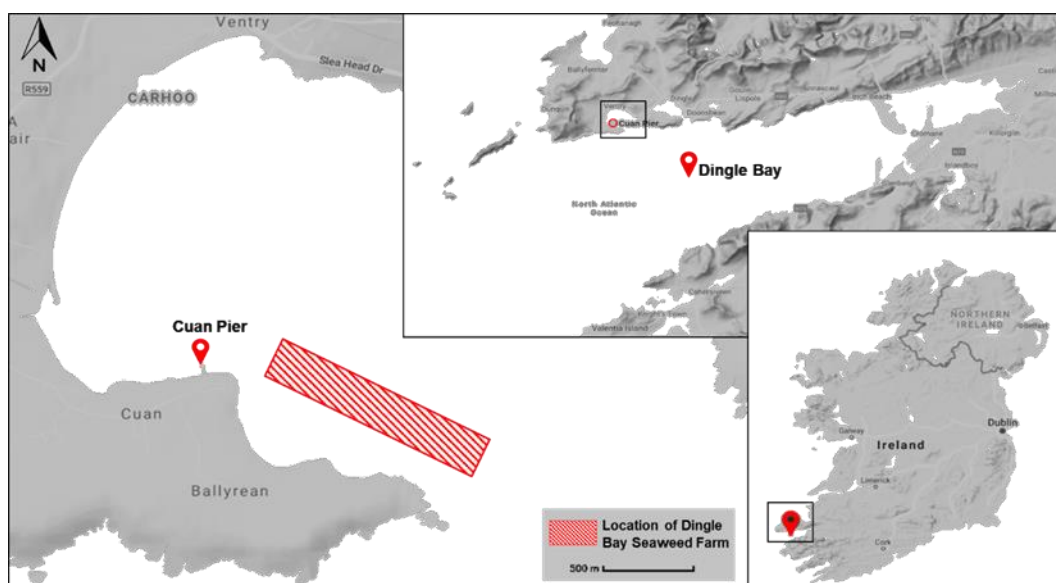


Figure 4 Map of Ireland showing the seaweed farm used as a test site to study the environmental footprint of seaweed cultivation between 2017 and 2019. The farm is in Ventry Harbour, Co. Kerry and is operated by Dingle Bay Seaweed Ltd.

The site is fully saline with mean salinity recorded during the study of 32 ppt, and bottom water temperatures ranging between 6.10 and 17.67 °C in February and July, respectively (Figure 5). Water depth within the boundaries of the licensed aquaculture site ranges between 6 m (CD) in its western limit (inner harbour) to ca. 20 m in the eastern, most exposed limit. The site is generally sheltered from significant wave action, but it is subjected to considerable bottom swell and turbulence (pers. obs. Jose M. Fariñas-Franco and Mike Murphy). The seafloor is primarily sandy with seagrass *Zostera*

marina meadows located in the southwest of the bay, under and in the vicinity of the farm. This seagrass dominated area is unusual for its depth, as it extends from the shoreline between Cuan Pier and Ballyrean in the south, to depths up to 13 m below chart Datum to the north and east of the farm, making it the deepest occurrence of *Zostera marina* in Europe (Whelan and Cullinane 1985). There are two protected areas adjacent to Ventry Harbour and Ventry Dunes and Parkmore Point proposed Natural Heritage Areas and the shoreline of the Kilfarnoge Peninsula within the Dingle Bay Spatial Protection Area. The bay, where the seaweed farm is located, is not a designated Natura 2000 site.

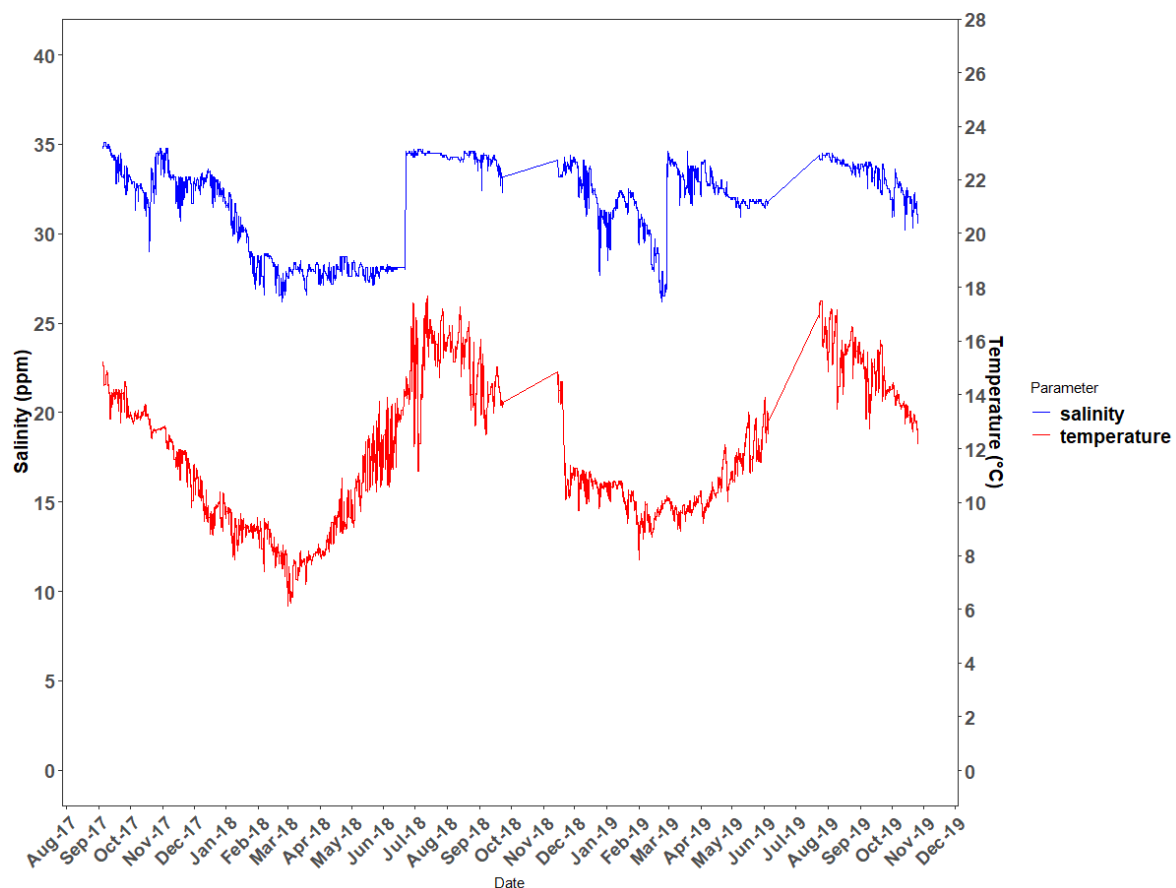


Figure 5 Salinity (PPM) and water temperature (°C) recorded under the test farm in Ventry Harbour, Co. Kerry (Ireland) during the 2017-2019 environmental study period.

The farm is divided into three sections, or 'blocks', each with their own anchoring ropes. In each section approximately 15 lines are usually deployed on a southeast-northwest direction, same as the licenced site. During the cultivation period, block 1 of the seaweed farm was utilised. The block consisted of 15 ropes, 200 m in length and 10 mm in diameter, separated approximately 15 m from each other. These longlines (or header ropes) usually sit 1.5 m below the water surface without seaweed and up to 5 m deep at peak biomass and are kept in position by weight blocks and buoys spaced every 10 m along the longlines. This layout is similar to other offshore algal culture set ups (see best practice advice from EnAlgae project in Mooney-McAuley et al. 2016). Two longlines were chosen

in the outer edges of the licensed site, Line 1 in the northern section and Line 15 to the south, the closest to the shore (Figure 6). In addition, Line 7 located in the centre of the site, was also selected for cultivation. The ‘edge vs centre’ layout was regarded as representative of the different hydrodynamic conditions in the site and accounted for potential spatial variability caused by shading, nutrient availability and current disruption caused by nearby longlines. The rest of the site is occupied by longlines cultivated with *A. esculenta*.

In total 20 collectors (1.7 mm Dyneema yarn, Schappe Technologies) were seeded with *S. latissima* sporophytes and deployed at the study site in Ventry Harbour on 9th December 2017 and 22nd October 2018 (two growing seasons). For the second season the collectors were seeded with *S. latissima* gametophytes originating from wild stocks, i.e. collected from the shore in Ventry Harbour. The collectors were deployed on the same longlines on both seasons (see Figure 6). Subsequent growth of the sporophytes was visually inspected every 3-4 weeks as part of the environmental monitoring survey work for WP6. Unfortunately, the cultivation of *S. latissima* in the test farm yielded low sporophyte densities during this sampling period in 2019 (3-4 individuals per meter), likely due to poor gametophyte growth or contamination prior or during deployment. It is worth noting the collectors and longline were also heavily colonised by filamentous red algae. A second deployment using three collectors directly seeded with *S. latissima* zoospores in December 2018, yielded the same poor sporophyte growth (Figure 2). The ecological assessment of seaweed farming continued using the long lines seeded with *A. esculenta* providing an opportunity to study potential species-specific differences in biodiversity and habitat creation hence generalising our conclusions.

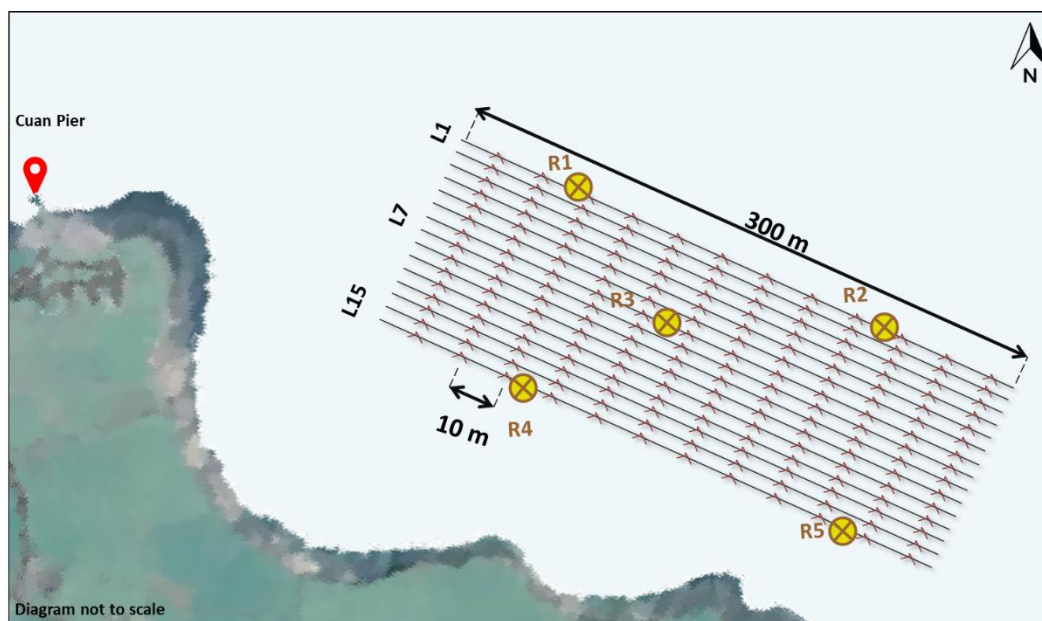


Figure 6 Schematic longline layout at Block 1, Dingle Bay Seaweed test farm in Ventry Harbour, co. Kerry (Ireland)

2.1.2 Schelphoek (Eastern Scheldt)

The commercial seaweed farm (Seaweed Harvest Holland) is located in Schelphoek near Schouwen Duiveland in the south-west of The Netherlands ($51^{\circ}41'29.2''\text{N}$ $3^{\circ}48'32.0''\text{E}$; Figure 7). The Schelphoek is a sheltered bay in the Eastern Scheldt, which is affected by tidal movement from the North Sea. The water is considered brackish as both salt water from the North Sea and fresh water from the Dutch rivers affect the Eastern Scheldt. The farm consists of 32 long lines of about 100 meters long that are kept afloat with buoys and separated with separators, and the size of the farm is approximately 1 hectare. Different methods were used varying from horizontal to vertical suspension, and from continuous culture ropes to single droppers. The water is approximately 7 meters deep around the farm.



Figure 7 Map of The Netherlands showing two seaweed farming locations in the Netherlands. [lower] Farm location 'Schelphoek', located in the Eastern Scheldt a semi enclosed estuarine area, which was investigated during the GENIALG project. [upper] The offshore farm location 'North Sea Innovation Lab' (NSIL) a site in the North Sea located at approx. 12 km out of the coast and investigated through a national research project (PROSEAWEEED).

Biological interactions

2.1.3 Longline biodiversity studies

Samples were collected to describe the biotic assemblage associated with cultivated seaweed during the growing season, from the early stages to pre-harvest peak biomass. This sampling was tied with

the biomass and growth monitoring work run in parallel to feed on ecological models developed as part of the GENIALG project. The samples were collected in 2018 from Lines 1 (n=2), 7 (n=1) and 15 (n=2) in April (two visits), May and June. In each visit, 25 cm of *S. latissima*, including the holdfast, were carefully removed from the longlines (see Figure 3 and 5). The samples were placed in plastic bags and stored in cool boxes during transport and frozen on return to the laboratory. The samples were later defrosted and processed for biodiversity and faunal community analysis. These longlines (lines 1, 7 and 15) were left unharvested over the summer, and further samples (n=5) were collected in October 2018, to quantify the biodiversity and changes in the faunal assemblage succession.

To provide more data to strengthen the statistical analyses, including meaningful comparisons and control for temporal and seasonal variability, a repeat survey was conducted during the 2018/19 algal growing season. Due to the unsuccessful cultivation of *S. latissima* in 2018, only *A. esculenta* was available. Thus, this study provided an opportunity to establish inter-specific differences in the biodiversity and faunal assemblage composition associated with the two most frequently cultivated kelp species in Ireland. The sampling approach replicated that of the 2017/18 season, collecting five 25 cm replicate samples of *A. esculenta* fronds and holdfast (Figure 8). The samples were collected in March, April, May, and June 2019 (one visit each of the four sampling timepoints). The first sampling was conducted earlier compared to the previous year (2018) to account for the higher water temperatures recorded in the spring of 2019 compared to the same period in 2018, collected on five occasions between March and June 2019. These samples (n=25) were preserved in ethanol and the fauna from Line 15, and the five replicates from the April survey, were identified.

In both cases, the aim was to describe the biotic assemblage associated during the growing season, from the early stages to pre-harvest peak biomass and use this opportunity to compare the assemblages and biodiversity associated with each cultivated kelp species.



Figure 8 Photograph showing collection of 25 cm *A. esculenta* longline samples for biodiversity analysis at Ventry Harbour farm, May 2019.

2.1.3.1 Sample processing and faunal identification

All samples were taken to the laboratory, weighed, and the number of individual kelp plants counted. The seaweed was then washed over a sieve (0.5 mm mesh size). All the material retained by the sieve and the seaweed itself were preserved in industrial methylated spirit (IMS), a mixture of 89% ethanol, 6% water and 5% methanol. All macrofauna were sorted under a stereoscope into four main taxonomic groups (Arthropoda, Mollusca, Annelida, and any other major phyla). Sample identification was carried out to the lowest possible taxonomic level e.g. species, using a stereoscope and an optical microscope when necessary. Groups such as nematodes, hydrozoans, bryozoans, and other minor or less common groups within the main phyla were identified to class, order, or family e.g. Ostracoda, Tanaidacea, or Halacaridae. All names were cross-checked using WoRMS (World Register of Marine Species, <http://marinespecies.org/>).

All the faunal taxa identified were organised in data matrices, and their abundances were arranged for statistical analysis. Colonial species, e.g. *Electra pilosa*, *Membranipora membranacea*, *Obelia* sp., and taxa serendipitously retained in the sieves (i.e. less than 0.5 mm in size) e.g. Copepoda and Nematoda, were only considered for calculating the total number of species in the community but not included in the statistical analyses.

2.1.3.2 Statistical analyses

Univariate and multivariate statistical analyses were used to investigate the differences in biodiversity indices and the composition of the faunal assemblages associated with each kelp species, as well as

their temporal variability. These analyses were carried out using the open-source software R (R Core Team, 2020). Graphs were created using the R package ggplot2 (Wickham, 2016).

Biodiversity indices included total abundance and total number of taxa in restricted taxonomic groups (Annelids, Arthropods and Mollusc). Total abundance of individuals (N), total number of taxa (S), Shannon–Wiener diversity index (H'), Margalef's species richness index (d) and Pielou's evenness index (J'). Data were checked for normality and homogeneity of variances. Parametric conditions were met, and Analysis of Covariance (ANCOVA) models were conducted to identify significant effects of kelp species and kelp biomass following correlation analysis between kelp biomass (g) and water temperature ($^{\circ}\text{C}$) as covariates. All tests of null hypotheses were assessed at significant threshold (α) of 0.05.

Multivariate analyses were conducted using the R package Vegan (Oksanen 2013) to identify any successional patterns in the macroinvertebrate assemblage associated with each cultivated algal species, and the species responsible for the differences between the groups. All counts were standardized by the total abundance of all individuals in that sample. A non-metric multidimensional scaling (nMDS) algorithm using the Bray-Curtis similarity coefficient was used to identify groupings between samples.

The relationship between the observed macroinvertebrate community present in the kelp species and the independent environmental variables (water temperature, kelp biomass and the number of holdfasts present in each sample) was studied through a distance-based redundancy analysis (dbRDA) using Bray-Curtis similarity coefficient. Permutational MANOVA (PERMANOVA) fixed models were used to determine if the variability in community composition is explained by the various categorical variables (i.e. Kelp species, position within the farm and months). SIMPER (Similarity Percentage routine) analysis was performed in R (R Development Core Team 2020) to determine which specific species contributed the most to the differences between samples and to identify the dominant species in each month and kelp species.

2.1.4 Biodiversity of growing structures and cultivated seaweed: dropper field experiments

Experimental droppers were used to test the faunal colonisation patterns and biodiversity associated with cultivated seaweed and the ancillary support and growing structures. These droppers consisted of 1 m long polypropylene ropes that were attached to the farm longlines and kept in position using a small concrete weight (Figure 9). Some of these droppers were seeded with *S. latissima* gametophytes ($n=90$) and kept in recirculating seawater tanks under 12h/12h light/dark cycles for three months until

deployment. The treated 'seeded' droppers were sprayed in the hatchery with the same *S. latissima* strains used to seed the collectors and deployed for biomass harvesting and biodiversity studies. The same number of unseeded (i.e. not sprayed) controls (n=90) were prepared and submerged in sterilised seawater for at least 24 hours.

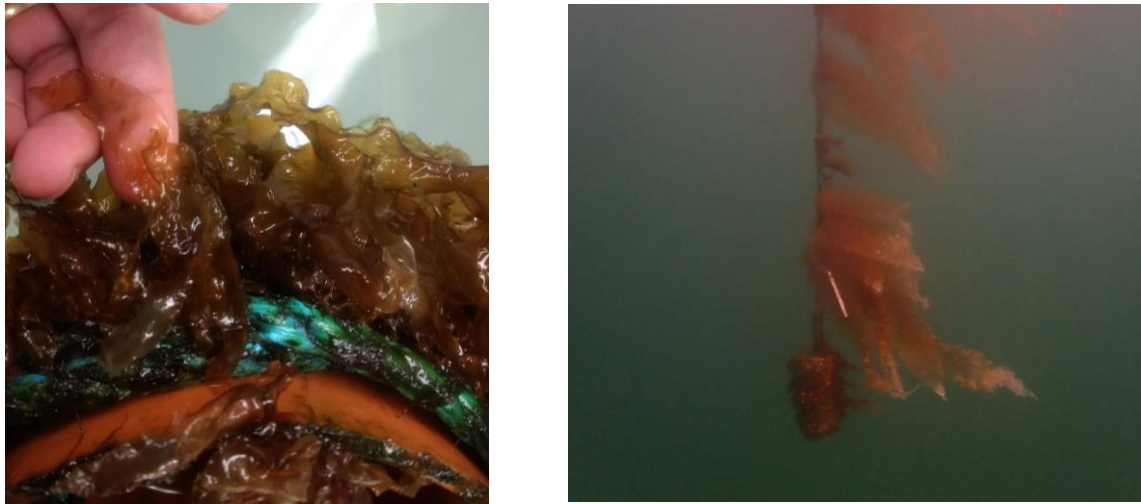


Figure 9 Seeded dropper in the hatchery before deployment (left) and attached to longline on test site, ca. two months after deployment in Ventry Harbour.

All droppers were deployed at the test farm in Ventry Harbour on 8th March 2017. To test the influence the position within the farm might have on colonisation and biodiversity, droppers were attached to Line 1, at one upcurrent location (exposed side of the farm, near the mouth of the harbour) and at the opposite end of the farm, in its most sheltered side to the west. This was repeated for lines 7 (centre) and 15 (southern edge). In total five replicate 1-m droppers, seeded (n=5) and unseeded (n=5), were attached to the header ropes at lines 1, 7 and 15 (see experimental layout in Figure 7). Six months

after deployment, three seeded replicates and unseeded controls were collected by scuba divers from all lines, in both up and down current positions (Figure 10 Figure 11).

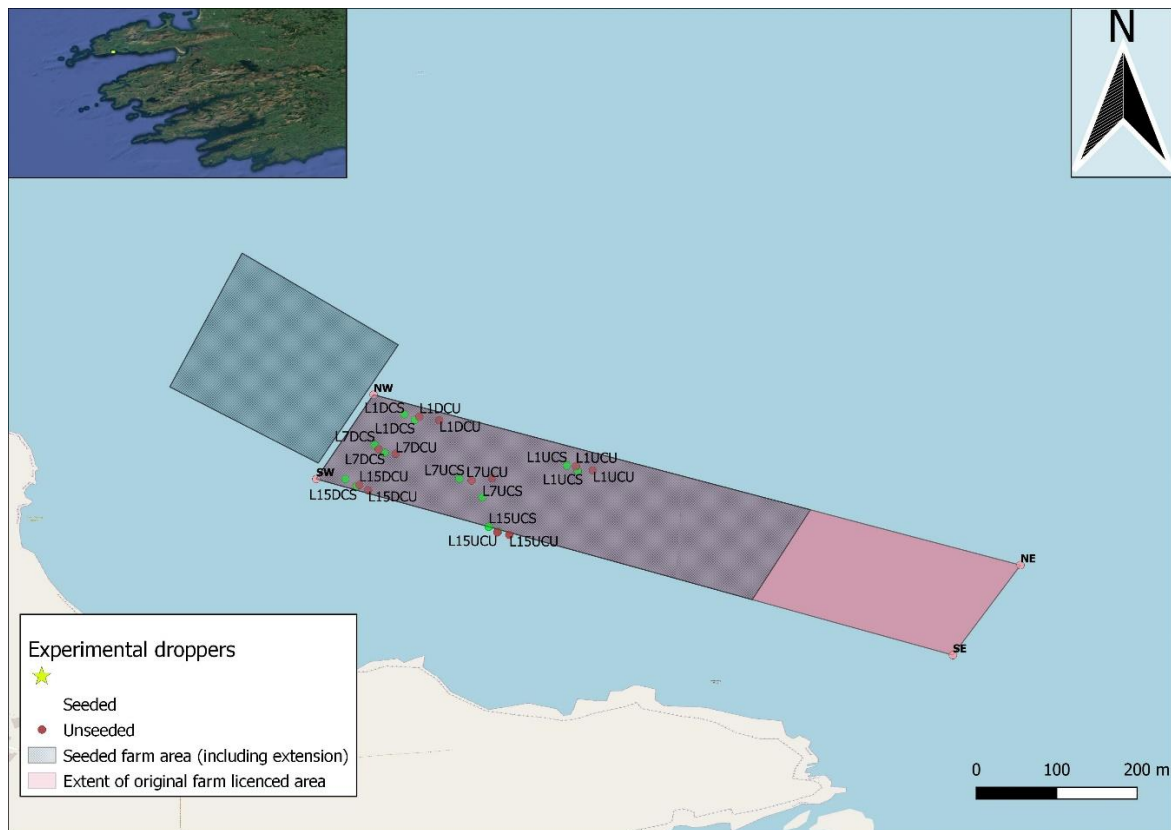


Figure 10 Map showing the position of the experimental droppers, seeded and unseeded controls, deployed in the Ventry Harbour test farm to assess biodiversity and colonisation patterns associated with seaweed cultivation structures. Map created using qGIS v3.16.

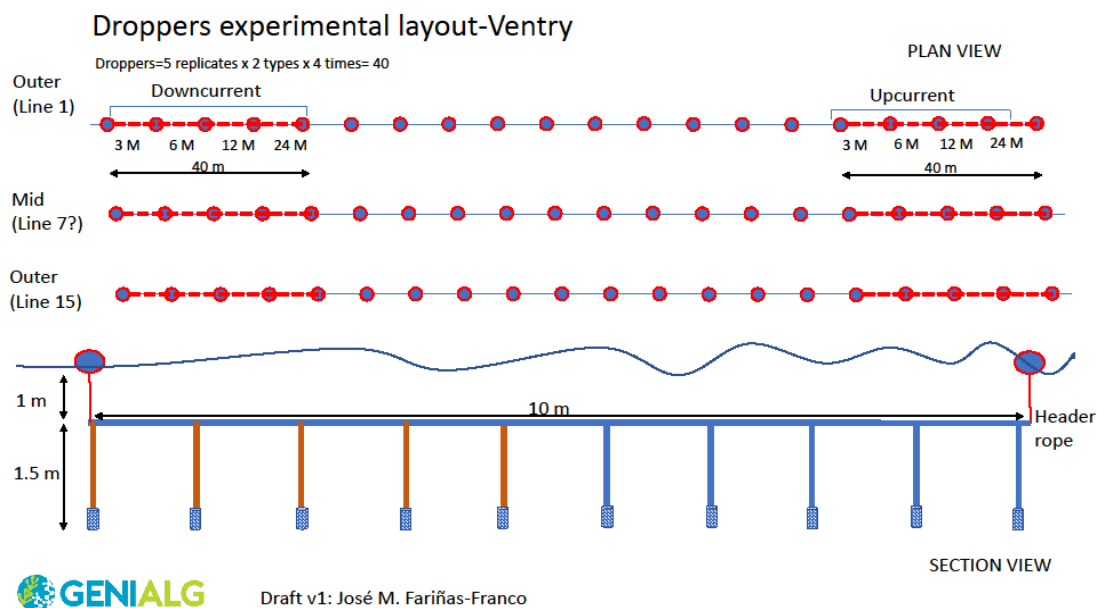


Figure 11 Layout of dropper experiment conducted at Ventry Harbour site, between March and October 2018.

The divers placed the droppers in 0.5 mm mesh size bags, tied both ends with cable ties and cut the dropper ropes. In total thirty-six replicates ($n=3 \times 2 \times 3 \times 2=36$) were collected by in June/July 2018, representing the six-month faunal succession and again in October 2018 ($n=12$) nine months after deployment for faunistic analysis. These samples were preserved in ethanol until they could be processed, and the fauna identified and enumerated. All 6-month dropper samples were sorted in 2019 into four main taxonomic groups after washing the samples over a sieve (0.5 mm mesh size), with full identification to species level and enumeration completed for lines 1 and 15. Faunal datasets were uploaded to the EMODnet public data repository (<https://www.emodnet-ingestion.eu/>).

The replicate droppers left on site over the summer season were collected by divers nine months after deployment, in October 2018 using the approach followed in July. The samples were preserved in IMS upon return to the lab. These samples were only partially sorted before the project came to an end. All statistical analyses were conducted using R (R Development Core Team 2020) including the R package Vegan (Oksanen 2013).

2.1.5 Effects on benthic primary producers and benthic biodiversity

To assess the potential effects from the farm on the benthos we followed survey methodologies laid out by Davies et al. (2001), informed by the BACI principle (i.e. Before-After-Control-Impact; Gray et al. 2006; Underwood 1991). BACI aims at differentiating natural variability in the measured parameters against changes potentially induced from an anthropogenic activity (as the source of impact) by sampling before and after the activity occurs, the area potentially impacted and distant control stations of similar characteristics but unaffected by the activity. In the design, 100 m transects were plotted along the direction of the prevalent current and perpendicularly to it, with the farm as the starting point. The layout aimed at detecting spatial changes in the sedimentology and infaunal diversity, from the farm as the source of impact, having a distant control transect to the north for comparison. These surveys consisted of underwater habitat surveys, conducted by SCUBA divers using video recording and still photography, and direct seabed sediment sampling by means of grab sampling from a boat. These radial sampling locations were positioned as follows: Down current (DC) located on the western section of the farm; Up current (UC) on the eastern, more exposed boundary; Orthogonal North (OT-N), perpendicular to the farm, on its northern side; Orthogonal South (OT-S), perpendicular to the farm on its southern edge; and Control located approximately 200 m to the north of the farm (Figure 12).

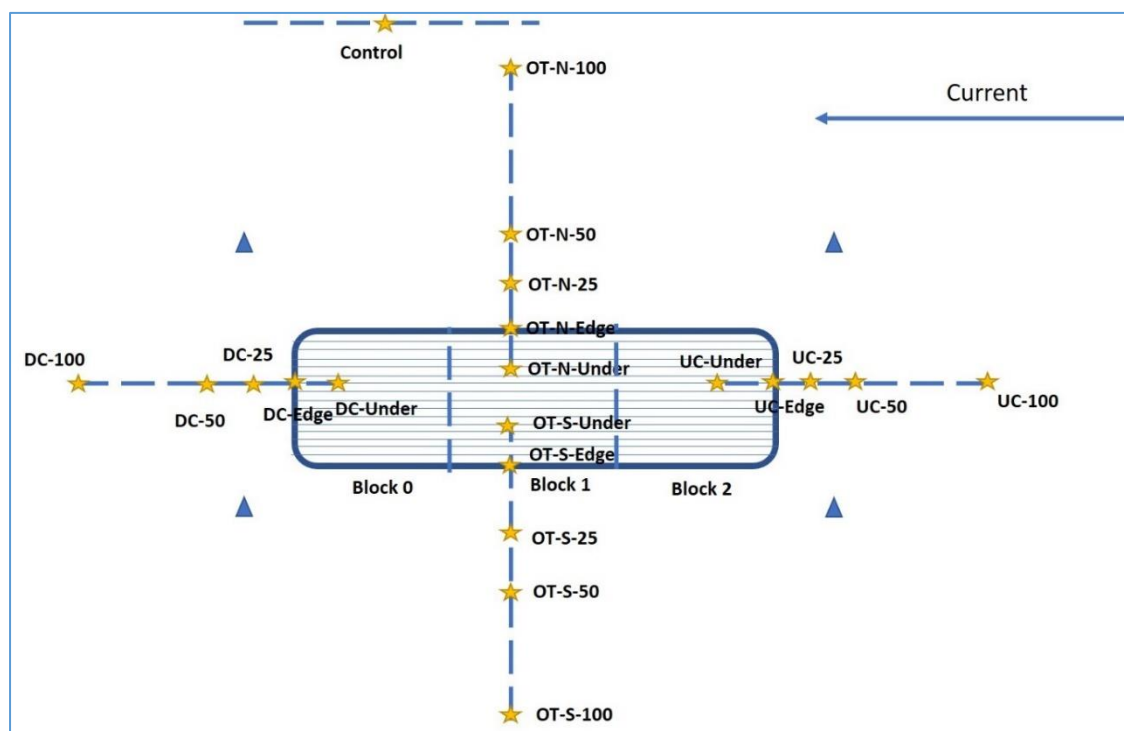


Figure 12 Schematic representation of the seabed surveys conducted at the test farms for GENIALG. In blue seabed transects surveyed by SCUBA divers using photo quadrats. Yellow stars represent grab sampling locations. Codes indicate positions: OT-N=Orthogonal North; OT-S=Orthogonal South; DC=Down current; UC=Up current. Numbers denote distance from edge of farm. Transects are 100 m long. Control station is situated approximately 200 m from the edge of the farm.

In total five benthic habitat surveys were conducted for GENIALG in two seasons (2017/18 and 2018/19). These surveys included a baseline survey conducted on 5th September 2017, prior to the deployment of culture string, mid-season surveys conducted on 28/29th March 2018, peak biomass surveys conducted at the time of the harvest on the 4th and 10th July 2018 and post-harvest/pre-seeding survey conducted on 24th September 2018. During the 2018/19 growing season, one additional benthic survey was conducted just before harvesting, representing the last seabed survey of the project. Grab samples for infaunal analysis and sedimentology were collected on 11th April 2019 while underwater habitat monitoring surveys (video and photography) were carried out in the first week of May 2019.

Each benthic survey consisted of:

- a. Five 100 m dive transects surveyed by a diver using photo quadrats and video (more than 100 dives, over 500 photographs collected). During the seafloor habitat surveys, SCUBA divers swam along the transects laid out from a boat using a leaded rope with two buoys attached to mark the beginning and end of the transects. The diver recorded notes on sediment and main habitat type, presence of conspicuous fauna or signs of their presence (e.g. burrows). The diver deployed a 0.25 m² quadrat at random positions starting directly under the farm, at

the edge and every 25 m (e.g. under, edge, 25 m, 50 m, 75 m, and 100 m). These quadrats were photographed using an Olympus® TG5® digital camera in an IKELITE® housing and video recorded using a GoPro Hero 5® camera mounted on the frame. Images were processed in the laboratory, quantifying percentage cover of seagrass, presence, and abundance of epifauna and general characteristics of the habitat and substratum.

- b. Sediment samples collected at each of the 21 stations displayed in Figure 12 using a Van Veen grab for macrofaunal analysis. The sample sorting has been completed for all samples and identification is ongoing; In total 315 sediment samples were collected at the test site, 210 for infauna analysis and 105 for granulometry/organic matter analysis. Infauna from all 210 samples were removed by sieving the sample through a sieve of 0.5 mm mesh diameter and sorting the macrofauna into four main taxonomic groups (Annelida, Crustacea, Mollusca, and other phyla), later identified to species level.
- c. Any seagrass present in the replicate grab samples was retained. All seagrass samples were weighed, dried and the dry weight and ash free weight calculated. The analysis for all five seagrass surveys (21 stations x 5 sampling points=105 samples in total) supplemented the coverage monitoring conducted using photo quadrats.

A replication of this methodology was tested in May 2018 at the Easter Schelde site. Dr. Jose M. Fariñas-Franco travelled to Yerseke (The Netherlands) to meet with Dr. Henrice Jansen (WUR) (authors in this report) and conduct a trial run of the ecological monitoring work at the site owned by Seaweed Harvest Holland. The fieldwork was conducted between 7th and 10th May 2018 and included:

- Seabed habitat video surveys (x5).
- UVC surveys along outer longlines.
- Deployment of a baited underwater camera.
- Collection of sediment samples at 21 representative stations.

2.1.6 Fish surveys: underwater visual census (UVC) and Remote Baited Cameras (BRUCs)

The use of farm structures and cultivated seaweed by fish species was assessed by means of underwater visual census surveys by divers (Tuya et al. 2005; Murphy and Jenkins 2010; Bosch et al. 2017). These fish surveys were conducted at the Ventry Harbour farm at three timepoints during the 2017/18 and 2018/19 seasons, namely: 1) before the deployment of the seeded material, 2) at mid-season (February/March), and 3) at peak biomass, just before harvesting. In each survey, a SCUBA

diver swam along three 30 m replicate transects on three different seaweed lines: two outer lines, north (Line 1) and south (Line 15) and the centre line (Line 7). Video footage was recorded using a HD GoPro® Hero 5® digital camera, taking photographs and notes on fish species present, their numbers and behaviour if possible. Surveys began at one of the longlines marking the north or southern boundaries (e.g. the outer longlines, Line 1 or Line 15). The diver started the first transect at one of the header buoys marking the edges of the farm against the direction of the prevailing current, videoing the longlines and swimming in a controlled, steady manner to ensure fish were not disturbed.

Using the header buoys as reference, and knowing they were approximately 10 m apart from each other, the diver stopped the video recording upon reaching the third buoy (i.e. 30 m transect length). To avoid any potential confounding effects the diver continued swimming at the same pace up until the following buoy, when they started recording the second transect and, subsequently, the third and last transect. The diver was retrieved onto the boat and the process repeated in the remaining two longlines. Control underwater transects were also surveyed on nearby seagrass/kelp habitats were also collected in June 2018 and January 2019, respectively. While in 2018 the three transects were carried out along longline cultivated with *S. latissima*, as expected, in 2019 the failure of the experimental *S. latissima* crop meant that only *A. esculenta* were available for the follow-up surveys. Similar surveys were trialled in the Schelphoek (Eastern Schelde) farm run by Seaweed Harvest Holland carried out by WUR and UOG (this report).

Underwater baited cameras were successfully tested in both Ventry Harbour (UOG) and Schelphoek (WUR). Qualitative and quantitative “one-off” data were recorded and are available in the results section.

Monitoring of environmental conditions

Continuous monitoring of abiotic parameters was conducted at the test farm in Ventry Harbour from September 2017 to October 2019. All data loggers were regularly checked, cleaned and the data downloaded to provide a detailed image of the conditions on site, namely temperature, salinity, turbidity, irradiance, tidal current speed and direction, and water quality. In addition, sediments were sampled at regular intervals, in parallel with the benthic infauna surveys, for granulometric and organic matter content analyses. These data were shared with SINTEF in Norway to inform their ecological models developed as part of work package 6.2 of the GENIALG project (Broch et al. 2021).

2.1.7 Effects on seafloor physical properties: granulometry, organic matter content

Seabed sediment samples were collected from a boat using a small, 2 l (0.025m²) Van Veen grab. Sampling stations were the same ones sampled for benthic fauna characterisation: control, under the farm (U), at the edge of the farm (E), and at 25, 50 and 100 meters from the edge of the farm taken along four transects (up current, down current, orthogonal north and south) and are indicated in Section 2.1.5 and Figure 12. All samples were kept in a cool box while on transit to the laboratory and frozen (-20°C). Granulometric analysis was carried out using a Malvern® Mastersizer® particle analyser on three sub samples from sediment sample (Forde et al. 2012). Raw data was processed to obtain grain size distribution data using the software GRADISTAT v8.0 (Blott and Pye 2001).

Total organic matter content in each sample was calculated using the loss of ignition (LOI) method. Samples were dried at 105°C for 24 hours, and the dry weight calculated before being incinerated at 550°C for another 24 hours. The organic matter content was then calculated as the difference between the dry weight and the ash free weight.

Data were plotted using QGIS 3.4.10 (QGIS Development Team 2019) to obtain a spatial visualization of the variation in sediment grain size and organic matter composition. The effect of season, location and distance on sediment composition and granulometry was carried out using PERMANOVA (Anderson 2005) in R (R Development Core Team 2020) using the vegan package (Oksanen 2013).

2.1.8 Hydrodynamics and sediment dynamics

A two-dimensional hydrodynamic model was constructed to assess the impact of the test farm on the hydrodynamics of Ventry Harbour. A Depth Integrated Velocity and Solute Transport model (DIVAST) was used with and without the seaweed incorporating water depths, current direction, and current velocities. The model only assessed the impact of the farm on fluid flow but could be later updated to incorporate biological and nutrient parameters.

To calibrate the model, two acoustic doppler profilers (current meters) and turbidity meters were successfully deployed and retrieved in April 2018 (pre-harvest) and July 2018 (post-harvest) at two sites (under the farm and between the shore and farm). This work was conducted in collaboration with the School of Earth and Ocean Sciences (University of Galway). A separate report with the results of the DIVAST 2-dimensional model is in preparation.

Sediment traps were used to study potential alterations to the siltation dynamics in the area caused by the presence of the farm. In total three set of sediment traps were deployed under the farm and

at the two control stations for a period of 7 to 12 weeks in January/February 2019, June/July 2019, and July/October 2019. The results from the first two deployments yielded significantly higher sediment deposition outside the farm compared to under it. The farm acts as a sediment trap itself, reducing turbidity and sedimentation.

2.1.9 Turbidity and light penetration (Irradiance)

Data loggers measuring temperature (HOBO®, Onset Ltd.), photosynthetically active irradiance (PAR; Odyssey® Dataflow Systems Ltd.), and conductivity (Seabird Scientific Ltd, SBE-37 MicroCAT®) were deployed at the Ventry site in September 2017. The temperature and PAR loggers were attached to stainless steel frames (Figure 13) and deployed on the seafloor at two control stations ca. 200 m north and ca. 100m south of the farm, between the farm and the shoreline. A third frame with PAR, temperature and conductivity loggers was placed on the seafloor, under the centre of the farm. Finally, one PAR and temperature logger were attached to one of the longlines in the centre of the farm, just below the sea surface. The loggers were programmed to take data every five or ten minutes for the duration of two full seasons (Sept-17 to Oct-19) and were regularly retrieved to download the data and ensure their maintenance. In addition, turbidity loggers (RBR Ltd.) were deployed in July 2018, before the start of the second season in October 2018, and retrieved in October 2019.

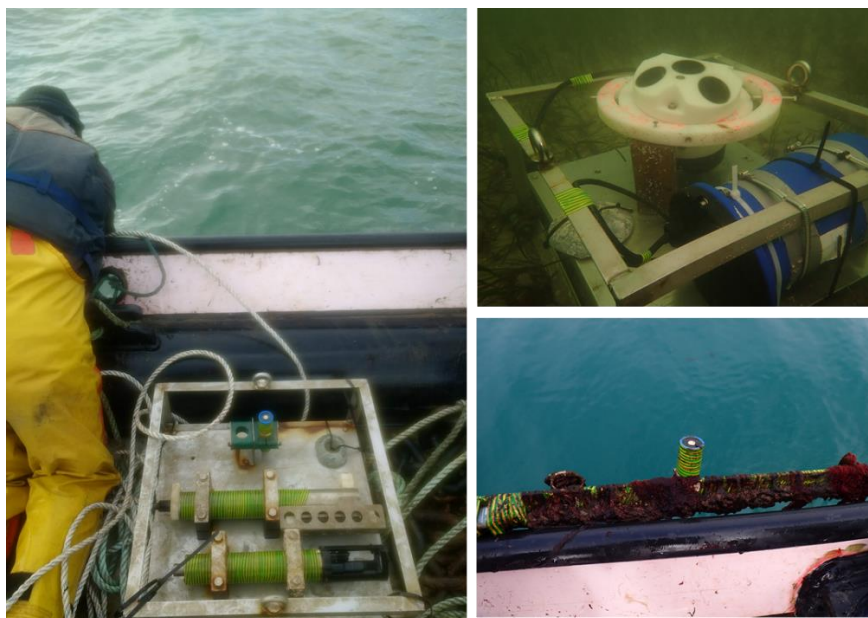


Figure 13 Instrumentation used to monitor abiotic conditions at the test farm in Ventry Harbour: left, Seabird conductimeter, RBR turbidity logger and PAR logger deployed under the farm; top right: acoustic doppler profiler use to measure direction and intensity of tidal currents; bottom right, Odyssey PAR logger attached to header rope, use to measure irradiance.

2.1.10 Water quality: visibility, suspended matter, and nutrient content

Water samples were collected monthly at the Ventry test farm between September 2017 and July 2019. The sampling was conducted from a boat at five sampling stations, namely at the centre of the farm, at two stations located 100 m east and west from the edge of the farm (i.e. 'up current' and 'down current'), and at two distant stations perpendicular to the prevalent current direction, i.e. one ~200 m to the north (named control), and ~1 km to the western edge of the farm, and at the inner part of Ventry Harbour close to Fionn Tra beach (named Inner Bay). See Figure 14.

In each sampling station, triplicate water samples were collected at the subsurface (ca. 10 cm deep) by hand using 500 ml acid washed, High Density Polyethylene (HDPE) bottles previously rinsed three times, in situ, with sea water. Three water samples were collected off the seabed (~1 m off bottom) from the boat using a Niskin Bottle (KC Denmark A/S®). The Niskin bottle was manually set, lowered to ca. 1 m off the seabed and closed using a messenger weight. Once hauled back into the boat, three 500 ml HDPE bottles (one for each replicate) were rinsed three times with the contents of the Niskin bottle before being filled. All the samples were placed into a cool box filled with ice packs and transported to the laboratory the same day. In total, 30 water samples were collected in each visit to the site (N=3 replicates x 2 depths (surface and bottom) x 5 sampling locations=30). In addition, water clarity was estimated in situ, on the day of the sampling, by means of a Secchi disk.

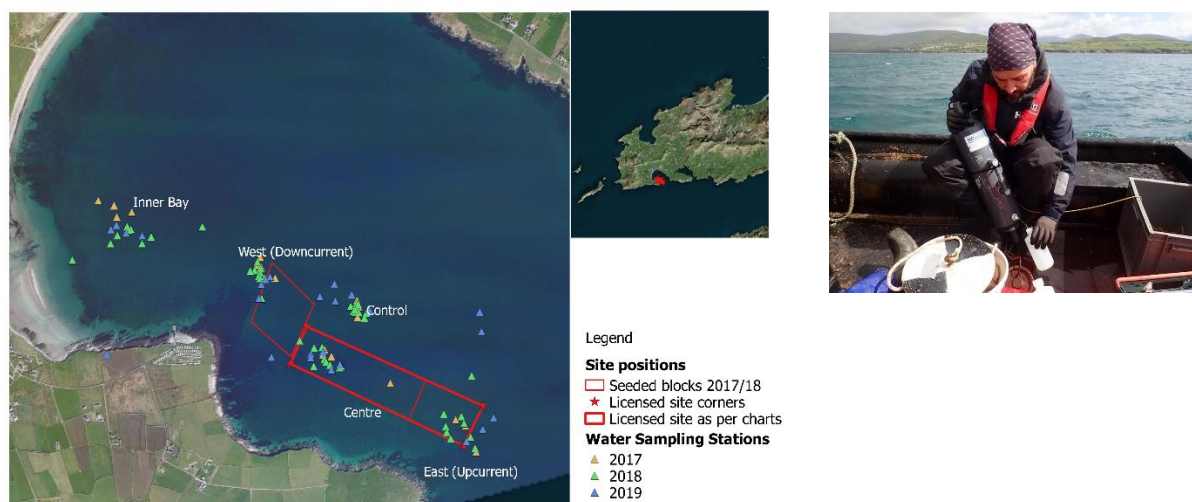


Figure 14 Location of water monitoring stations in Ventry Harbour, co. Kerry. Photograph: water samples collected using a Niskin bottle. A Secchi disc was used for water clarity measurements.

Upon return to the laboratory, all samples were frozen (-20°C) until processed for nutrient and suspended matter content analysis. Samples were fully defrosted in a constant temperature room at ~4°C before analyses were carried out. A 50 ml filtrate subsample was obtained using 0.45 µm pore

size syringe filter and kept for nutrient analyses. The remaining sample (~450 ml) was vacuum filtered through a Whatman glass microfiber filter (0.7 μm pore size). Once the full volume was filtered, the filters were oven dried at 105°C for 24 hours and weighed to estimate total suspended solid (TSS) content, and then ignited to constant weight at 550°C using loss on ignition (LOI) method to calculate the inorganic and organic fractions of the TSS content. Nutrient analyses were conducted using a microplate spectrophotometer following methodology laid-out in (Murphy and Riley 1962; Vargas-Albores and Hernandez-Lopez 2003; Patey et al. 2008; García-Robledo et al. 2014).

All statistical analyses were conducted using R (R Development Core Team 2020).

3 Results

Effect on biodiversity and surrounding habitats: Ventry Harbour test farm

3.1.1 Biodiversity and community assemblage of cultivated kelp

Observations made by divers and visual records of epifauna and colonial taxa found on the fronds during the sample sorting process (but not included in the quantitative analysis), indicated the presence of an epifaunal assemblage of bryozoans (*Electra pilosa*, *Membranipora membranosa*), Hydroids (*Obelia geniculata*), occasional snake locks anemones *Anemonia viridis*, sponges (*Sycon ciliatum*), along with tube building amphipods (*Jassa falcata*) on the fronds and in the holdfast crevices where sediment accumulated. These species represented the climax assemblage associated with the cultivated seaweed in the summer months. As the seaweed deteriorated and was grazed upon, the assemblage transitioned towards a blue mussel *Mytilus edulis* dominated one in October, with associated anemones (*Metridium senile*), hydroids and sea squirts (*Asciidiella aspersa*, *Corella parallelogramma*). The longlines and cultivated seaweed attracted several species of crustaceans including shrimp (*Crangon crangon*), squat lobster (*Galathea* spp.) and crabs (*Cancer pagurus* and *Necora puber*). From April to October high numbers of juvenile pollock *Pollachius pollachius* (shoaling in groups of 5-30 individuals), two-spotted gobies *Gobiusculus flavescens* and lumpsucker *Cyclopterus lumpus* were recorded.



Figure 15 Photographs taken by divers during underwater visual census surveys conducted at the Ventry Harbour farm, Left: Juvenile pollock (*Pollachius pollachius*) sheltering in cultivated *Saccharina latissima*; Right: Two-spotted goby (*Gobiusculus flavescens*) in *Alaria esculenta* longlines. Photo: Dr. Jose M. Fariñas-Franco.

3.1.1.1 Macroinvertebrate biodiversity in longline seaweed samples

In total 94 distinct macroinvertebrate taxa belonging to seven phyla were recorded across all 25 cm replicate samples collected in both seasons, in both kelp species. Out of that total, 78 taxa and 17,591 individuals were associated with *S. latissima* and 74 taxa and 8,934 individuals were found in cultivated *A. esculenta*. The mean number of invertebrates associated with each 25 cm section of cultivated longline was 3,840 individuals (153 ind. cm⁻¹ or 462 ind. kg⁻¹) for *S. latissima* and 5,221 individuals (209 ind. cm⁻¹ or 450 ind. kg⁻¹) for *A. esculenta*. There were 16 taxa exclusively found in *S. latissima*, and 20 taxa found only in *A. esculenta* e.g. *Monocorophium sextonae*. The most common species included the polychaete *Harmothoe viridis* (polychaete), the amphipods *Dexamine thea*, *Caprella mutica*, *Jassa falcata*, *J. herdmani*, *J. marmorata*, *Idotea granulosa* (arthropods) and the gastropod *Rissoa parva* (gastropod). Caprellid amphipods included two species, *C. penanti* and the non-native Japanese skeleton shrimp, *C. mutica*, was dominant with a total of 48 individuals per cm (1207 individuals) recorded on *S. latissima* in the peak biomass (June) surveys. The abundance of ghost shrimp in *A. esculenta* samples, taken also in June, was noticeably lower with 250 individuals per sample or 10 ind. cm⁻¹.

Truncated data for statistical analyses comprised 60 taxa belonging to seven phyla namely Annelida, Arthropoda, Bryozoa, Cnidaria, Mollusca, Nematoda and Porifera. The samples from *S. latissima* yielded 48 taxa while *A. esculenta* returned 50 taxa. Arthropoda was the phylum with greater diversity for both kelp species (40.8 taxa in *S. latissima* and 36.5 taxa in *A. esculenta*), followed by Mollusca (28 taxa) and Annelida (16.8 taxa). Annelida was the second most diverse group recorded in cultivated *A. esculenta* with 23 different taxa, followed by Mollusca with 19 taxa. Overall mean numbers of all taxa recorded were similar in both kelp species (23.81 in *S. latissima* and 22.25 in *A. esculenta*). Total number of taxa increased with time for both kelp species, except for May, where lower values were of Arthropods and Molluscs were identified in *A. esculenta* samples (Figure 16).

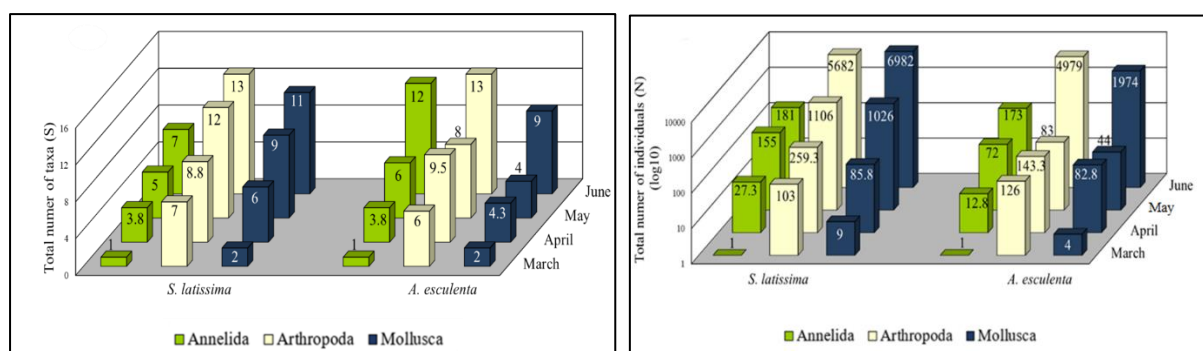


Figure 16 Bar plot showing, Left: Total number of taxa; Right: Total abundance (Logarithmic scale) in *Saccharina latissima* and *Alaria esculenta* among the sampling months.

Cumulative numbers of individuals (i.e. total abundance) also increased each month peaking in June, including numbers for those species that were consistently recorded in all months (Figure 16).

Shannon's diversity (H'), richness (d) and evenness (J) values of the macroinvertebrate assemblages were generally low, indicating high dominance of some taxa (e.g. *Jassa* spp.) and not remarkably different between the two cultivated kelp species (Figure 17). The samples were similarly rich (d) with mean values of 2.74 in *S. latissima* and 2.75 in *A. esculenta*. Mean Shannon's biodiversity (H') and evenness were low, albeit slightly higher in *A. esculenta* ($H' = 1.36$ and $J = 0.48$, respectively) compared to those recorded in *S. latissima* samples ($H' = 1.29$ and $J = 0.45$).

Richness (d) increased along the sampled period. There was an increase in H' index in the first three months and a decrease in the month prior to harvest (Table 6.4). J' index showed higher evenness in April and May for *S. latissima* (~0.5) but decreased to 0.32 in June (Figure 17; Table 1). However, evenness in the *A. esculenta* assemblage increased in the first months (with a peak of 0.69 in May) and decreased in June ($J=0.37$).

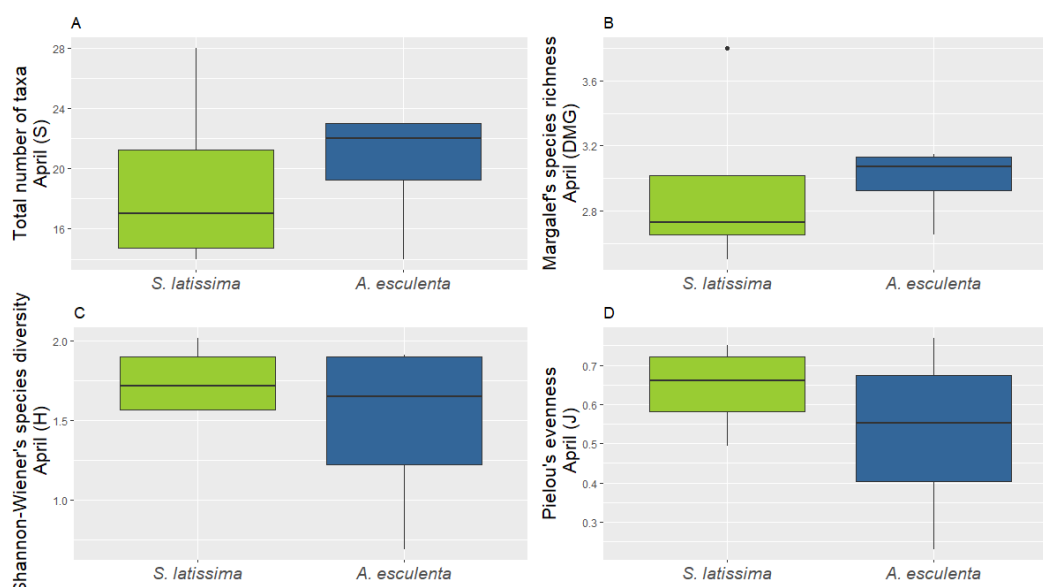


Figure 17 Boxplots of (A) total number of taxa (S); (B) Shannon–Wiener's species diversity (H'); (C) Margalef's species richness (d); and (D) Pielou's evenness (J') grouped by kelp species. The horizontal line within each box mark the median and the box portion of the box plot is defined by two lines at the 25th percentile and 75th percentile.

Table 1 Biodiversity indices associated with cultivated *Saccharina latissima* and *Alaria esculenta* samples. Kelp biomass (kg) represents kelp collected from 25cm samples. Total number of taxa (*S*), Abundance of individuals (*N*), Margalef's species richness index (*d*), Shannon–Wiener diversity index (*H*) and Pielou's Evenness index (*J*). April shows mean values (*n*=4).

Variable	<i>Saccharina latissima</i>				<i>Alaria esculenta</i>			
	March	April	May	June	March	April	May	June
Kelp biomass (Kg)	1.0	1.95	2.06	3.41	1	3.06	2.8	2.8
Total number of taxa (<i>S</i>)	10	20.25	29	36	11	19	21	38
Total abundance (<i>N</i>)	113	373.5	2290	12850	131	239.3	201	7128
Margalef's species richness (<i>d</i>)	1.69	2.99	3.10	3.17	1.44	2.94	3.02	3.61
Shannon–Wiener species diversity (<i>H</i>)	0.89	1.48	1.68	1.11	0.45	1.75	1.94	1.30
Pielou's evenness (<i>J</i>)	0.41	0.53	0.52	0.32	0.22	0.64	0.69	0.37

Linear regression models (see plots in Figure 18) showed significant ($p < 0.05$) collinearity between total number of taxa and temperature and kelp biomass as the independent variables with R^2 values of 0.61 and 0.8. The relationship between total abundance of individuals (*N*) and both explanatory variables was more tenuous, especially for biomass with a poor model fit ($R^2 = 0.49$; $p > 0.05$). There was no significant correlation between biodiversity (as total number of taxa *S*) and *N* and kelp biomass. However, temperature was positively, significantly correlated with both metrics, e.g. *S* $p = 0.8$; $p < 0.001$ *N* $p = 0.66$; $p < 0.01$.

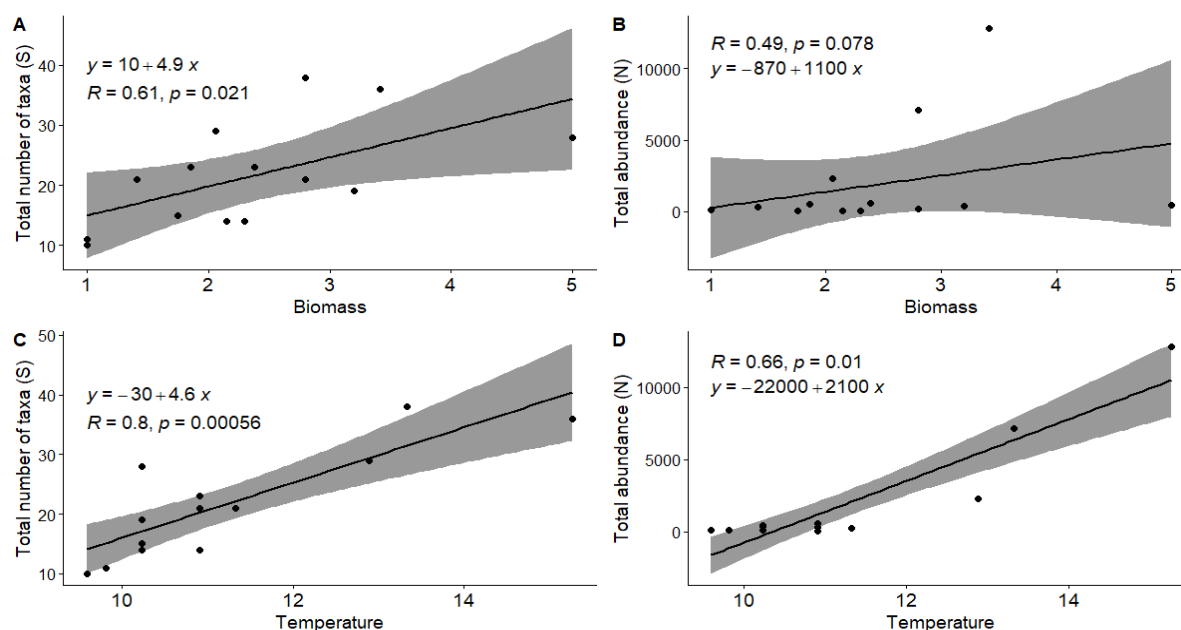


Figure 18 Scatterplot showing the linear regression between kelp biomass and water temperature with the Total number of taxa (S) and Total abundance (N). R^2 and its corresponding p -values are represented in the graph. Grey areas represent the confidence intervals.

3.1.1.2 Macroinvertebrate assemblage and succession in cultivated seaweed

Distance-based redundancy analysis performed with standardized data showed none of the explanatory variables (holdfast numbers, biomass and water temperature) had a significant impact on the samples and their community composition (Table 2).

Table 2 Summary of the dbRDA model relating the abundance matrix and the matrix of explanatory factors. The p -values are shown as indicators of the significance of the model, the CAP1 and CAP2 axes, and the independent variables. The proportion of variance explained by the CAP1 and CAP2 axes is also indicated.

	Df	SS	F	Pr(>F)
Model significance				
Model	3	0.47	1.07	0.4
Axes significance				
CAP1	1	0.28	1.93	0.46
CAP2	1	0.12	0.82	0.83
Explanatory variables				
Holdfast	1	0.12	0.84	0.53
Biomass	1	0.16	0.12	0.34
Water Temp.	1	0.18	1.25	0.25

PERMANOVA analysis results showed that kelp species and position within the farm, and their interaction, are not significant factors explaining community composition (Table 3). Position within

the farm was marginally not significant at ($p=0.07$) and the MDS plots shows a separation between assemblages found on exposed (outer) and sheltered (centre) farm longlines (Figure 19).

Table 3 Multivariate analysis of variance (PERMANOVA) showing the effect of each variable (Kelp species and Position) in the faunal assemblages associated with April samples. (Df=degrees of freedom; SS=sums of squares; MS=mean square).

Factor	Df	SS	MS	F.Model	R ²	Pr(>F)
Kelp species	1	47.30	47.30	1.12	0.16	0.20
Position	1	52.78	52.78	1.25	0.17	0.07
Kelp species * Position	1	32.26	32.26	0.76	0.11	0.97
Residuals	4	168.65	42.16	-	0.56	-

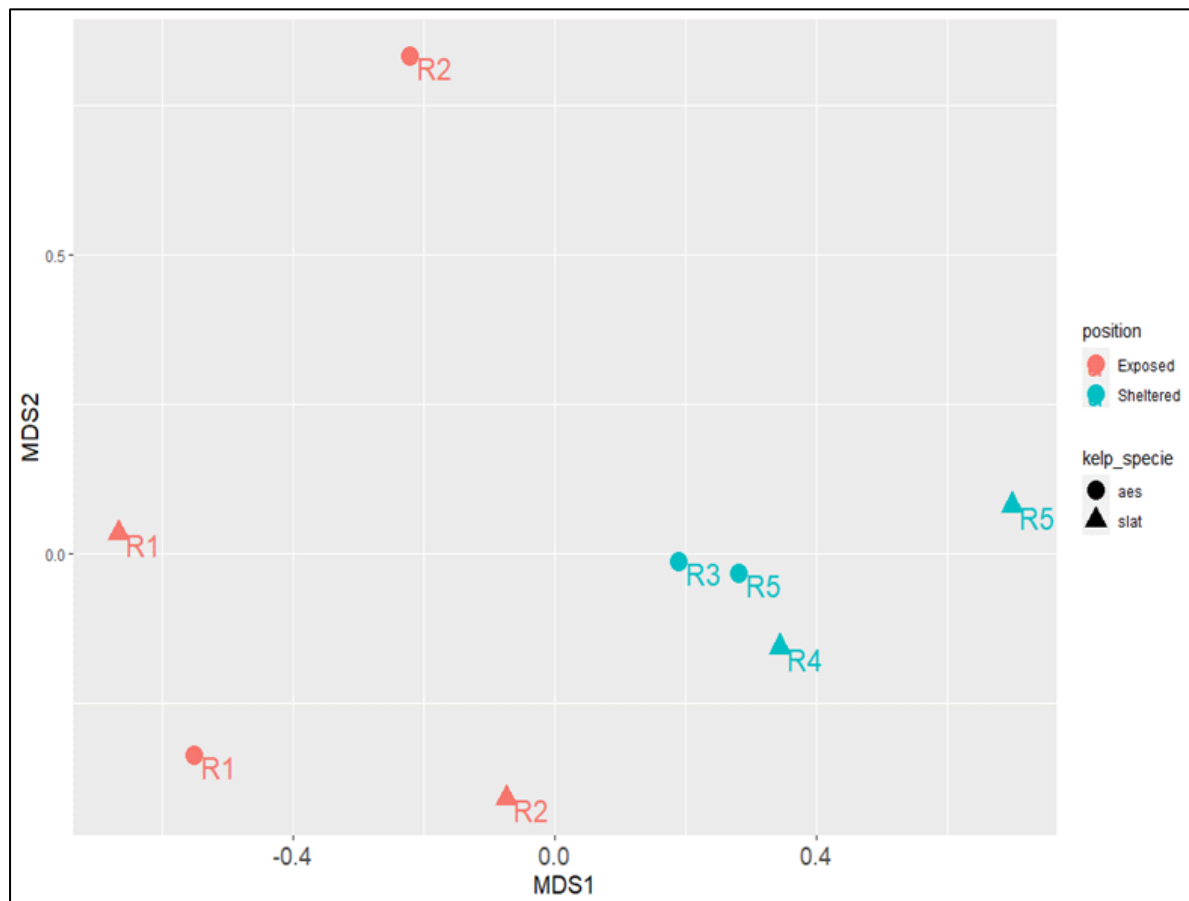


Figure 19 nMDS ordination graph based on Bray–Curtis similarity coefficient of standardized fauna data taken on April. The position of the samples in the farm is represented in different colours. *Alaria esculenta* (Aes) is represented by circles and *Saccharina latissima* (Slat) by triangles.

SIMPER analysis (Table 4) identified seven taxa being responsible for more than 70% of the variation in the assemblages between months, largely due to different abundances. Three taxa were Arthropods (namely *Caprella* spp., *Idotea* spp. and Ischyroceridae), three were Molluscs (*Fjordia lineata*, *Hiatella arctica*, and *Mytilus edulis*) and one Annelida (Oligochaeta). Differences in numbers of Ischyroceridae amphipods and *M. edulis* were consistently responsible for the bulk of the dissimilarities when comparing the March assemblage with subsequent months (up to 70% between the March and June samples). Other taxa included oligochaetes and the isopod *Idotea* sp. (March-April), and *Caprella* spp. (March-May). As diversity increased with time, new taxa explained the changes between these late Spring/early Summer communities (e.g. the bivalve *Hiatella arctica* and the nudibranch *Fjordia lineata*, see Figure 20).



Figure 20 The nudibranch *Fjordia lineata* feeding on hydroids *Tubularia indivisa* growing on longline structures used to cultivate kelp. Ventry Harbour test farm, co. Kerry (Ireland). Photo: Dr. Jose M. Fariñas-Franco.

Using kelp species grown as the grouping factor, SIMPER identified 46.61% dissimilarities, largely the result of taxa being present in both assemblages but with different abundances. Seven taxa that accounted for 70% of the differences between months, explained ~80% of cumulative contribution between kelp species. Ischyroceridae and *M. edulis* also contributed to most dissimilarities with more than 50% of the cumulative contribution. The biggest difference in average dissimilarity corresponded to Ischyroceridae, with 14.50%, and to *Mytilus edulis*, with 9.26%. The rest of the taxa presented lower and more even average dissimilarity (Table 5).

Table 4 SIMPER analysis comparing dissimilarities in community composition between months sampled. The taxa showed is contributing to the ~70% of the dissimilarity between two months.

	Average dissimilarity (%)	Average relative abundance Month 1	Average relative abundance Month 2	Cumulative contribution (%)
a) March vs April				
Ischyroceridae	16.78	0.856	0.521	43.06
<i>Mytilus edulis</i>	7.51	0.008	0.158	62.34
Oligochaeta	2.31	0	0.046	68.26
<i>Idotea</i> spp.	2.26	0.021	0.052	74.07
b) March vs May				
Ischyroceridae	32.08	0.856	0.215	44.01
<i>Mytilus edulis</i>	10.11	0.008	0.21	57.88
<i>Caprella</i> spp.	9.02	0	0.18	70.25
c) March vs June				
Ischyroceridae	16.8	0.856	0.52	36.65
<i>Mytilus edulis</i>	16.58	0.008	0.339	72.84
d) April vs May				
Ischyroceridae	15.3	0.521	0.215	28.88
<i>Caprella</i> spp.	9.02	0.006	0.18	45.89
Oligochaeta	7.34	0.046	0.147	59.74
<i>Mytilus edulis</i>	4.55	0.158	0.21	68.33
<i>Idotea</i> spp.	2.37	0.052	0.021	72.81
e) April vs June				
<i>Mytilus edulis</i>	9.72	0.158	0.339	28.05
Ischyroceridae	5.92	0.521	0.52	45.12
<i>Hiatella arctica</i>	3.12	0.065	0.003	54.13
<i>Idotea</i> spp.	2.42	0.052	0.006	61.1
Oligochaeta	2.29	0.046	0	67.7
<i>Fjorda lineata</i>	2.02	0	0.041	73.51
f) May vs June				
Ischyroceridae	15.29	0.215	0.52	28.26
<i>Mytilus edulis</i>	9.69	0.21	0.339	46.18
<i>Caprella</i> spp.	9.02	0.18	0.033	62.85
Oligochaeta	7.34	0.147	0	76.42

Table 5 SIMPER analysis results of *Saccharina latissima* vs *Alaria esculenta*. The taxa showed is contributing to the ~80% of the dissimilarity between both kelp species.

	Average dissimilarity (%)	Average relative abundance	Average relative abundance	Cumulative contribution (%)
		<i>S. latissima</i>	<i>A. esculenta</i>	
Ischyroceridae	14.50	0.489	0.567	31.11
<i>Mytilus edulis</i>	9.26	0.232	0.126	50.97
<i>Caprella</i> spp.	4.99	0.101	0.009	61.67
Oligochaeta	4.25	0.016	0.081	70.79
<i>Hiatella arctica</i>	2.09	0.022	0.049	75.26
<i>Idotea</i> spp.	1.77	0.009	0.041	79.06
<i>Fjordia lineata</i>	1.70	0.024	0.020	82.70

3.1.2 Biotic interactions: comparison between rope structures and cultivated seaweed

Sorting and identification of the macroinvertebrates found in the 1-m long droppers deployed in the Ventry Harbour test farm revealed high cumulative abundances of associated macrofauna after three months in the field (March – July) with a total 166,815 individual invertebrates belonging to 31 taxa. Faunal assemblages associated to seeded and unseeded droppers were dominated by the amphipod *Jassa* spp. (77,624 individuals), blue mussel spat *Mytilus edulis* (67,934 ind.) and the non-native Japanese ghost shrimp *Caprella mutica* (15,111 ind.). According to the results of the SIMPER analysis (Table 7), the main dissimilarities between the assemblages corresponded to much higher average abundances for the dominant species (*Jassa* spp., *M. edulis*, *C. mutica*, *Dexamine thea*, Copepods and *Rissoa parva*, representing 98% of the cumulative abundances) found in the droppers with *S. latissima* compared to the bare controls.

After four months since deployed in the field, droppers seeded with sugar kelp sporophytes yielded more than 128,747 individuals and 25 taxa while the total abundance in unseeded controls was 38,068 individuals belonging to 19 taxa. While biodiversity was higher in seeded droppers compared to controls, for all indices calculated, it was noticeably higher in the up current droppers compared to

the down current locations, except for evenness (J) (see Table 6 and Figure 21). Mean Margaleff's diversity (d) in seeded droppers was 1.04, slightly higher than unseeded droppers (0.91). Similarly, Shannon's diversity (H') in seeded droppers exceeded controls (1 and 0.75, respectively) while mean Pielou's evenness (J) was marginally higher in seeded droppers (0.43) compared to unseeded controls (0.40).

Seeded droppers located in the outer up current side of the farm yielded one order of magnitude higher abundance of macroinvertebrates compared to those droppers located in the western, down current section of the farm (104,507 ind. compared to 24,240 ind.). Similarly, unseeded control droppers in the up current position yielded 31,172 invertebrates compared to 6,896 individuals recorded in the down current section. Biodiversity indices were also higher in the up current locations for both seeded and unseeded droppers. However, evenness scores were higher in the down current droppers, seeded and unseeded (Figure 21).

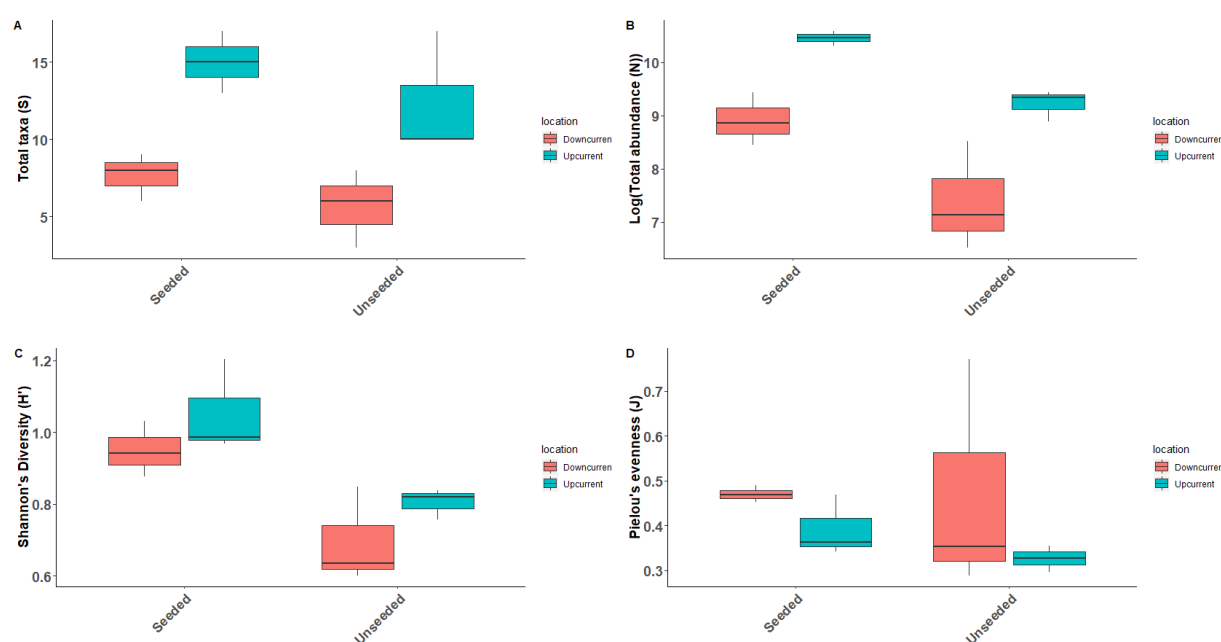


Figure 21 Boxplots for biodiversity recorded from seeded and unseeded droppers deployed at the Ventry test farm between March and July 2018. A) Total taxa (S); B) Log transformed total abundance of individuals (N); Shannon-Wiener's diversity (H'); and D) Pielou's evenness (J). Location represents position in the eastern (upcurrent) or western (downcurrent) sections of the farm.

Table 6 Summary of biodiversity indices associated with seeded and unseeded droppers in July 2018 after three months deployment

Dropper	Treatment	Location	S	N	d	H	J
L1.UCU.3A	Unseeded	Upcurrent	17	12559	1.70	0.84	0.30
L1.UCU.3B	Unseeded	Upcurrent	10	7195	1.01	0.76	0.33
L1.UCU.3C	Unseeded	Upcurrent	10	11418	0.96	0.82	0.36
L1.UCS.3A	Seeded	Upcurrent	17	34901	1.53	0.97	0.34
L1.UCS.3B	Seeded	Upcurrent	15	29993	1.36	0.99	0.36
L1.UCS.3C	Seeded	Upcurrent	13	39613	1.13	1.20	0.47
L1.DCU.3A	Unseeded	Downcurren	8	4957	0.82	0.60	0.29
L1.DCU.3B	Unseeded	Downcurren	6	1260	0.70	0.63	0.35
L1.DCU.3C	Unseeded	Downcurren	3	679	0.31	0.85	0.77
L1.DCS.3A	Seeded	Downcurren	6	4673	0.59	0.88	0.49
L1.DCS.3B	Seeded	Downcurren	8	7035	0.79	0.94	0.45
L1.DCS.3C	Seeded	Downcurren	9	12532	0.85	1.03	0.47

The generalised linear models indicated a significant effect of seaweed and position in the farm in the total abundance of individuals (N) found in the samples ($p < 0.001$). The number of taxa (S) and Margaleff's diversity (d) were not significantly higher in seeded droppers compared to controls ($p = 0.133$) while position in the farm was a significant factor ($p < 0.001$). However, Shannon's diversity (H') was significantly higher in the experimental droppers with seaweed ($p < 0.001$) but not affected by position in the farm ($p = 0.08$). None of the factors affected the generally low evenness of these early communities associated with cultivated seaweed and the farm structures ($p = 0.69$ and $p = 0.16$, respectively).

Table 7 Results of SIMPER analysis. Values represent average abundance for taxa recorded in unseeded (bare rope) and unseeded (*S. latissima*) dropper samples collected from the test farm in Ventry Harbour, co. Kerry

	Unseeded	Seeded	Cumulative
<i>Jassa</i> spp.	2588.167	1.04E+04	0.4744
<i>Mytilus edulis</i>	3461.833	7.86E+03	0.8316
<i>Caprella mutica</i>	89	2.43E+03	0.949
<i>Dexamine thea</i>	76	2.08E+01	0.9599
Copepoda	39	2.42E+02	0.9705
<i>Rissoa parva</i>	11.6667	1.69E+02	0.9796
Annelida	41.6667	1.88E+02	0.9875
Harpacticoida	4.5	5.30E+01	0.9928
Dexaminidae	4.3333	5.77E+01	0.9952
<i>Polycera quadrilineata</i>	0	5.05E+01	0.9972
Nematoda	15.6667	2.67E+01	0.9988
Amphipoda	7.5	0.00E+00	0.9992
<i>Stenothoe</i> sp.	1	5.33E+00	0.9995
Cirripedia	1.3333	8.33E-01	0.9996
<i>Galathea</i> sp.	0	8.33E-01	0.9996
<i>Caprella</i> spp.	1.1667	0.00E+00	0.9997
<i>Lacuna vincta</i>	0	1.67E+00	0.9998
<i>Musculus subpictus</i>	0.5	3.33E-01	0.9998
<i>Anomia ephippium</i>	0.5	1.67E-01	0.9998
<i>Skeneopsis planorbis</i>	0	3.33E-01	0.9999
<i>Doto</i> sp.	0	5.00E-01	0.9999
<i>Rissoa</i> sp.	0.3333	0.00E+00	0.9999
<i>Idotea</i> sp.	0.1667	1.67E-01	0.9999
<i>Palliolus</i> sp.	0	3.33E-01	0.9999
<i>Idotea balthica</i>	0	3.33E-01	1
Pleocemata	0	3.33E-01	1
Acaridae	0.1667	0.00E+00	1
<i>Pseudamussium peslutrae</i>	0.1667	0.00E+00	1
<i>Hiatella arctica</i>	0	1.67E-01	1
Cumacea	0	1.67E-01	1

Seven months after deployment, in October 2018, the kelp dominated habitat transitioned to one structured by blue mussels, following the degradation of the sugar kelp fronds, which nonetheless retained their holdfast. The cultivated kelp holdfast added structural complexity to the mussel matrix, forming a type of biogenic structure that attracted substantial numbers of commercially important edible *Cancer pagurus* and velvet swimming crabs *Necora puber* recorded feeding on the longlines and droppers covered by mussels. These mixed biogenic/artificial structures (effectively reefs) were used as nursery grounds by these species as indicated by the presence of high numbers of crab juveniles in the samples (Figure 22).



Figure 22 Velvet swimming crab *Necora puber* and edible crab *Cancer pagurus* were commonly recorded (*C. pagurus* in high numbers as juveniles) within the mussel *Mytilus edulis* dominated droppers collected from the Ventry site in October 2018, nine months after deployment.

3.1.3 Potential effects on benthic primary producers: seagrass beds

3.1.3.1 Seagrass biomass from removal samples

Seagrass beds were recorded under the test farm in Ventry Harbour and within the 100 m radius area surveyed, but they were prevalent to the south and east becoming naturally more scarce and less dense with depth and exposure, e.g. to the north and east, the habitat transitioning to bioturbated fine to medium sand frequently disturbed by bottom currents and swell. To the southwest the seafloor transitions from medium sand to large boulders and the dominant habitat are kelp *Laminaria* spp. beds. Seagrass were indirectly monitored as part of the benthic grab sampling surveys conducted at five time points during the 2017-2019 period (see methodology and Figure 12).

Mean wet seagrass biomass across the site and sampling points was 0.65 kg m^{-2} (dry weight 29 g m^{-2}), with a maximum value of 4.3 kg m^{-2} (50 m from the edge of the farm's southern limit). Highest values were recorded in the downcurrent and south farm sections, ranging from 0 kg m^{-2} to 4.2 kg m^{-2} and 4.3 kg m^{-2} , respectively. To the north and east (upcurrent) *Z. marina* biomass was comparatively low, values ranged from 0 to 1.9 and 1.39 kg m^{-2} . The control stations yielded the lowest seagrass biomass recorded, with a maximum value of 0.22 kg m^{-2} . Variability in seagrass biomass was explained by time of sampling (season, as display in Figure 18 showing significant decreases in the winter and early spring surveys) and sampling location in relation to the farm (e.g. up current, down current, north, south) as significant factors in the model ($p < 0.001$; $p < 0.01$) but not by distance from the farm. Seagrass biomass was therefore not significantly different under the farm compared to distant controls indicating the presence of the farm is not a factor in explaining decreases in biomass and abundance of *Z. marina* in

the area. The results, displayed in Figure 23 indicate clear seasonal changes in seagrass across all locations, with values of biomass higher under and near the farm decreasing away from it, and largely unchanged in the shallower areas.

Seagrass shoot density followed a similar pattern although the models indicated distance from the farm was a significant factor in explaining the variability due to the significant decreases in seagrass away from the farm in the deeper areas (north and down current; Figure 23).

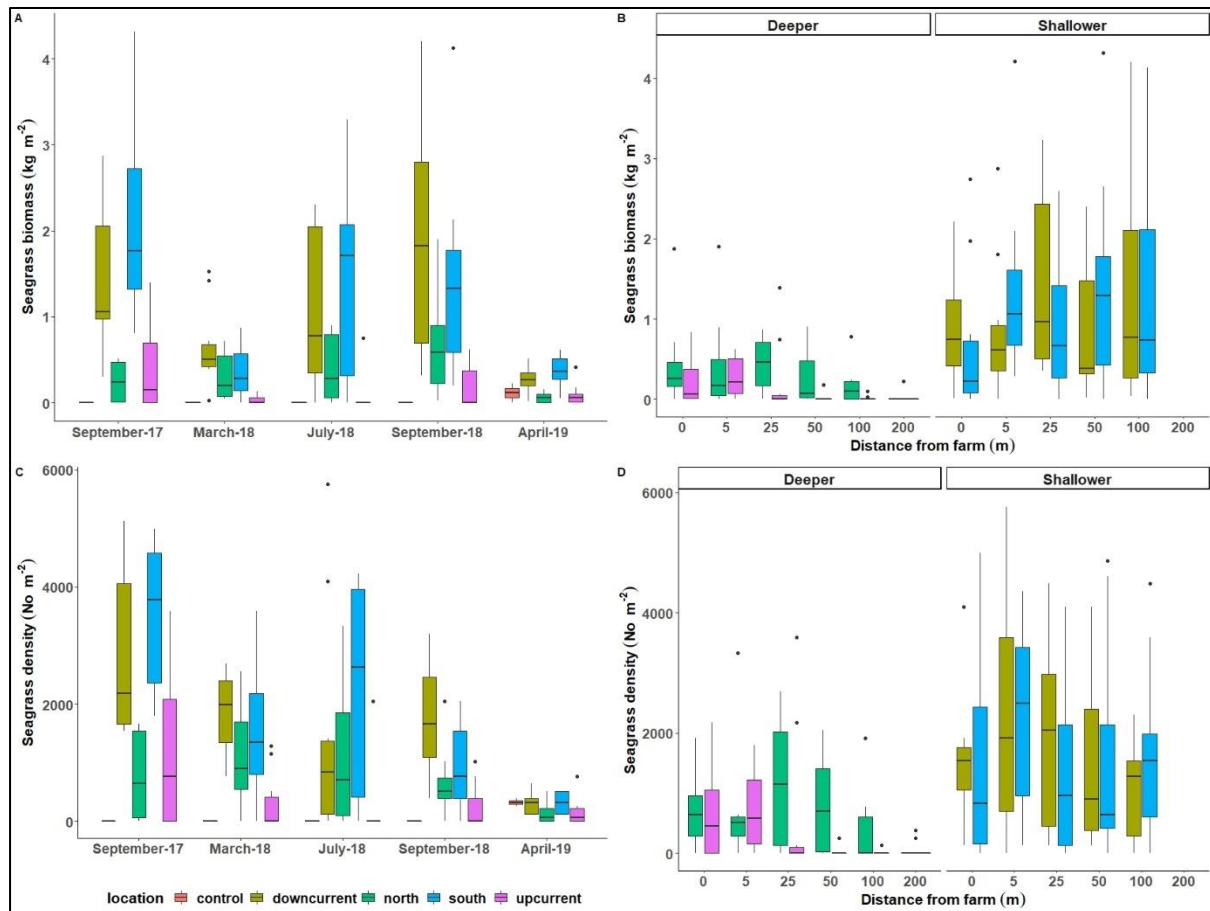


Figure 23 Boxplots for seagrass biomass (top) and density (bottom) recorded in the Ventry Harbour test farm seabed survey area between 2017 and 2018. A) and C) Temporal changes in seagrass; B) and D) spatial changes in measured seagrass abundance, from the farm (0 m) to 100 m.

Shoot density values ranged from 0 in areas north and east of the farm, to a maximum of 5,760 shoots m^{-2} recorded at the western edge of the farm, on the down current transect. Mean density was 1,077 shoots m^{-2} . In agreement with the biomass values recorded, highest densities of seagrass were recorded to the south (mean 1699.8 shoots m^{-2}) and west of the farm (1643.52 shoots m^{-2}). The values recorded in the north and east sections were 757.76 and 409.60 shoots m^{-2} , respectively. Highest densities were generally recorded in September 2017 and July 2018 (Figure 23C). The potential effect of the farm on the underlying seagrass beds was not apparent based on the densities recorded, e.g.

1,548 shoots were recorded on average under the farm on its eastern edge while 1,433 and 1,062 shoots m^{-2} were recorded 50 and 100 m from it, respectively. Similarly, in the south section, the average density was 1,600 shoots m^{-2} while 50 m and 100 m to south 1,587 and 1,677 shoots m^{-2} were recorded (see Figure 23D). To the north and east (i.e. 'upcurrent') mean seagrass densities were higher under and near the farm compared to most distant stations, e.g. 755 shoots m^{-2} under the farm in its northern edge compared to 384 and 64 shoots m^{-2} respectively recorded 100 and 200 m to the north. On the eastern edge (up current) mean seagrass density was 704 shoots m^{-2} which gradually decreased to 12.8 shoots m^{-2} 100 m to the east.

3.1.3.2 Analysis of photo quadrats: seagrass coverage

Quadrat photographs taken by divers between 2017 and 2019 painted a similar picture to that presented by the seagrass biomass and density data: seagrass in Ventry Harbour is largely limited to the area under the farm, between the farm and the shoreline to the south and patch that extends ca. 100 m to the west, ~100 m to the north and ~30m to the east from the edge of the farm (Figure 24; Figure 25).

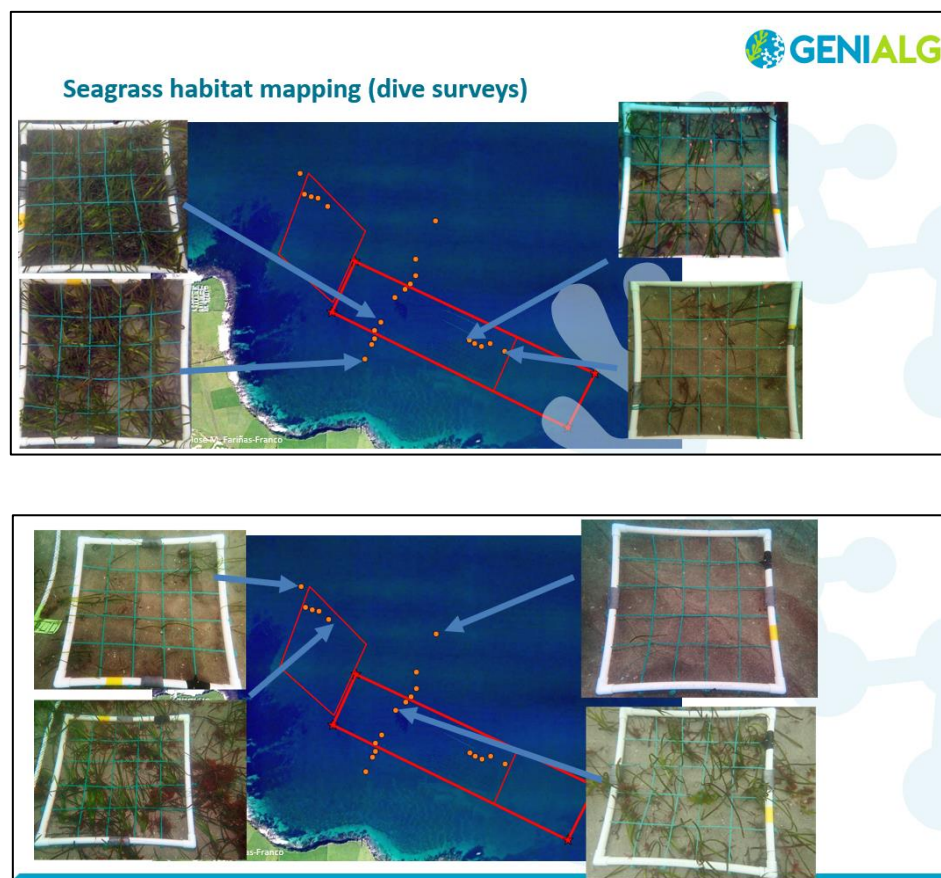


Figure 24 Map showing the location of the Ventry Harbour test farm and the positions of the photo quadrat stations along 100 m radial transects. Photographs are representative of the conditions, with highest seagrass *Zostera marina* coverage in the down current (west) and OT-N (South) transects and lowest coverage to the north and east (© Google Inc.).

On these areas, coverage was relatively high with mean values between 30% and 98% (OT-S) and 20% to 95% (Downcurrent). The control transects ~200 m north of the farm yielded the lowest seagrass cover, ranging from a mean cover of 35% during the 2017 baseline to 11% in the April 2019 pre-harvest surveys (Table 8; Figure 24).

Table 8 Summary of mean cover ratios of seagrass recorded from 0.25 m² photo quadrats during benthic surveys conducted at the Ventry Harbour test farm and surrounding area between 2017 and 2019.

Sample	Date	Month	Location	Zcover_ 0-25m	Zcover_ 25-50m	Zcover_ 50-75m	Zcover_ 75- 100m	Zcover_ Mean
2017 Baseline	06/09/2017	September	Control	0.29	0.43	0.04	0.63	0.35
2017 Baseline	05/09/2017	September	Upcurrent	0.09	0.09	0.00	0.00	0.04
2017 Baseline	05/09/2017	September	Downcurrent	0.71	0.09	0.00	0.00	0.20
2017 Baseline	05/09/2017	September	OT-N	0.73	0.54	0.38	0.42	0.51
2017 Baseline	06/09/2017	September	OT-S	0.97	0.98	0.96	1.00	0.98
2018 +3months	30/03/2018	March	Control	0.04	0.09	0.06	0.19	0.09
2018 +3months	29/03/2018	March	Upcurrent	0.52	0.46	0.67	0.48	0.53
2018 +3months	30/03/2018	March	Downcurrent	0.87	0.63	0.66	0.00	0.54
2018 +3months	26/03/2018	March	OT-N	0.16	0.04	0.00	0.00	0.05
2018 +3months	29/03/2018	March	OT-S	0.41	0.30	0.15	0.29	0.29
2018 +6months	04/07/2018	July	Control	0.19	0.19	0.10	0.14	0.15
2018 +6months	04/07/2018	July	Upcurrent	0.30	0.09	0.00	0.00	0.10
2018 +6months	11/07/2018	July	Downcurrent	0.99	1.00	0.86	0.94	0.95
2018 +6months	04/07/2018	July	OT-N	0.86	0.66	0.79	0.63	0.73
2018 +6months	11/07/2018	July	OT-S	0.99	0.99	0.96	1.00	0.98
2018 +12months	03/10/2018	October	Control	0.00	0.00	0.00	0.00	0.00
2018 +12months	04/10/2018	October	Upcurrent	0.40	0.41	0.35	0.25	0.35
2018 +12months	03/10/2018	October	Downcurrent	0.46	0.63	0.72	0.58	0.60
2018 +12months	04/10/2018	October	OT-N	0.50	0.34	0.46	0.35	0.41
2018 +12months	04/10/2018	October	OT-S	0.60	0.78	0.70	0.75	0.71
2019 +6months	01/05/2019	May	Control	0.18	0.10	0.04	0.11	0.11
2019 +6months	01/05/2019	May	Upcurrent	0.00	0.00	0.00	0.00	0.00
2019 +6months	02/05/2019	May	Downcurrent	0.59	0.44	0.48	0.60	0.53
2019 +6months	01/05/2019	May	OT-N	0.45	0.46	0.26	0.26	0.36
2019 +6months	01/05/2019	May	OT-S	0.70	0.73	0.75	0.69	0.72

Seagrass percentage cover varied seasonally, with lowest coverage recorded in March 2018 and April 2019 and highest values recorded in the summer and early autumn surveys (Figure 25). Seagrass was generally sparser to the north and east, percentage cover ranging from 4 to 50% in the eastern (UC) section and 5 - 75% along the north (OT-N) transects. In these areas to the north and west, seagrass was generally denser and had higher coverage in the vicinity of the seaweed farm, thinning out towards the control station 200 m to the north and approximately 30 m away from the edge of the farm to the east, where the seagrass patches ended giving way to a sandy, ripple seabed with

numerous tracks made by mobile fauna (Figure 25; Figure 26; Table 8). Depth along the north transect increased from ~10 m CD to 11 m CD, and more noticeably along the up current transect ranging from 10 m to 12 m away from the farm. The down current and southern transects were in slightly shallower water, starting at depths of 10 m (CD) and finished in ca. 8 m (CD). The shallower depths potentially contributed to the healthier condition of the seagrass recorded in these transects.

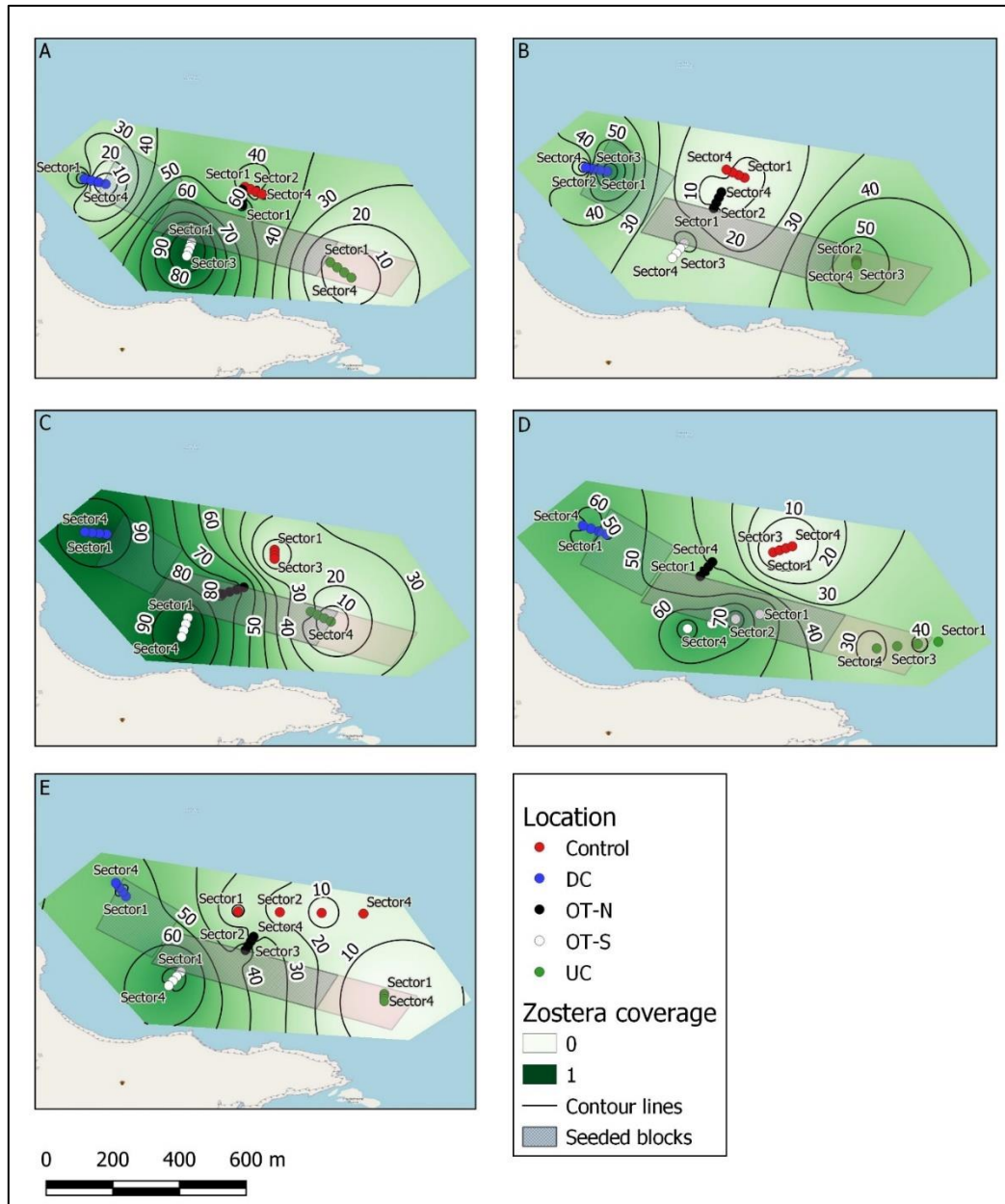


Figure 25 Temporal and spatial variability in seagrass percentage cover in the Ventry harbour survey area calculated from photo quadrats: A) September 2017 (+2 months post-harvest); B) March 2018 (+3 months post seeding/mid-season); C) July 2018 (harvest/peak biomass); D) October 2018 (post-harvest/pre-seeding); E) April 2019 (pre-harvest/peak biomass).

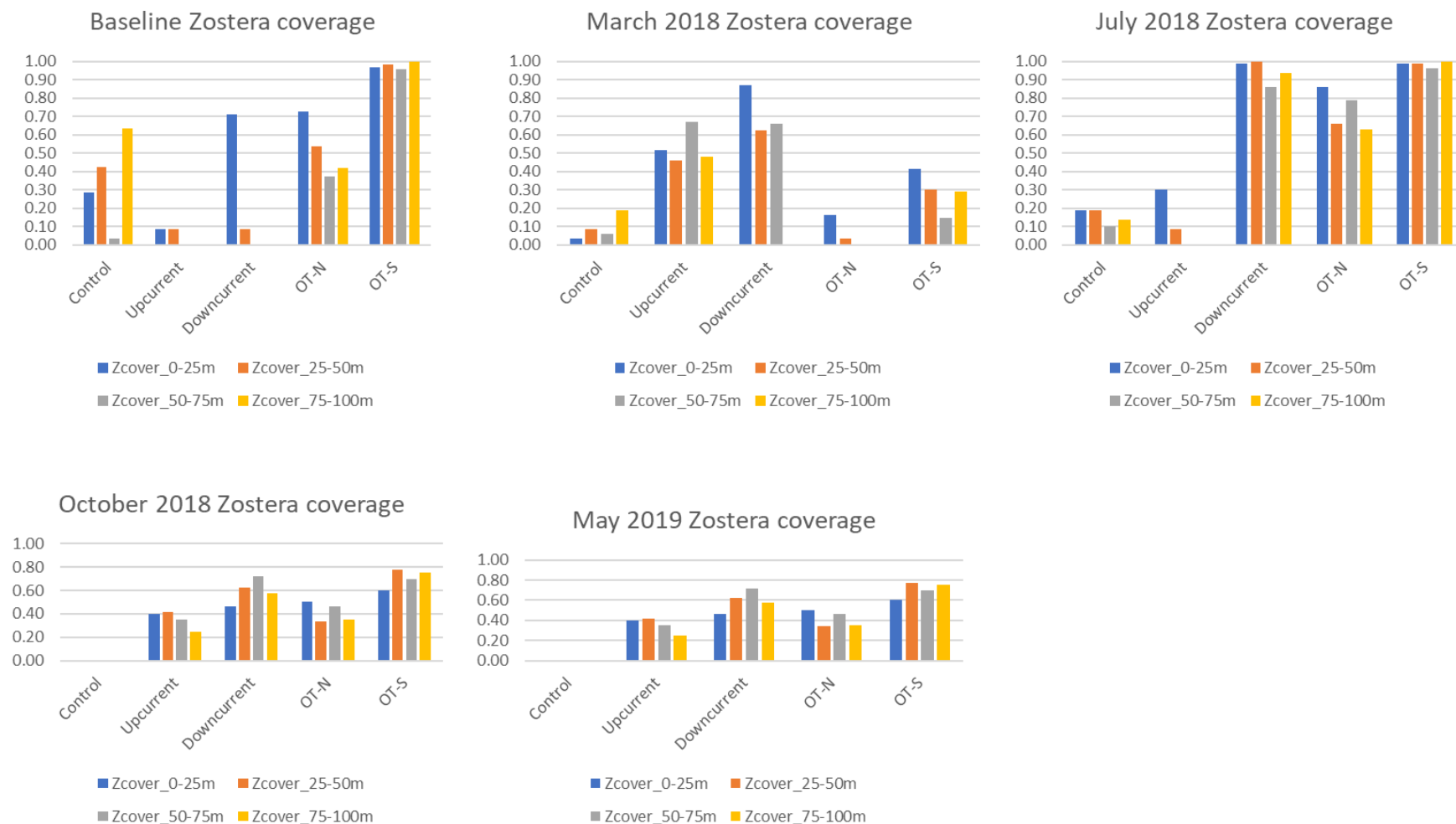


Figure 26 Barplots displaying mean ($n=5$) seabed percentage covered by *Zostera marina* along transects surveyed by divers between 2017 and 2019, Ventry Harbour test farm. Values are calculated from a total of five photographs of quadrats deployed every 25 metres from the farm.

3.1.3.3 Mobile fauna recorded in video transect surveys

Ventry Harbour

In total 35 faunal taxa were recorded in the video footage across all five surveys and transects (N=25) (see Table 9; Figure 1). These taxa belonged to five major phyla, namely Cnidaria (1), Polychaete Annelids (3), Arthropoda (11), Gastropod Molluscs (4) and Bivalves (1), Echinoderms (4) and Chordates (9). The most frequently recorded species were fish, namely sand goby *Pomatoschistus minutus*, two-spotted gobies *Gobiusculus flavescens*, and juvenile pollack *Pollachius pollachius*, the latter recorded in September 2017 and in April 2018, in high numbers. Other taxa frequently encountered included crabs *Maja brachydactyla*, *Cancer pagurus*, *Necora puber* and *Corystes cassivelaunus* (only recorded in spring, in March 2018 and April 2019, burrowed in the sand). Of interest was the presence of thornback ray *Raja clavata* and lesser spotted catshark *Scyliorhinus canicula* recorded under and near the farm from July 2018 onwards (see Figure 27).



Figure 27 Photographs of the representative benthic fauna recorded by divers during the benthic surveys carried out under and in the vicinity of the test seaweed farm in Ventry Harbour, co. Kerry (Ireland). All photos: Dr. Jose M. Fariñas-Franco

The most abundant species were fish, chiefly juvenile pollack *P. pollachius* with 590 individuals, 582 of which were recorded during harvest in the July 2018 peak biomass surveys. These pollack were associated with areas of seagrass in all transects except the control. The second most abundant species were two-spotted gobies *G. flavescens* (436 individuals, recorded predominantly in the September 2017 baseline survey and during and after the 2018 harvest (July and October). The third most abundant taxa were sand gobies *P. minutus* with 193 individuals, most of which were recorded

during the October 2018 post-harvest surveys in sandy areas in the control, upcurrent and OT North transects. The crab *C. cassivelaunus* was abundant in sandy areas during the April 2019 surveys (24 specimens out of a total of 27 recorded during the study period) along with the common starfish *Asterias rubens* (12 individuals out of 16).

Total abundance of individuals was highest during the July 2018 surveys tallying to 679 individuals recorded along the transects, predominantly fish. In total 355 individuals, mostly *G. flavescens*, were recorded in September 2017 followed by 258 individuals in October 2018 (mostly *P. minutus*), 63 in May 2019 and 26 in March 2018. Diversity was highest in March 2018 (16 taxa), followed by October 2018 (12), July 2018 and May 2019 yielded both 11 taxa, and September 2017 had the lowest diversity with 10 taxa.

The areas closest to the farm (within 25 m) reported the highest abundances of fauna with over 590 individuals, 25-50 m more than 71, 160 between 50 and 75 m and ~ 300 individuals between 75 and 100 m away from the farm. The richest areas in fauna were the downcurrent and upcurrent transects with 592 and 303 individuals recorded, followed by OT-S (187), OT N (152). The control transect was overall poor in faunal abundance with 146 individuals recorded cumulatively in all surveys.

Eastern Schelde

No quantitative data is available from the baseline benthic and longline surveys conducted at the site in May 2018. The sediment consisted of mud with a shallow redox layer, predominantly hypoxic or anoxic, and no apparent visual change along the transects surveys. Transects in the inner bay and those close to a mussel farm were potentially affected by heavy organic loads and low oxygen conditions. At the time blooms of *Phaeocystis* spp. algae were common, particularly in the inner, less exposed areas of this shallow enclosed embayment. Signs of bioturbating activity were observed, for example polychaete casts (probably *Arenicola* spp.) and bivalve burrows (cockle *Cerastoderma* spp. and *Macoma balthica*) but overall, the seabed habitats were species poor. Some epifaunal taxa were recorded during the benthic dive surveys, namely green crab *Carcinus maenas*, burrowing anemones *Cerianthus* spp. and occasional mussels *Mytilus edulis*, probably from the nearby farm.

Although no fish were recorded during the dive surveys conducted along the outer seaweed lines at the time (harvest had just concluded) amphipods *Jassa* spp. and their tubes were commonly recorded in high abundances on them, along with mussels *M. edulis* and a variety of filamentous red seaweeds. Of note was the presence of non-native species including Japanese knotweed *Sargassum* sp. clumps

(as drift) which hosted great numbers of the also non-native amphipod *Caprella mutica*, Pacific oysters *Crassostrea gigas* and the invasive sea squirt *Styela clava* (Figure 28).

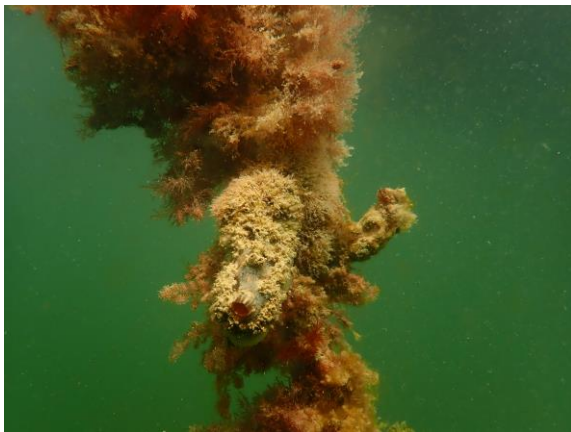


Figure 28 Non-native species associated with the studied seaweed farm in the Eastern Schelde (Netherlands). Top left: *Caprella mutica*; Top right: *Crassostrea gigas*; Bottom left: *Styela clava*; Bottom right: *Sargassum muticum*.

[illegible]

<i>Euspira nitida</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0
<i>Rissoa membranacea</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0
Bivalvia																									
<i>Pecten maximus</i>	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Echinodermata																									
Asteroidea																									
<i>Marthasterias glacialis</i>	0	0	0	0	0	0	0	1	2	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
<i>Astropecten irregularis</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0	3	2	0
<i>Ophiura ophiura</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0
<i>Asterias rubens</i>	0	2	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	4	2	5
Chordata																									
Elasmobranchii																									
<i>Raja clavata</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Scyliorhinus canicula</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	0	2	1	0	1	1	0
Actinopteri																									
<i>Pomatoschistus minutus</i>	0	10	17	0	0	2	0	0	0	0	1	0	6	0	0	114	28	0	11	3	0	0	1	0	0
<i>Pollachius pollachius</i>	0	0	2	0	6	0	0	0	0	0	0	208	134	110	130	0	0	0	0	0	0	0	0	0	0
<i>Gobiusculus flavescens</i>	0	30	267	0	3	0	0	0	0	0	0	0	77	0	0	0	0	30	7	22	0	0	0	0	0
<i>Symphodus</i> sp.	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0
<i>Symphodus melops</i>	0	0	4	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
<i>Pleuronectes platessa</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8	3	0	0	0	0	0	2	1	0
<i>Mycteroperca rubra</i>	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fish indet.	0	0	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	2	0	0
Benthic fish indet.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0

Abiotic interactions: Ventry Harbour test farm

3.1.4 *Effects on seafloor physical properties: granulometry and organic matter content*

3.1.4.1 *Granulometry*

The seabed under the Ventry Harbour test farm and surrounding area, was heavily dominated by medium sand (i.e. 250-500 μm grain diameter ϕ) with some areas of fine sand (125-250 μm ϕ) in its easternmost section. The exception were some stations east of the farm (up current, under the farm and 100 m from the edge) which yielded 76.28 and 39.76 % of mud in the 2017 baseline survey. The fraction of gravel remained absent or very low in all samples (Figure 29). Highest percentages of fine sediment fraction were consistently recorded in the down current stations, located in the most sheltered side of the farm to the northeast, towards the inner, more sheltered section of Ventry Harbour where, although sand was the dominant fraction, the mud component could be as high as 11% (Figure 13, Figure 30). Under the farm and away from it to the south, along the OT-S transect, the percentage of mud increased to 7.34% 100 m from the edge, in an area where fine silt is probably retained by the thick seagrass beds present there. To the north, OT-N transect and the Control site were predominantly sandy (over 90%) with mud representing less than 7% in all samples collected (Figure 29).

Mean grain size across all stations was 354 μm , within the medium sand category. The lowest average values were recorded in the down current stations in the 2017 baseline, 2018 mid-season and 2019 peak biomass periods (267 – 231 μm ; medium to fine sand), and south of the farm (OT-S) after the 2018 harvest, in September 2018 (mean = 255 μm ; medium sand). Seabed in the down current area had a mean grain size lower than 250 μm (fine sand) and the content in mud particles (i.e. < 65 μm diameter) was generally higher compared to all other areas, regardless of sampling period. Mean sediment grain size across all stations under the farm was similar to the global mean (354 μm , i.e. medium sand), lower than the value recorded at the control station (410 μm , also medium sand) and similar to the mean value recorded 100 m from the edge of the farms (343 μm). Interpolation maps of seabed mean grain size across the survey area are displayed in Figure 31.

Statistical analysis (Table 10) showed no significant differences in the dominant sediment type along the distance gradient away from the farm (Distance factor, pseudo- $F_{(4,41)}=1.07$; $P=0.39$), or (to a lesser degree) between sampling periods (Season factor, pseudo- $F_{(4,41)}= 2.61$; $p=0.06$). Location and its interaction with season were a significantly factor (Table 10;

Table 11, $p < 0.001$) largely due to the dominance of finer sand in the down current stations west of the farm and the area between the farm and the shoreline in the Autumn of 2018. Overall, the evidence gathered between 2017 and 2019 at the Ventry test site provides no indication of a direct effect from the kelp farm on seabed composition, grain size composition being dependent of seasonal and spatial factors likely influenced by topography, depth, storm events and prevailing hydrodynamic conditions at the site.

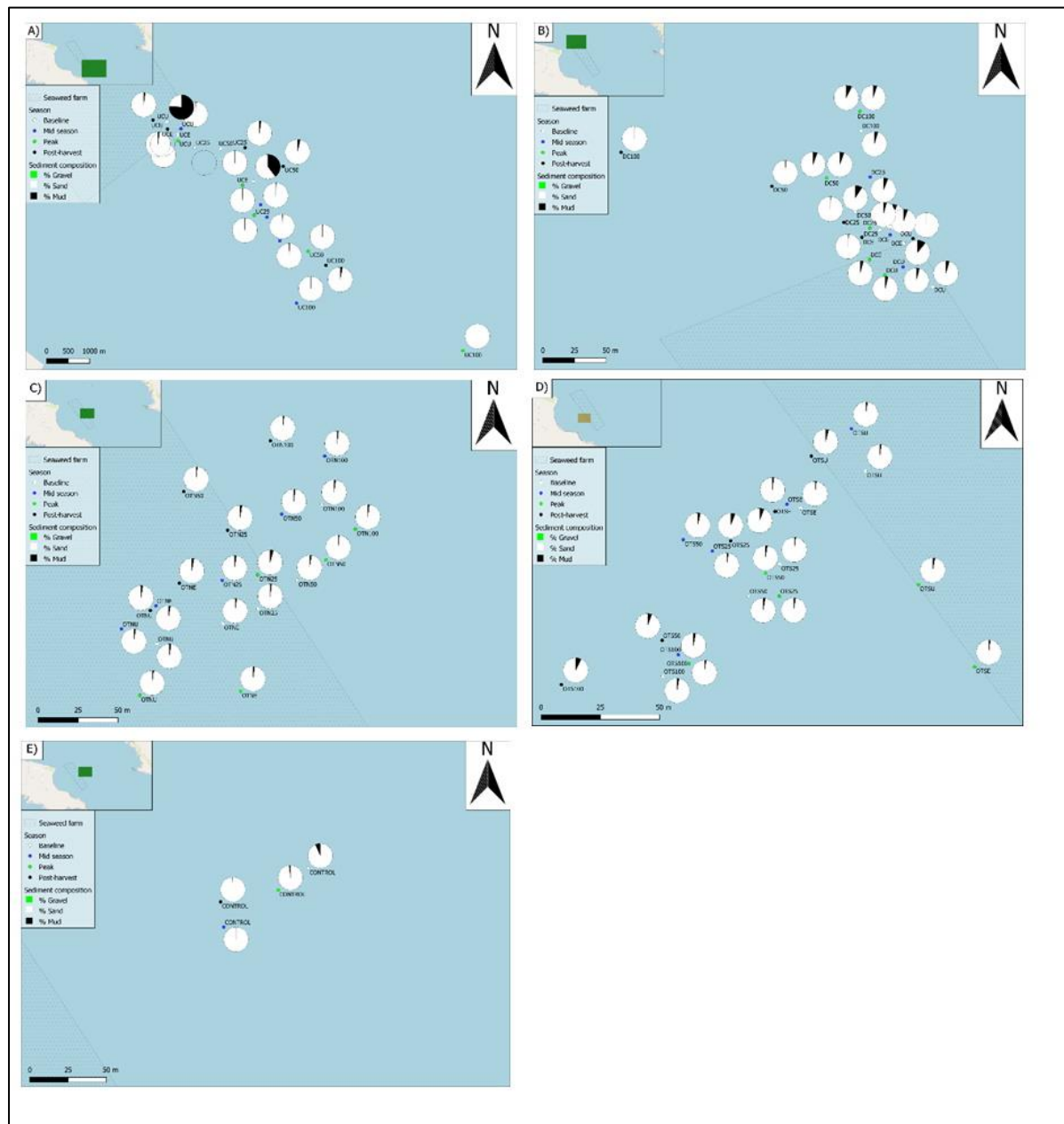


Figure 29 Maps of all sampled sites per location: A) Up-current (UC); B) Down-current (DC); C) Orthogonal-north (OTN); D) Orthogonal-south (OTS); and E) Control. Pie charts show the sediment composition: green for gravel, white for sand and black for mud (empty charts represent no data available). The colour of the site represents the season: white for Baseline, blue for Midseason, green for Peak and black for Post-Harvest.

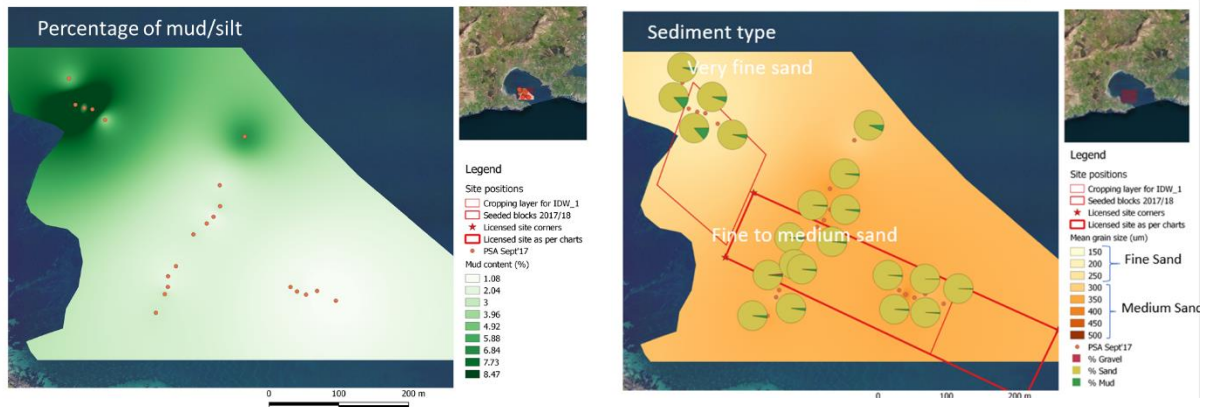


Figure 30 IDW maps showing typical mud content (left) and sediment types under the Ventry Harbour test farm and surrounding area.

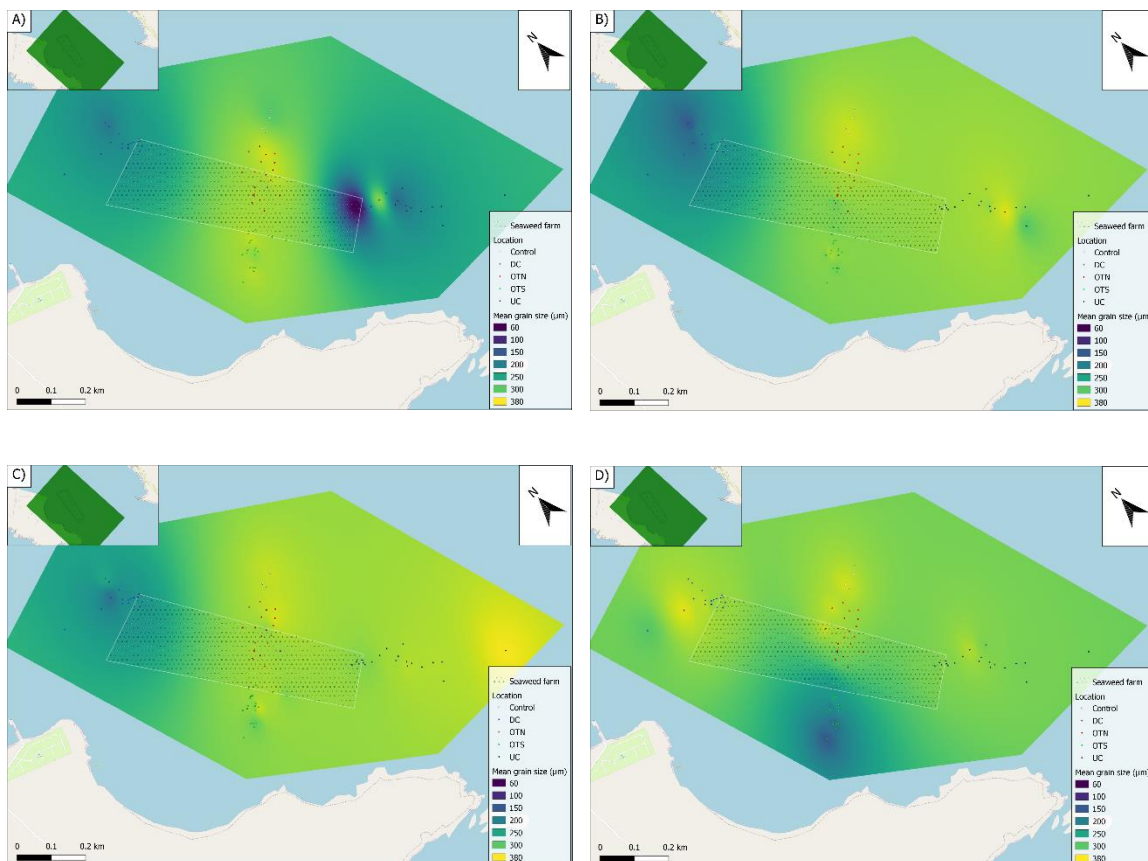


Figure 31 Interpolation map of mean grain size for each season: A) Baseline; B) Midseason; C) Peak; D) Post-Harvest. Dark tones represent low values. The colour of sites represents the location: purple for UC, blue for DC, red for OTN, green for OTS and white for Control.

Table 10 PERMANOVA analysis of the mean grain size in sediment samples from the seaweed farm in Ventry Bay. *p*-values were obtained using 999 permutations

Source	df	SS	MS	Pseudo <i>F</i>	<i>p</i>	
Season	3	8537	2845.5	2.61	0.063	.
Location	4	100044	25011.0	22.93	0.001	***
Distance	4	4675	1168.8	1.07	0.389	
Season × Location	12	173599	14466.6	13.26	0.001	***
Season × Distance	12	9189	765.8	0.70	0.764	
Residuals	46	50180	1090.9	-	-	
Total	81	346224	-	-	-	

Table 11 Pair-wise comparison of mean grain size among locations. The average distances between pairs of conditions were calculated using the Euclidean distance algorithm

Pair-wise comparison	df	SS	F Model	<i>p</i>	<i>p adjusted</i>	
UC vs DC	1	3.92e+04	8.51	0.005	0.05	.
OTS vs OTN	1	1.77e+04	8.69	0.001	0.01	*
OTN vs DC	1	9.02e+04	47.27	0.001	0.01	*

3.1.4.2 Organic matter in seabed sediment samples

The percentage of organic matter (TOM) in sediments was low across sites (<3%; Figure 31) but slightly increased in each of the surveys conducted during the studied period (Figure 32; Figure 33). TOM values ranged from 1.24% in the 2017 baseline, 1.45% in the March 2018 Midseason, 1.74% in the 2018 Peak-biomass sampling in July (before harvest) and two months after harvest (September 2018), to finally increase to 2.53% in the sampling conducted in the pre-harvest survey in April 2019.

In the September 2017 baseline survey, organic matter contents in sediments were lowest in the up current and north sections (~1% in both) while the areas south (OTS), west (DC) and 200 m north of the farm (control) were slightly more enriched (~1.5%). During mid 2018 growing season, total organic carbon (TOC) in the sediment was more heterogeneous among sampled sites, with higher values compared to the September 2017 samples, especially in the down current section (mean TOC=1.7%). An increase in TOC was registered across all areas just before and after the 2018 harvest, with up to 2.55% TOC in OTS-50, while the TOM content in the OTN transect, near the farm, and in locations DC and UC remained between 1.26% and 2.02% (OTN-E and DC-25 respectively), and 1.68% in the control. After the harvest (September 2018) TOC peaked nearest (OTN), with 2.63% of TOC in OTN-50, and in the rest of the locations it remained relatively high, decreasing in Control and DC, with 1.19% in DC-50 as the lowest value. In April 2019, there was a noticeable increase in organic matter in the sediments, in all areas including the control, with mean values ranging from 2.3% north of the farm to 3.1% to the south. Generally, there was a slight positive distance gradient in the mean organic fraction present in the sediments, with highest values recorded under the farm (1.89-1.98%) compared to distant (100m) stations (1.5-1.8%). This gradient was most noticeable in the peak and post-harvest biomass sampling periods, towards the north and south of the farm (Figure 33).

Significant differences were found for TOM according to Season ($p < 0.001$) and to the interaction between the factors Season and Location ($p < 0.001$) (Table 12). The pair-wise comparison resulted in significant differences between seasons Baseline and Peak-biomass ($p < 0.006$), Baseline and Post-harvest ($p < 0.006$), Midseason and Peak-biomass ($p < 0.027$), and between Midseason and Post-harvest ($p < 0.036$) (Table 12; Table 13).

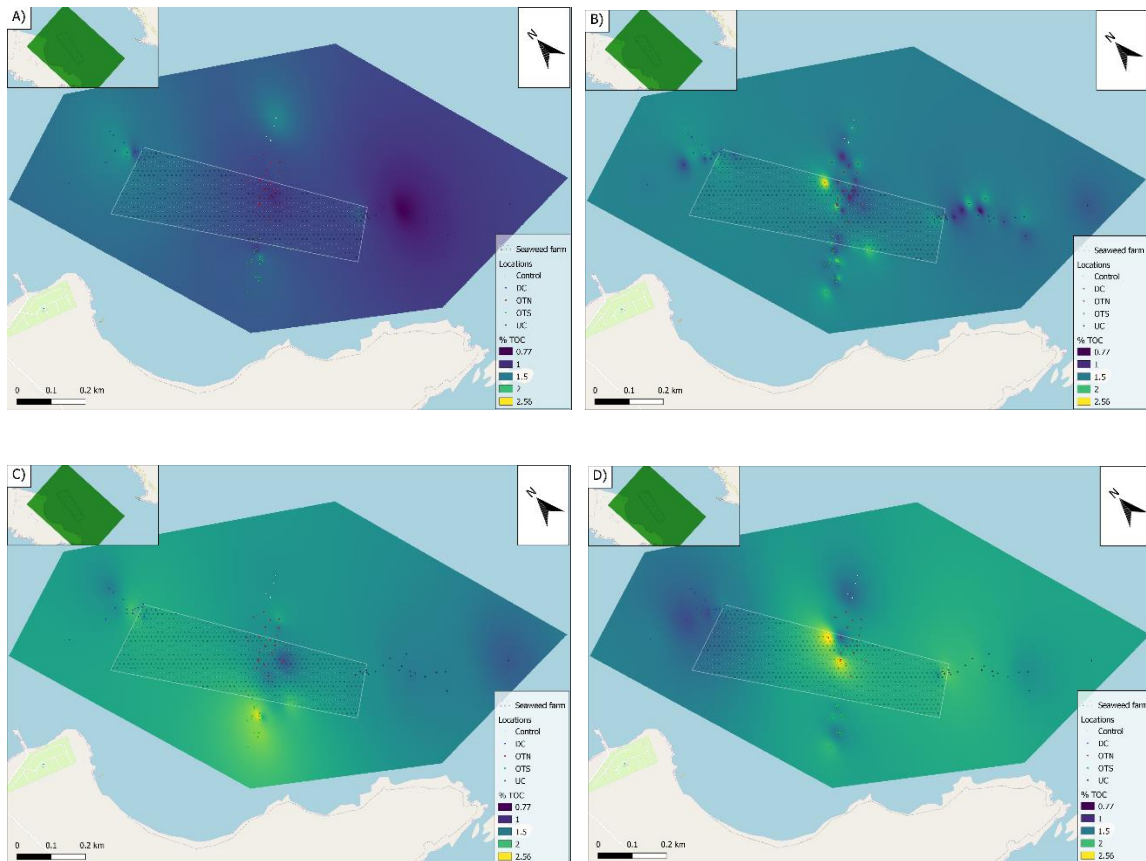


Figure 32 Interpolation map of TOM (%) for each season: A) Baseline; B) Midseason; C) Peak; D) Post-Harvest. Dark tones represent low values. The colour of sites represents the location: purple for UC, blue for DC, red for OTN, green for OTS and white for Control.

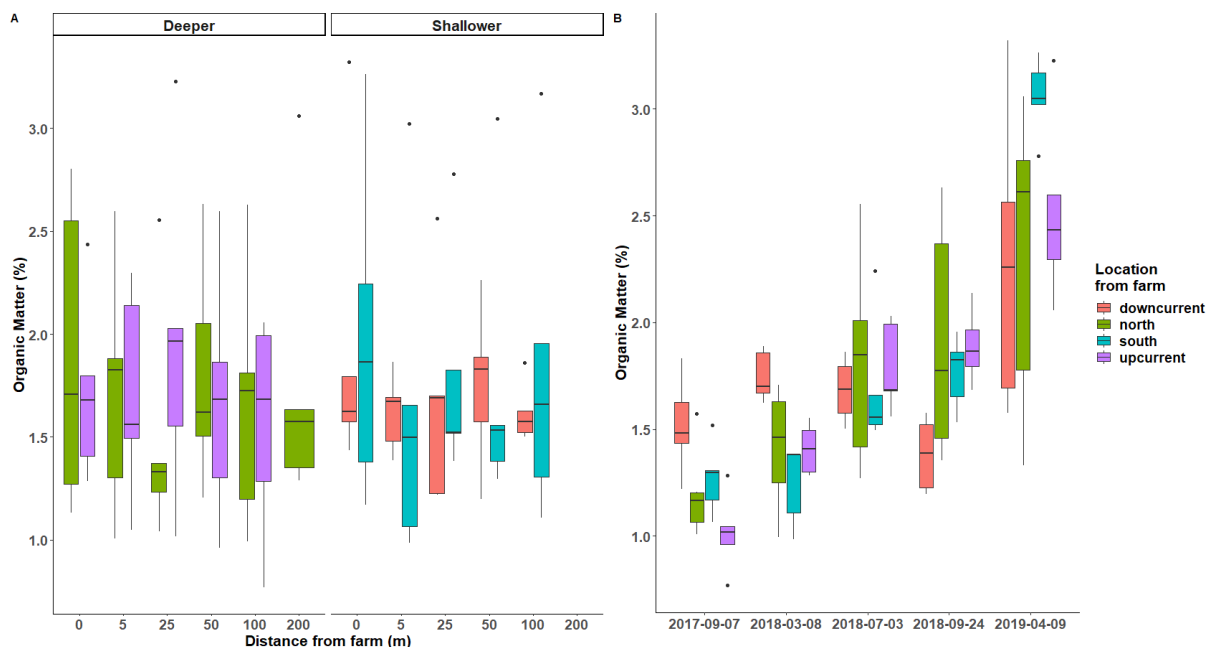


Figure 33 Boxplots for organic matter content (%) in sediments sampled in Ventry Harbour between 2017 and 2019. A) Values recorded from the farm (under/0m) to the outer distant stations (100 m) and control (200 m). B) Temporal variation in each area monitored. X axis represents sampling points, namely: 2017-09-07 (Baseline 2017), 2018-03-08 (Mid 2018 season), 2018-07-03 (peak biomass 2018), 2018-09-24 (post-harvest 2018) and peak biomass/pre 2019 harvest.

Table 12 PERMANOVA analysis of TOM in sediment samples from the seaweed farm in Ventry Bay. *p*-values were obtained using 999 permutations.

Source	df	SS	MS	Pseudo <i>F</i>	<i>p</i>	
Season	3	3.75	1.25	20.79	0.001	***
Location	4	0.33	0.08	1.37	0.25	
Distance	4	0.26	0.07	1.09	0.37	
Season × Location	12	3.64	0.3	5.05	0.001	***
Season × Distance	12	0.28	0.02	0.39	0.96	
Residuals	48	2.88	0.06	-	-	
Total	83	11.15	-	-	-	

Table 13 Pair-wise comparison of TOM among seasons. The average distances between pairs of conditions were calculated using the Euclidean distance algorithm

Pair-wise comparison	df	SS	F Model	<i>p</i>	<i>p adjusted</i>	
Baseline vs Peak-biomass	1	2.64	32.68	0.001	0.006	*
Baseline vs Post-harvest	1	2.60	24.93	0.001	0.006	*
Midseason vs Peak-biomass	1	0.91	11.37	0.004	0.027	.
Midseason vs Post-harvest	1	0.9	8.60	0.006	0.036	.

3.1.5 Hydrodynamics and sediment dynamics

3.1.5.1 Summary of hydrodynamic model results

The following represent a summary of the results of the 2-D hydrodynamic simulation and the acoustic data collected in the field and used to calibrate the model. A full report with the details results of the model is in preparation.

Observations from the models indicated current velocity is attenuated within the farm up to 31% and up to 34% in the area adjacent to the farm. The introduction of the farm in the model (M2) results in

flow being diverted around the farm. There is a phase lag between both models with increased water circulation in centre of harbour and flow direction reversing between both models. Previous studies attribute possible causes to the increased frictional resistance exerted by the seaweed which forces the water to diverge around the farm, accelerating the current speeds. There is also an evident phase shift between models without and with the farm which suggests that the impeded flows are “building up” behind the obstruction.

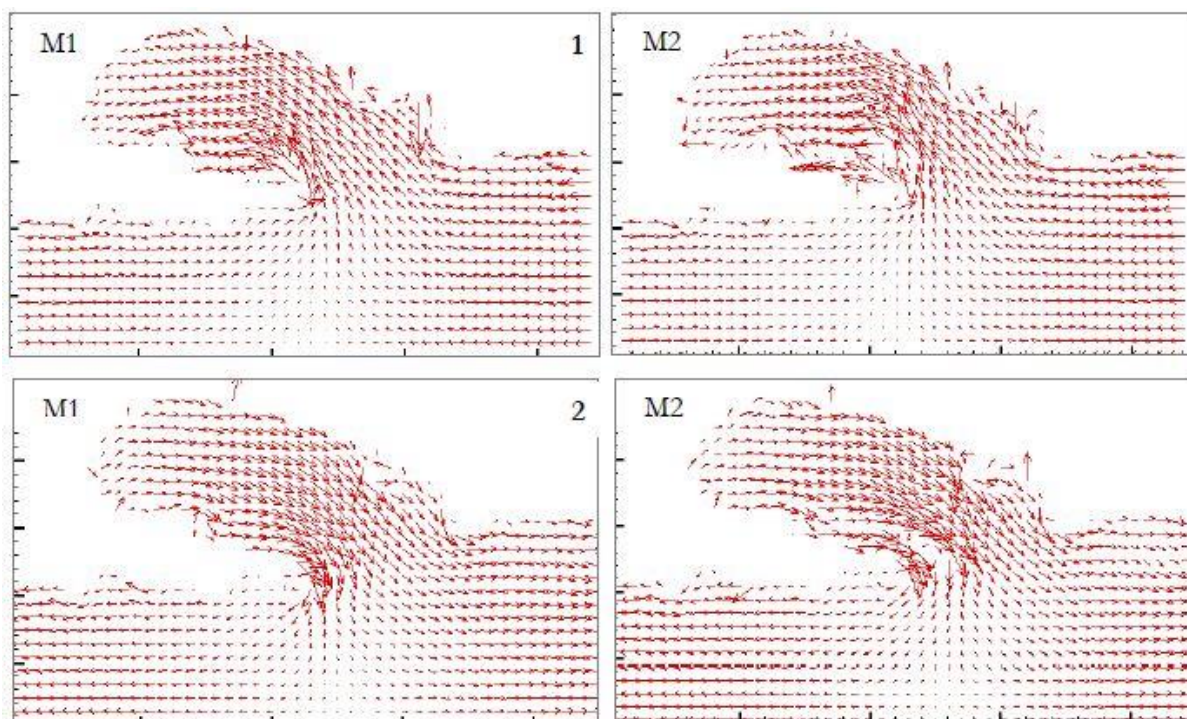


Figure 34 Two-dimensional hydrodynamic model current velocity vector maps during flood (top) and ebb (bottom) tides in Ventry Harbour. M1= Model without farm; M2=Model with farm obstruction.

Both models indicated a few large velocity vectors concentrated around the southwest corner of the mouth of Ventry, where flows coming into the harbour during mid-flood are forced around the peninsula at a sharp angle (see Figure 34). This event forces currents towards the centre of the harbour and produces low-flow conditions in the area immediately behind the end of the peninsula. As water exits the harbour, the currents are more evenly distributed among the mouth and do not exhibit the same forcing conditions as with the mid-flood flows. This area is where the seaweed farm was present, and the complex bathymetry is taken into consideration.

Physical data from the Acoustic Wave and Current (AWAC) profiler indicated the average current velocities outside of the seaweed farm are generally unchanged for both the M1 and M2 conditions. The exception was during the period between 50 and 150 hours, where M2 experiences higher velocity magnitudes. Flow velocities during this period had an average increase of up to 60%, whereas

the rest of the measured values only averaged an increase of 6%. This period corresponded to spring tide and could be influenced by the deeper and stronger currents.

The Nortek Aquadopp® acoustic profiler deployed under the seaweed farm, recorded an average decrease of 14% in current velocities. There was not a distinct velocity deviation as evident in the AWAC location, which suggests that these flows are not strongly influenced by spring tides. Current direction, however, is noticeably more consistent during M1 conditions.

The observations indicated that the model underestimated current speeds while physical data was subjected to stronger environmental conditions (noisy). Both model and field measurements showed flow decrease within farm, of smaller magnitude for physical data (14%) and no apparent flow direction reversal in physical data.

3.1.5.2 Sediment deposition study

Sediment accumulations were obtained from triplicate sediment traps deployed in the test farm in Ventry Harbour over seven weeks (between 9th January 2019 to 26th February 2019) indicating less sedimentation under the test farm compared to the controls). Mean dry sediment weight deposited under the farm was 72.1 g while controls yielded 91.8 g (north control station) and 317.2 g (control south stations) (Figure 35).

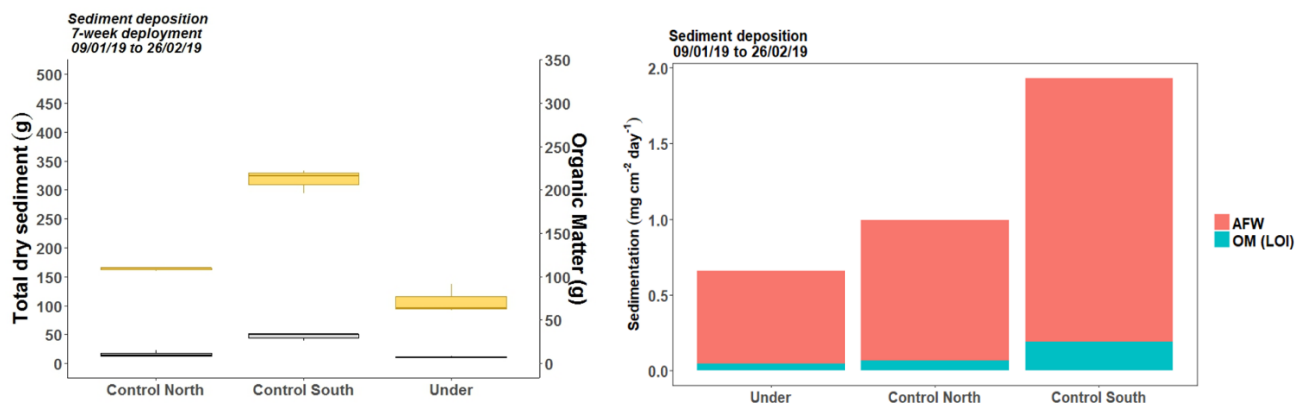


Figure 35 Sediment deposition under the Ventry test farm and at two controls, to the north and south. Left: boxplots representing range in total dry sediment (yellow) and organic matter content (grey) in the three replicate sediment traps deployed; Right: Sediment deposition rate during the studied period (49 days).

According to General Linear Models (GLMs), position was a significant factor ($F_{(2,11)}=16.82$; $p<0.001$). According to the post-hoc analyses, the significance was due to the significantly higher depositions recorded in the south control station compared to the norther control and the under-farm locations

($p < 0.001$). Although higher deposition was recorded in the control north compared to the values recorded under the farm these differences were not statistically significant ($p = 0.86$).

Loss on ignition (LOI) analyses indicated the organic fraction in the deposited sediment was minimal, ranging from 1.3% to 2.5% across all positions but highest in the control south station, located within a seagrass bed (Figure 35). These results indicate sediment deposited on the seafloor under and near the test farm was predominantly sand.

3.1.6 *Water turbidity and suspended matter*

Two years' worth of monthly water sampling at the Ventry Harbour test farm revealed that ambient total suspended matter content is naturally high, over 50 mg l^{-1} . Across the sampled period, mean concentrations of total suspended solid (TSS) in water samples were similar regardless of position, e.g. 53.2 mg l^{-1} in the centre of the farm, 51.1 mg l^{-1} (Control, $\sim 200 \text{ m}$ to the north), 52.01 mg l^{-1} (Downcurrent station), 50.3 mg l^{-1} (up current, more exposed location $\sim 100 \text{ m}$ to the east) and 47.7 mg l^{-1} in the Inner Bay, $\sim 1 \text{ km}$ to the west of the farm. Seasonally, highest TSS concentrations in the water column were recorded in winter, e.g. 64 mg l^{-1} in December 2017 and October 2018, decreasing in the spring and summer periods, e.g. 31 mg l^{-1} in April 2018 and 12.4 mg l^{-1} in May 2018 (both during active seaweed cultivation season). Mean off bottom turbidity across all stations and sampling periods was marginally higher compared to sub surface values (51.4 and 50.2 mg l^{-1} respectively). Monthly results for TSS are displayed in Figure 36.

Calculations of ash free weight for water sample filtrates indicated total particulate suspended matter in Ventry Harbour is predominantly inorganic in nature (i.e. sand) with particulate inorganic matter (PIM) fraction ranging between 87% mean PIM content in the centre station to 89% PIM in the down current station. Particulate organic matter (as percentage of TSS) was consistently under 28% across all stations sampled in all months. Highest POM fraction was recorded in the Inner Bay station in May 2018 (28%) while lowest (0.01%) was recorded in the centre of the farm in October 2018. On average highest POM content was recorded in the centre of the farm and the up current station (13%) while in the remaining sites POM varied between 11 and 12%. Seasonally, POM was generally lowest during the autumn period (after the harvest) with the exception of the sampling area within the farm and Inner Bay in June 2018, which recorded an increase in POM content in water samples near bottom at 26% and 43% compared to all other sites. Overall, there were no differences across all sampling stations and seasons in average POM content in water samples ($\sim 12\%$). Monthly results for POM in water samples are displayed in Figure 37

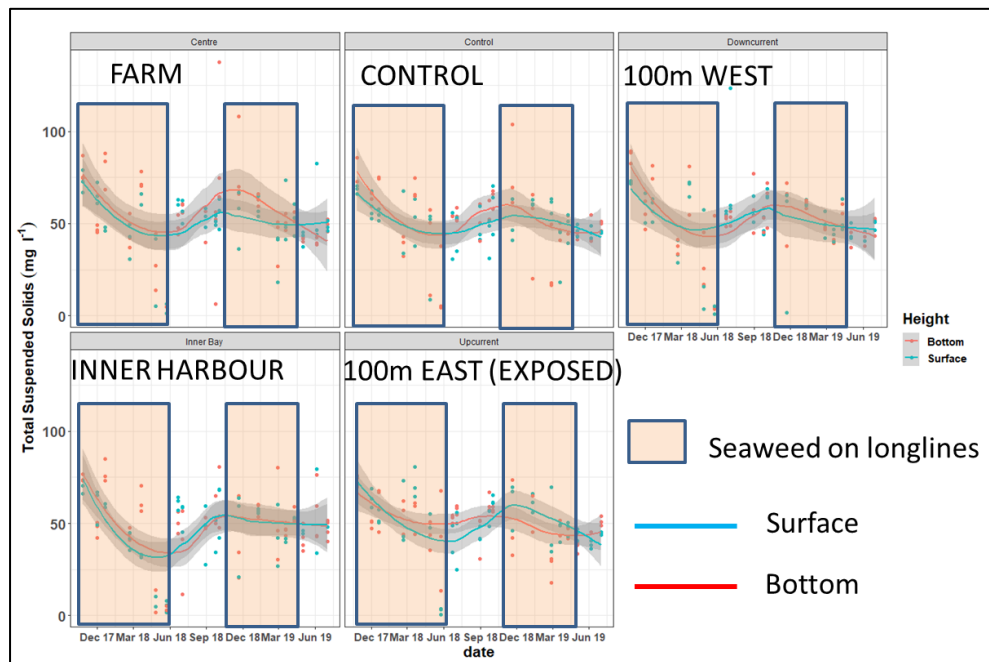


Figure 36 Variation in total suspended solids (mg l^{-1}) measured from monthly water samples collected at water stations in and around the test farm in Ventry Harbour between September 2017 and July 2019.

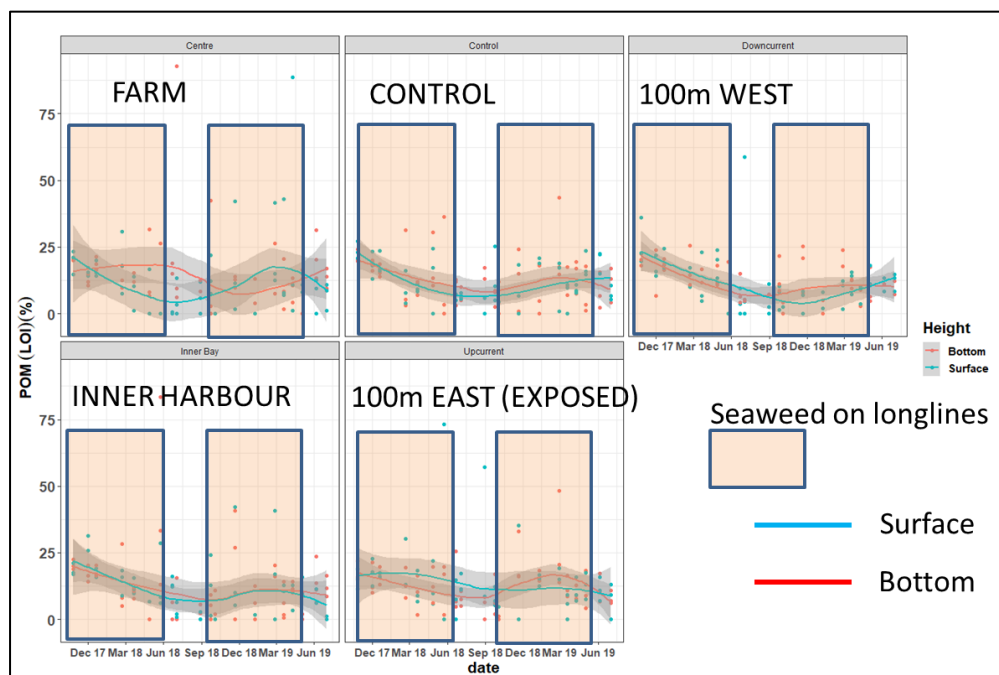


Figure 37 Variation in Particle Organic Matter (POM) (%) from TSS values measured from monthly water samples collected at water stations in and around the test farm in Ventry Harbour between September 2017 and July 2019.

3.1.7 Nutrients

Generally, there was an increasing trend in ammonia (as NH_4^+) concentrations in water samples collected at the Ventry Harbour sites between September 2017 to July 2019. Highest concentrations were recorded in August 2018 (0.36 Mm) and lowest in October 2017 (0.03Mm) with lowest

concentrations in Autumn/Winter and highest in Summer. Overall, there were no substantial differences in ammonia concentrations between sites with mean value ~ 0.15 mM.

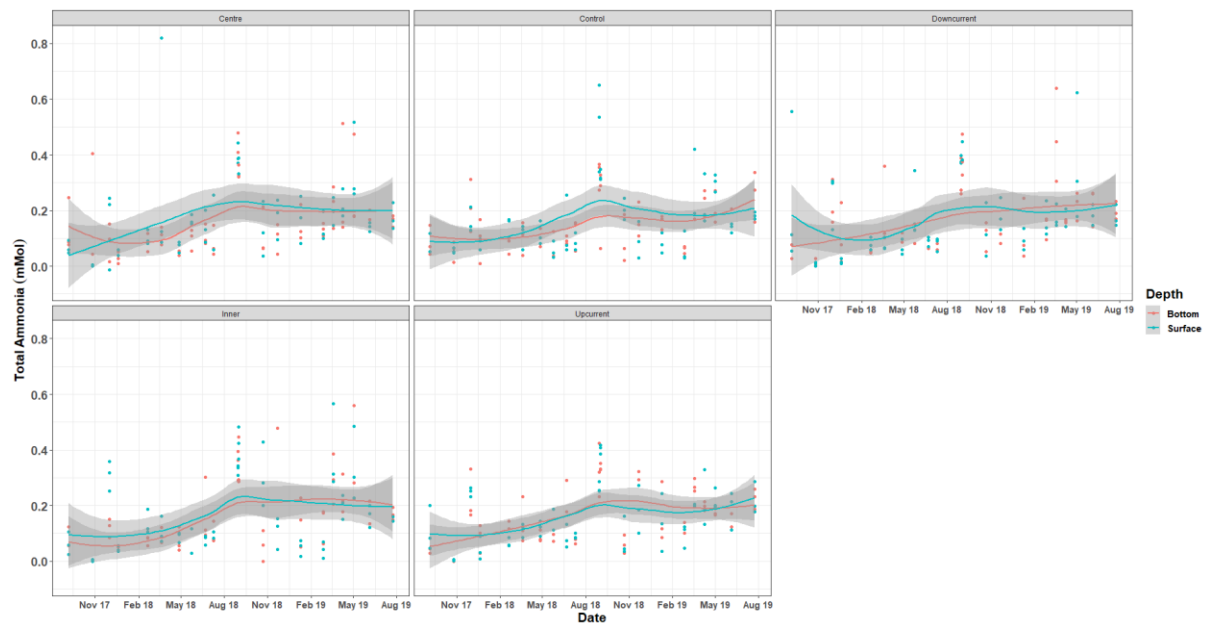


Figure 38 Total ammonia (NH_4^+) concentrations in water samples in Ventry Harbour.

Dissolved inorganic nitrogen (DIN) concentrations in seawater remained relatively constant between September 2017 to July 2019 with highest concentrations in the winter (0.01 mM February 2018) and lowest in Summer (0.002 mM May 2019). There were no substantial differences in DIN concentrations between sites with mean value between 0.019 mM (Control) and 0.026 mM (Inner Bay). Bottom DIN concentrations were lower on average (0.020 mM) compared to subsurface mean values (0.024 mM).

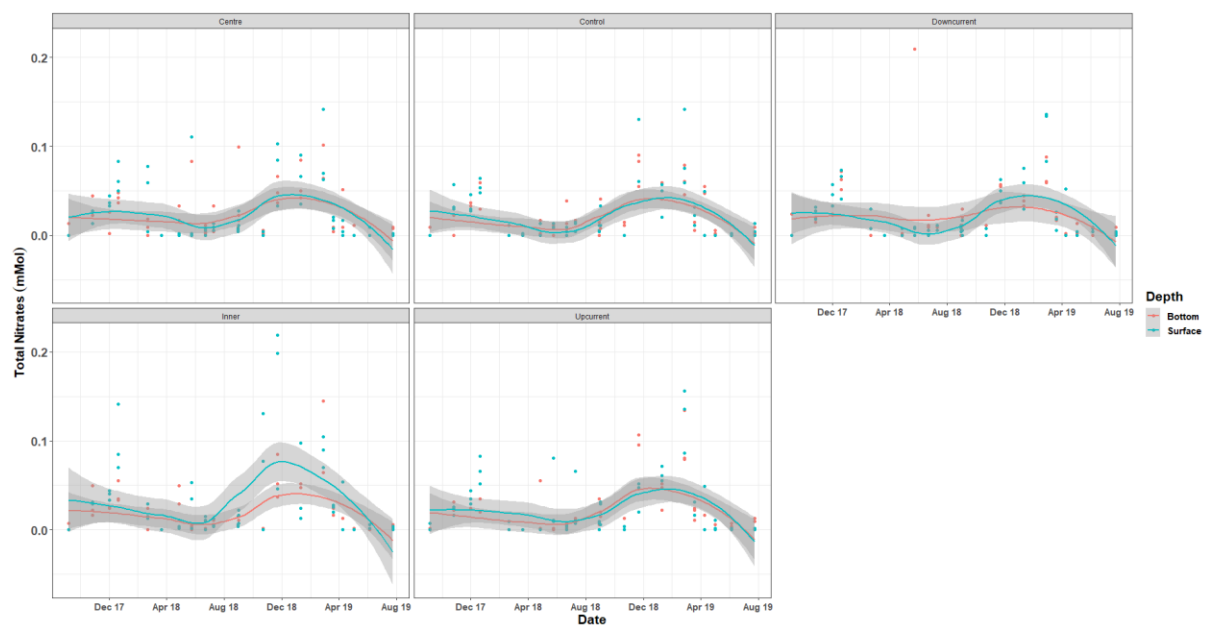


Figure 39 Total dissolved inorganic nitrogen (DIN) concentrations in water samples in Ventry Harbour.

Phosphates were generally found at concentrations below the detection threshold and remained relatively constant and very low between September 2017 to July 2019 with some slight peaks in December 2017 (0.004 Mm) and lowest in in Summer ($4.8 \cdot 10^{-5}$ mM in June 2019). Overall, there were no substantial differences in phosphate concentrations between sites with mean value between 0.0009 mM (Downcurrent) and 0.014 mM (Inner Bay). Surface concentrations were lower on average (0.0009 mM) compared to near bottom mean values (0.0012 mM).

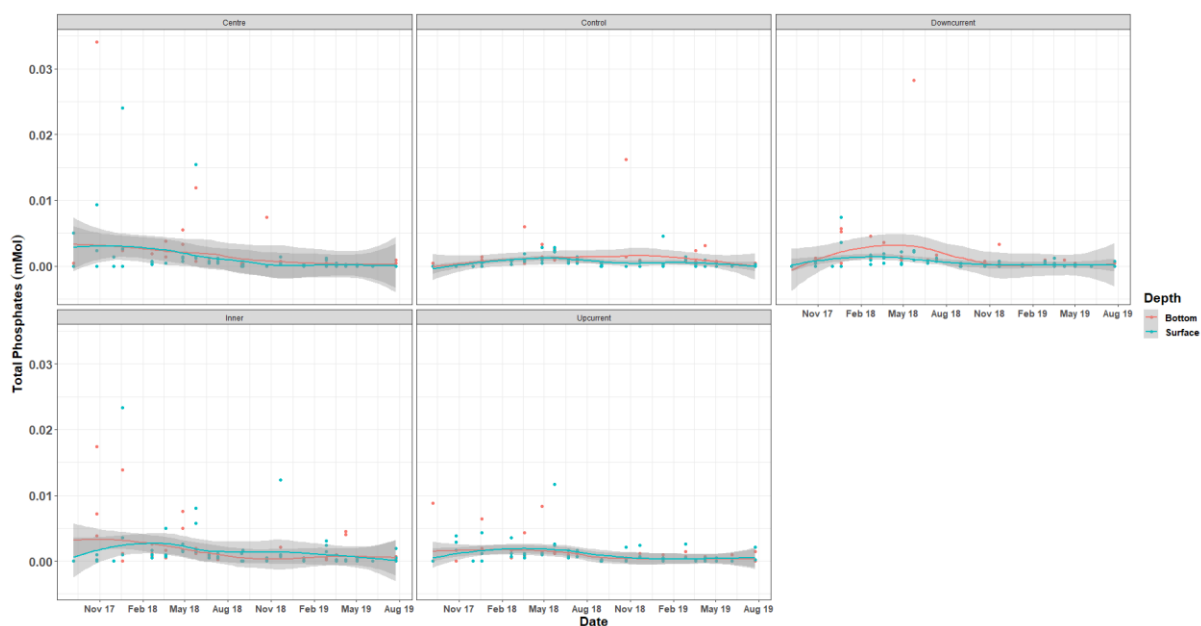


Figure 40 Total phosphate concentrations in water samples in Ventry Harbour.

3.1.8 Light penetration (Irradiance) and Turbidity

Measurements of PAR recorded by the light loggers deployed on the seabed yielded up to double the values of irradiance reaching the benthos under the farm compared to controls outside farm, even in the shallower (4.5 m CD) south control station located between the farm and the shoreline. Mean PAR values for the duration of the monitoring period (25 months) were highest in the subsurface logging station in Line 1 (~1.5 to 3 m depending on seaweed weight) at $23.13 \text{ mol m}^{-2} \text{ s}^{-1}$. Mean PAR recorded under the seaweed farm, under Line 7 approximately in the centre of the cultivated area, was $11.59 \text{ mol m}^{-2} \text{ s}^{-1}$. Mean PAR measurements by the loggers attached to the frames deployed in the control stations were $3.36 \text{ mol m}^{-2} \text{ s}^{-1}$ (Control North, ~200 m from the northern farm boundary) and $5.69 \text{ mol m}^{-2} \text{ s}^{-1}$ (Control South). The highest mean PAR value recorded in any day during the monitoring period was $207 \text{ mol m}^{-2} \text{ s}^{-1}$ in the sub-surface station, followed by $128 \text{ mol m}^{-2} \text{ s}^{-1}$ reaching the seafloor under the farm, $30 \text{ mol m}^{-2} \text{ s}^{-1}$ in the seafloor south and north controls. See Figure 41.

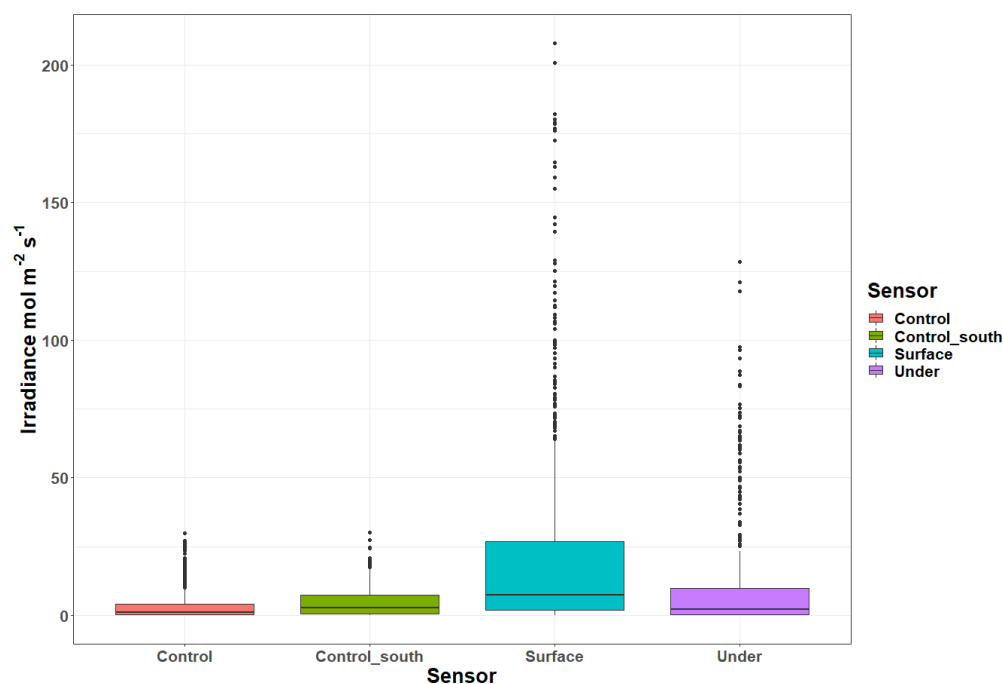


Figure 41 Boxplots for PAR measurements obtained between September 2017 and October 2019 at the Ventry Harbour test farm in Co. Kerry, Ireland.

Daily mean PAR values followed a typical seasonal pattern with highest values recorded during the summer months, with longer daylight periods and more frequent clear skies, decreasing in the autumn and winter. It should be noted that the sub-surface logger experienced a decay in PAR values due to the development of algal biofilms during the warmer summer months only to return to normal ambient values after the logger was manually cleaned in each monthly visit to the site, which explains the peaks and troughs in Figure 42. While light attenuation under farm did occur, this was likely due to depth and not to shading by overlying seaweed, as daily logger readings showed near-bottom irradiance under the farm being consistently higher compared to distant controls (Figure 42). Light (PAR) monitoring results were matched by turbidity readings by the RBR turbidity sensors (Figure 43). The data indicated very high natural turbidity at the site with peak readings over 1000 mg l⁻¹ during Autumn and Winter when stormy conditions were more common. Nonetheless, turbidity was overall higher outside the farm compared to under it, with mean turbidity values of 310 mg l⁻¹ outside the farm compared to 149 mg l⁻¹ under it.

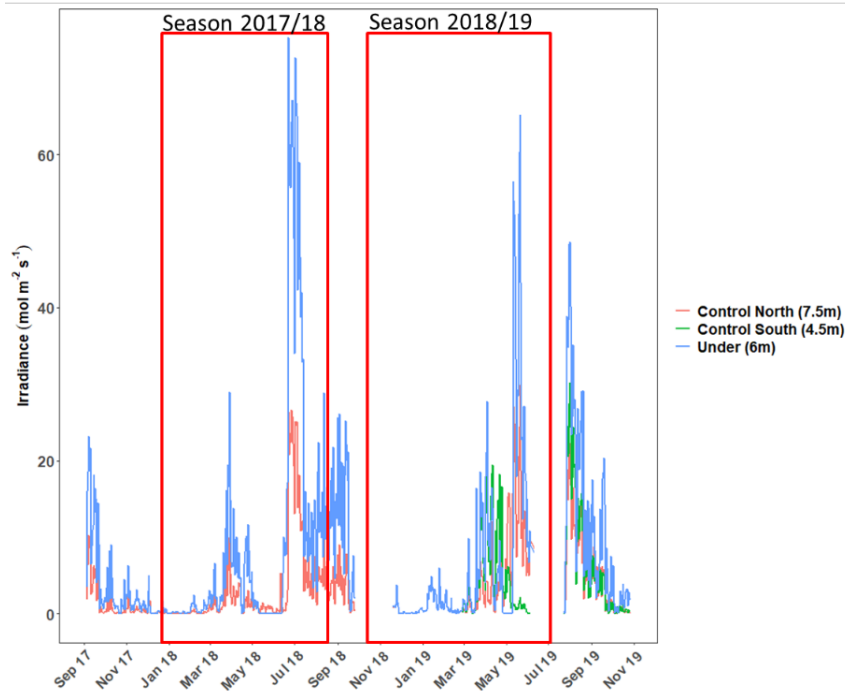


Figure 42 Daily mean Photosynthetically Active Irradiance (PAR) values recorded at three locations in the Ventry Harbour site between September 2017 and October 2019.

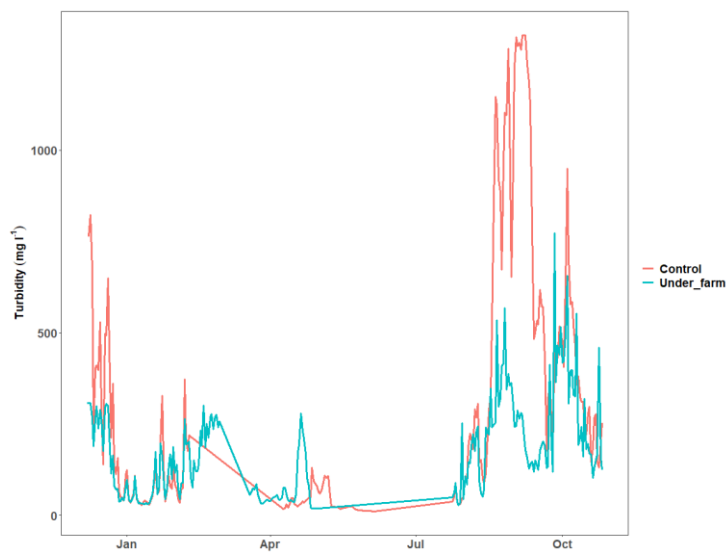


Figure 43 Turbidity (mg of suspended matter per l) recorded under the test farm in Ventry Harbour and a control station located ~100 m to the south. The values are transformed from native NTU values after calibration in the laboratory. Deployment was intermittent between January and October 2019 with some data gaps.

Eastern Schelde test farm

3.1.9 Biological footprint

3.1.9.1 Sampling campaign 1

After the field investigations it was decided that benthic surveys are not relevant for this site. This is because of its terrestrial history (see Materials & Methods section) and other impacts, such as nearby shellfish farming. Grab samples have therefore not been further analysed in the lab (taxonomy and sediment characteristics), and video and photo material has been stored but not further analysed.

During the diving transects within the (harvested) farm, hardly any fish was observed. However, the baited remote underwater video system (BRUV) seemed effective in attracting fish within the first minute of deployment at the seaweed farm site. The fish assemblages were dominated by horse mackerel (*Trachurus trachurus*) and big-scale sand smelt (*Atherina boyeri*). Fish number peaked at about 5 to 6 minutes. The visibility worsened throughout the deployment and therefore the presented number of horse mackerel is likely an underestimation. In addition, three types of jellyfish were identified: sea gooseberry (*Pleurobrachia pileus*), barrel jellyfish (*Rhizostoma pulmo*) and moon jellyfish (*Aurelia aurita*). Maximum number of fish visible in the video frame at any time during a 15 second interval was approximately 10 to 15 individuals.

3.1.9.2 Sampling campaign 2

Environmental conditions are presented in the annex. A total of 15276 individuals were found on the 12 ropes, representing (at least) 13 different species. The most abundant species on the inshore farm were amphipods from the genus *Jassa*, *Gammarus*, and *Caprella* (Figure 44). These species were found on seeded and non-seeded lines. *Caprella mutica* is the only introduced species that was found in this samples, all other species are native to the Eastern Scheldt. However, during the baseline sampling campaign conducted, other fauna species that were found in the farm were far less abundant than the before mentioned amphipod species. *Cyclopterus lumpus*, the only fish species that was found on the farm, was only found on the seeded lines, both in the centre and on the edge of the farm. Despite that an individual fish was observed during the sampling for sessile fauna (Table 14), and the promising results obtained during a pilot test in 2018 (see report by Tonk et al. (2019)), no fish were observed when the BRUV system (baited camera) was deployed in April 2019.



Figure 44 Most abundant species associated with the seaweed ropes: Amphipods a) *Jassa* sp., b) *Gammarus* sp. c) *Caprella mutica*. Eastern Schelde test farm.

The abundance identified on the ropes differed significantly between the two locations and between seeded and non-seeded lines (Figure 45; Table 14). Highest numbers of individuals were found on the non-seeded lines on the edge of the farm. A factor explaining these results could be that the ropes on the edge of the farm were containing a significant biomass of wild seaweeds, in these wild seaweeds amphipods seemed to be very abundant. More individuals were found on the seaweeds grown in the outer edge than on the ones grown in the centre of the farm (Figure 45A). The difference between seeded and non-seeded ropes was dependent on the position in the farm. In the centre of the farm, significantly more animals were found on the seeded ropes than on the non-seeded ropes (Figure 45A). The opposite was observed on the outer edge of the farm (Figure 45A). Generally, species richness on seeded ropes was higher than on non-seeded ones (Figure 45B) at both locations in the farm.

A two-way ANOVA was conducted that examined the effect of location and presence of seeded seaweed on fauna abundance and species diversity. For fauna abundance there was a statistically significant interaction effect between location and presence of seeded seaweed, $F_{(1, 8)} = 21.473$, $p = .002$. The main effect of location was also statistically significant ($p = .000$), while the main effect of the presence of seeded seaweed was not statistically significant ($p = .340$). For species diversity, there was a statistically significant interaction effect for location and presence of seeded seaweed as well, $F_{(1, 8)} = 10.268$, $p = .012$. The main effect for location is not statistically significant ($p = .316$), while the main effect of the presence of seeded seaweed is statistically significant ($p = .001$).

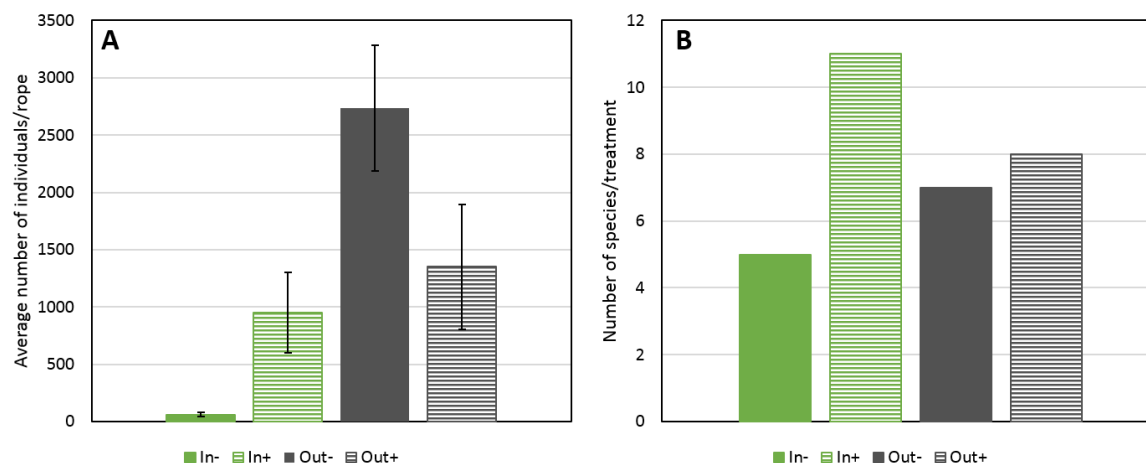


Figure 45 In = seaweed cultivated in the middle of the farm (N=3). Out = seaweed cultivated on the outer edge of the farm. - = non-seeded line. + = seeded line. A. Average number of detected individuals per rope. B. Total number of detected species per treatment. Eastern Schelde test farm.

Table 14 Fauna observed on the ropes from centre (in) and the edge (out) of the farm for seeded (+) and unseeded (-) droppers (N=3 for each treatment). Eastern Schelde test farm.

order	species	In-	In+	Out-	Out+
Amphipods	<i>Gammarus locusta</i>	2±3.5	158±25.9	321±176.7	178±33
Amphipods	<i>Jassa</i> sp.	41±10.3	731±387.8	1859±499	1148±533
Amphipods	<i>Caprella</i> sp.	14±16.1	46±42	547±174.8	20±9.8
Isopods	Unknown species			1±0.6	
Isopods	<i>Idotea balthica</i>		2±1	2±3.5	1±1
Tanaidacea	Unknown species		1±1.2		1±0.6
Decapoda	<i>Caridea</i> sp.		1±0.6		
Worms	<i>Leptoplana tremellaris</i>		1±1.2		
Worms	<i>Lepidonotus squamatus</i>	1±1.2	6±6		1±0.6
Worms	Round worm sp.	5±3.5	4±1.7	2±2.3	2±1
Fish	<i>Cyclopterus lumpus</i>		4±2.5		1±0.6
Bivalves	<i>Mytilus edulis</i>		1	1±1.2	
Biofouling				x	x
# individuals		62±18.6	948±351.1	2732±548.3	1350±541.8
# different species		5	11	7	8

3.1.9.3 Sampling campaign 3

The abundance was three to four times higher on ropes that were deployed in the beginning of the culture cycle, and diversity was also higher for the ropes deployed in November (Table 15). Although a total of 44 different species have been observed, it was clear that two phyla were dominating: Arthropoda and Mollusca. Within the phylum Arthropoda, *Jassa* spp. (*J. marmorata*, and *J. falcata*) were dominant while *Mytilus edulis* was the most abundant and frequently recorded Mollusc species associated with the seaweed ropes. *Jassa* spp. abundance varied between 15024 ± 5110 for ropes deployed in November and 4687 ± 1877 for ropes deployed in February. Abundance of *Mytilus edulis* also decreased from ropes deployed in November (25578 ± 6918) to February (6155 ± 2561). It thus seems evident that immersion time increases the biodiversity estimates.

Table 15 Summary of fauna observed on the ropes (1m) deployed in respectively November and February, Eastern Schelde test farm.

phylum	Abundance November	Abundance February
<i>Annelida</i>	436 ± 14	102 ± 36
<i>Arthropoda</i>	15164 ± 2971	5328 ± 1494
<i>Bryozoa</i>	165 ± 53	45 ± 57
<i>Mollusca</i>	23344 ± 6739	6164 ± 2085
<i>Nemertea</i>	48 ± 34	0
<i>Polychaeta</i>	560 ± 161	1 ± 2
Total abundance	43927 ± 10860	11640 ± 3353
Total number of species	39	23

3.1.10 Abiotic parametrisation

3.1.10.1 Temperature

Daily average temperature was similar at 0, 1 and 2 m depth (Figure 46). Daily average temperature raised from 7°C in the end of February to up to 15°C in the end of April. It was also similar in the middle of the farm and the outer edge, with an exception of the surface temperature in April (Figure 46A).

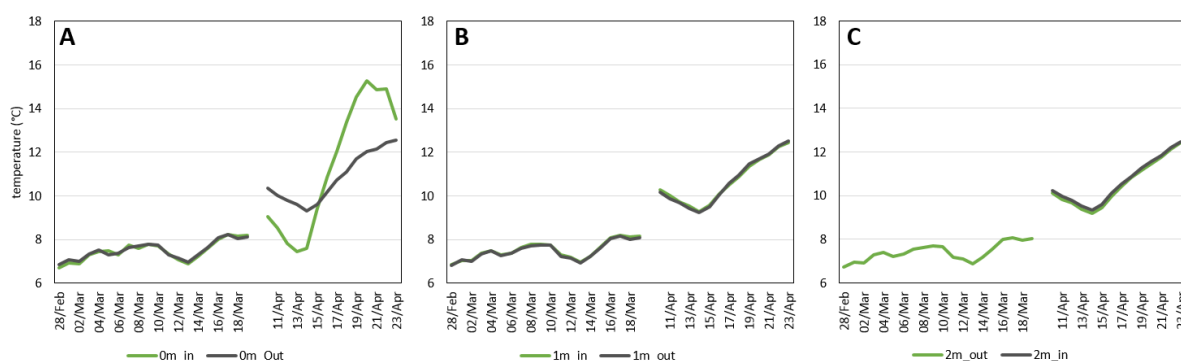


Figure 46 Daily average temperature measured by HOBO loggers. A: Daily average temperature at the surface in the middle (green) and outer edge (black) of the farm. B: Daily average temperature at 1m depth in the middle (green) and outer edge (black) of the farm. C: Daily average temperature at 2m depth in the middle (green) and outer edge (black) of the farm. Eastern Schelde farm.

3.1.10.2 Light

The measured daily average irradiance differed significantly in the middle and outer edge of the farm (Figure 47). In April, irradiance was higher in the outer edge of the farm (black) than in the middle (green), especially at the surface and at 1m depth (Figure 47B). This result suggests that the growing seaweed biomass and the cultivation structures decrease the available amount of light.

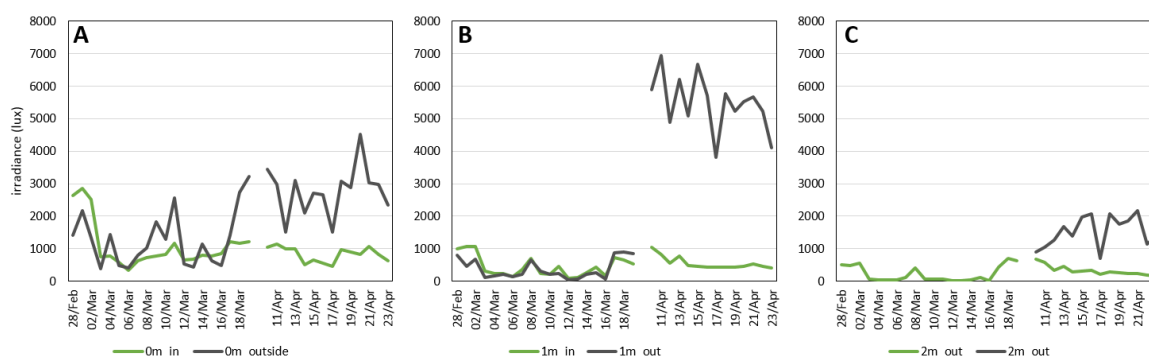


Figure 47 Daily average irradiance measured by HOBO loggers. A: Daily average irradiance at the surface in the middle (green) and outer edge (black) of the farm. B: Daily average irradiance at 1m depth in the middle (green) and outer edge (black) of the farm. C: Daily average irradiance at 2m depth in the middle (green) and outer edge (black) of the farm. Eastern Schelde farm.

3.1.10.3 Current speed

Current speed differed at the two measurement locations (Figure 48). In agreement with current measurements obtained at the Irish test site in Ventry Harbour, currents at the Schelphoek farm were higher in the outer edge compared to the middle, suggesting that the current conditions may be altered by the growing seaweed biomass and farming structures.



Figure 48 Daily average current speed in the middle (green) and outer edge (black) of the farm

3.1.10.4 Nutrients

Dissolved inorganic nitrogen (DIN) concentrations in the seawater increased from December 2018 to February 2019 (Figure 49). In December 2018, DIN concentrations were slightly higher outside of the farm, whereas in January and February 2019, they were higher in the middle of the farm.

Phosphate concentrations in seawater were similar from December 2018 to February 2019, only slightly higher on the outside of the farm in December 2018, whereas in January and February 2019 higher concentrations were detected in the middle of the farm (Figure 49B).

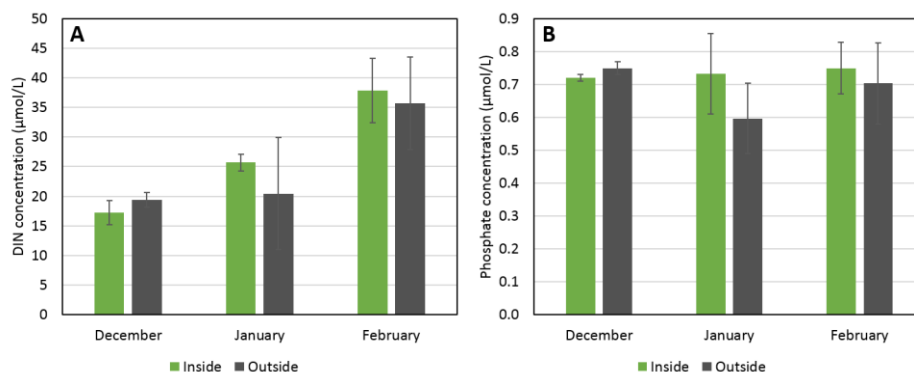


Figure 49 Nutrient concentrations (μmol/L) in the Eastern Scheldt (N=3). A. Dissolved inorganic nitrogen (NH_4^+ ; NO_2^- ; NO_3^-). B. Phosphate.

4 Discussion

One of the main objectives of the work conducted at the two test farms, as part of GENIALG's work package 6.1, was to “monitor biodiversity in and around the reference farms, to assess effects of this new habitat and increase of seaweed biomass, on diversity of other species, including commercial fish and unwanted invasive species”.

The present study represents a comprehensive account of the temporal and spatial variability in environmental conditions under and near two medium sized seaweed farms located in Europe: a 1.8 ha farm in the southwest coast of Ireland and 1 ha farm in the southwest coast of The Netherlands. The survey approach specifically focusses on identifying trends that could be linked to direct effects from the presence of cultivated seaweed and ancillary structures as the drivers of that variability. The aim of the surveys conducted between 2017 and 2019 was to demonstrate that cultivated kelp can replicate the function that wild seaweeds can have as keystone or foundation species and ecosystem engineers, altering the biophysical environment through their presence and their role in ecosystem functions and chemical processes, e.g. nutrient removal, energy dampening, food web dynamics, effects on seabed composition and organic matter deposition, shading effects and effects on biodiversity. The study thus contributes to increase the evidence base that can be used in informed a seaweed specific marine licensing system in Europe and elsewhere. The outputs contributed to inform the advice for monitoring including in Deliverable 6.9 of GENIALG (Tett et al 2021).

Adding to the results of recent evidence-based studies from Sweden (Hasselström et al. 2018; Visch et al. 2020b, c) and previous work from Ventry Harbour, Ireland (Walls et al. 2016, 2017a) this study increases the knowledge on the environmental effects and potential ecosystem services provided by seaweed. These results were used in other reports published by GENIALG, namely carrying capacity and nutrient uptake models (Broch et al, in prep) and ecosystem service valuation (to be made available from www.genialgproject.eu). The results obtained are in general agreement with desk-based studies (Wood et al. 2017; Hasselström et al. 2018; Campbell et al. 2019) and site-specific studies that report a largely positive effect on the environment, with cultivated kelp promoting, supporting and regulating ecosystem services. See references in Wood et al. (2017) and Campbell et al. (2019) and more recently the studies by Visch et al. (2020c). By increasing biodiversity, the farms play a role in local food web dynamics, playing a nursery role for juvenile fish and other commercial fish species and protecting autotrophic benthic habitats such as seagrass beds from direct physical impacts, e.g. potting, trawling and anchoring. The indirect positive effect that seaweed farms have in promoting the conservation of natural seaweed habitats by removing pressures from them by direct

harvesting should also be considered. While there is history of impacts from shallow seaweed aquaculture in Asia and Africa largely a result of nutrient competition, siltation and direct abrasion of benthic communities, (e.g. Eklöf et al. 2005, 2006 and references therein), these effects were not recorded in the type of offshore aquaculture sites that are common in Atlantic Europe, i.e. those considered in this study.

Effects of seaweed aquaculture on biodiversity

From the qualitative and quantitative biodiversity data collected in the test farms, via direct observations by divers and sampling, we conclude that seaweed cultivation increases species richness and contradicts the idea that one-species aquaculture is somewhat similar to barren monocultures found in terrestrial habitats. Biodiversity recorded in both test farms seemed however strongly affected by a broad variety of parameters including the timing of seeding, timing of harvest, and position within the farm (Walls et al. 2017b; Visch et al. 2020b). Biotic richness and diversity in seaweed dominated habitats are usually not dependant on the number of seaweed species, although the composition of the faunal assemblages is (Bates and DeWreede 2007; Walls et al. 2016). This study provided evidence that the biodiversity of the faunal assemblages associated with the two kelp species studied *Saccharina latissima* and *Alaria esculenta* is similar, not very high compared to other biogenic habitats (for example *Modiolus modiolus* reefs, see Fariñas-Franco et al. 2013) but on par with natural kelp beds (Walls et al. 2016).

It should be emphasised that although, in theory, the test farms were monocultures most of the longline samples had a variety of other seaweed present within the holdfast of the cultivated species, largely red filamentous taxa. In the Eastern Schelde study, biodiversity differed between lines cultivated in the centre and outer edge of the farm, but differences between the seeded and non-seeded lines were also detected as did the study from Ventry Harbour. Ropes that contained seaweed showed higher species richness, confirming the ecosystem engineering effect of the seaweed as the habitat forming species, in addition to that from the cultivation structures.

4.1.1 Cultivated seaweed and growing structures as habitat for marine fauna

Results from infaunal and epifaunal analysis of cultivated seaweed samples are similar to those reported by others (Walls et al. 2016; Visch et al. 2020b and references therein) showing a community dominated by tube building amphipods, the dominant element of the fouling community in seaweed grown in sheltered sites, followed by bryozoans, hydroids and colonial tunicates (Figure 50). This was also shown for three sites in Norway where number of species in *S. latissima* cultivation varied between 49 and 64 per plant, and abundance between 22750-75833 individuals m⁻¹ (Christie et al.

2009). The significantly higher abundances and number of species found in seeded droppers compared to unseeded controls confirms the cumulative habitat forming effect of the grown seaweed as the foundation species. The timing of the deployment of the seeded lines in autumn or winter allows for maximum growth of the kelp before seawater temperature increases facilitating growth and settlement of planktonic larvae and colonisation of these new habitats.

Taxonomic analysis of the dropper and longline samples from Ventry and Schelphoek (this report), showed that the communities associated with cultivated seaweed are largely dominated by amphipods and mussels, initially during the growing season structured by the seaweed itself as the habitat forming taxa, and later after the summer solely by mussels and remaining holdfast. In Ventry, *M. edulis* was clearly more abundant in *S. latissima* samples in June compared to *A. sculenta* in the same month, but different years. Factors including temperature affect the annual spat recruitment events for *M. edulis* and the difference might not be due to either kelp species being preferred by swimming *M. edulis* as a settlement substrate (Bayne 1964; Filgueira et al. 2014; Capelle et al. 2019). Nonetheless, it is possible that the more complex surface of the *S. latissima* is indeed more attractive for pediveligers as a settlement substrate compared to the smoother fronds of *A. esculenta*, which could be the reason for the differences in biodiversity and faunal assemblages, especially the abundance of bryozoans and other epifaunal species, associated with each kelp species (Figure 50). For example, the sea mat *Membranipora membranacea* is an epiphytic bryozoan whose presence in the algae fronds can cause the weakening and loss of blades and canopy density (Dixon et al. 1981; Denley and Metaxas 2017), and produce a decrease in the value of the kelp biomass for the industry (Førde et al. 2016). The presence of this bryozoan was notable among the *S. latissima* samples collected in 2018 but was anecdotic on *A. esculenta* samples collected in 2019.

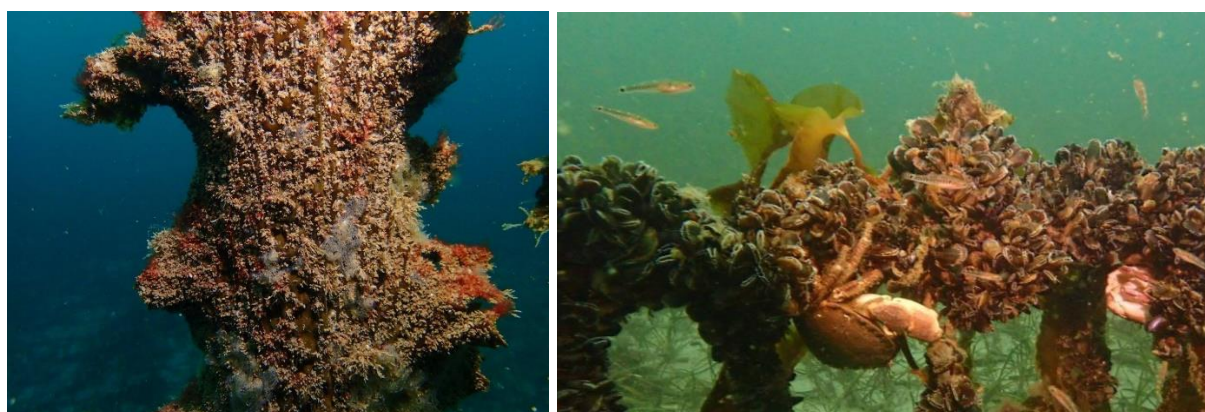


Figure 50 Left: Frond of cultivated sugar kelp *S. latissima* heavily colonised by amphipod tubes, bryozoans, and colonial tunicates. Longlines left unharvested over the summer periods developed into a biogenic habitat formed by mussels and left over kelp holdfast. Notice the presence of two-spotted gobies *G. flavescens* and edible crab *C. pagurus*

The effect of surface complexity on settlement of planktonic larvae is well known (e.g. Buck and Buchholz 2005) and in kelp species with rougher surface, e.g. *Laminaria hyperborea*, have a more diverse epibiotic community compared to species with smoother surface such as *L. digitata* (Blight and Thompson 2008; Walls et al. 2016); primary and secondary laminae in *Ecklonia radiata* (Jennings and Steinberg 1997). Biodiversity in the frond could be strongly influenced by the presence of different bacterial biofilms, known to affect larval settlement behaviour, covering the surfaces of either kelp species (Orlov 1997).

These differences in survey timing possibly explain mussels being absent in the May 2020 East Schelde samples compared to the previous season in April when they were found in high abundances. The sampling moment will influence the abundance of certain species, such as *Mytilus edulis*, which reproduce between March and May, depending on water temperatures, and settle from April to June (Capelle et al. 2019; Capelle 2020). Therefore, the fauna sampling in 2019 was likely too early to find any *Mytilus edulis* and possibly other species, while in 2020 mussels were very dominant on the seaweed ropes.

The relationship between biodiversity and the presence of habitat forming mussels and kelp, even at low densities, has been widely documented in the wild for mussels (Fariñas-Franco et al. 2018b; Coolen et al. 2020) and kelp (this study; Christie et al. 2009; Walls et al. 2016). The presence of two habitat forming ecosystem engineers, the cultivated kelp and the blue mussels, has a cumulative effect in the biodiversity as structural habitat complexity increases. The spaces in the holdfast and mussel clumps bound by byssal threads, are used by crevice fauna while the sediment accumulated serves as suitable habitat to detritus feeding infauna. The kelp fronds and mussel shells increase the surface available for settlement of epifaunal species (Fariñas-Franco et al. 2013; Teagle et al. 2018). Wijsman et al. (2009), showed that shellfish aquaculture attracts a broad variety of species and encountered 55 taxa on suspended mussel cultivation (n=6) and 49 on mussel spat collectors (n=10) situated in the same in-shore estuary as the seaweed farm presented in this research. The number of species found in our study roughly comply to the numbers found in those studies.

The dropper and longline samples collected in October 2018, were not fully processed at the time of the writing of this report but field observations and partial sorting in the laboratory confirmed that the mussel 'reef' community that developed onto them over the summer is used as nursery habitat by decapods of commercial importance, predominantly edible crab *Cancer pagurus*, velvet swimming crab *Necora puber* and squat lobster *Galathea* sp. Thus, the mixture of kelp holdfast and tightly packed mussels replicate the complex interstitial structures of natural mussel and kelp beds providing the

same opportunities for refuge and food these crab and lobster species seek in the wild (Commito and Boncavage 1989; Lindsey et al. 2006; Bertolini et al. 2018; Commito et al. 2018) (Figure 50).

Nonetheless, time is an important factor on the longevity of the impact on biodiversity. With cultivation substrates being present only for a limited period (late autumn until late spring) and fully extracted during the harvest, the increase of biodiversity in the water column is primarily boosting opportunistic species unless the holdfast and mussels are left on the ropes. To minimise these impacts, the existence of suitable habitats that can accommodate migrants displaced by the removal of the cultivated kelp should be considered in an ecosystem-based approach to site selection of seaweed farms. In the case of the Ventry site, with seagrass and kelp beds near the farm, it is likely that these impacts are minimal considering that the fish observed using the kelp longlines had probably immigrated from nearby habitats where they were observed in high abundances. These local migrations have indeed been recorded for both fish species recorded in the Ventry site, *G. flavescens* and *P. pollachius* (Heino et al. 2012; Furness and Unsworth 2020).

Consideration needs to be given to leaving ropes or lines untouched or ‘fallowed’ for maximum biodiversity increase and ‘maturity’ so they remain a habitat. Bak et al. (2018) found that partial harvesting of the *S. latissima* and *A. esculenta* biomass could be very advantageous and economically beneficial, as there was no need to reseed the lines the following seasons due to the natural re-growth of the kelp biomass. Alternative coppicing or harvesting the blades leaving the holdfast to promote natural regeneration and recruitment, or growing algal polycultures that can be harvested at different times of the year (Campbell et al. 2019) can be an efficient and economically beneficial alternative to provide raw produce, removing the need for cleaning and reseeded and maximising biodiversity and ecosystem service provision (Bak et al. 2018).

4.1.1.1 Presence of invasive species

Non-native species were recorded in both test sites. In Ventry Harbour *Caprella mutica*, the Japanese skeleton shrimp, was the only non-native taxa observed while in the East Schelde site there were at least four non-native species found associated with the structures: the Japanese knotweed *Sargassum muticum*, *C. mutica* (itself associated with the non-native *S. muticum*), the Pacific oyster *Crassostrea gigas* and the globally invasive tunicate *Styela clava*, all species that originate in the NE Pacific region (Goldstien et al. 2010, 2011).

Caprella mutica was frequently recorded in high abundances and was the dominant caprellid and third most abundant invertebrate (second most abundant if *M. edulis* were excluded) in longline and dropper samples, predominantly associated with the kelp and filamentous red seaweeds growing on

the longlines and substantially outnumbered native caprellids. This caprellid amphipod native to the sub-boreal waters of the north-west Pacific Ocean often found associated with areas of human activity; marinas, harbours and aquaculture sites (Buschbaum and Gutow 2005; Ashton et al. 2007, 2008). Its dispersion is likely linked to vessel movements (Cook et al. 2007) and the first record of the species in Irish waters was in 2004 in the west coast associated with salmon farms (Tierney et al. 2004; Minchin and Holmes 2006). Due to its possibility of attaching to boat hulls, engines, poles, etc. its range spread in the past few years and is present elsewhere on Irish coasts (Cook et al., 2007). This species is also inhabiting algae and mussel or tubeworm reefs and is a strong competitor of the native European caprellid *Caprella linearis* both under laboratory and field conditions (Cook et al., 2007).

Of note is the presence in the East Schelde farm of the invasive non-native sea squirt *Styela clava*, which originates in the NW Pacific (Goldstien et al. 2011). The first record of the species in European waters was observed in the Plymouth Sound (UK) in 1952 and since then it has spread throughout the NE Atlantic (Lützen 1998). The species was not recorded in the Irish test but was observed by divers during the baseline surveys carried out in the East Schelde seaweed farm in 2018, commonly attached to clumps of *Sargassum muticum*, itself an invasive species and a known vector of dispersal for *S. clava*. *Styela clava* is fast growing, large and can outcompete native tunicate species (Lützen 1998).

The presence of Pacific oyster *Crassostrea gigas* in the farm was probably serendipitous, growing on strands of *Sargassum* spp. or being caught in them if dislodged from the nearby shoreline, where they were commonly observed. Pacific oysters *C. gigas* are well established in The Netherlands, expanding rapidly and creating extensive biogenic reef structures. Although it is considered potentially an invasive species, reproducing thanks to water temperatures reaching their spawning and recruitment threshold (Guy and Roberts 2010), their role in the functioning of the ecosystems they colonise has been generally regarded as positive, promoting water quality and enhancing biodiversity, with no obvious effect outcompeting other bivalves. Additionally, they have been known to facilitate the settlement of native mussels *M. edulis* and endangered native oysters *Ostrea edulis* (Troost 2010; Walles et al. 2016; Zwerschke et al. 2016; Christianen et al. 2018)

4.1.2 Seaweed farms as essential fish habitats

Seaweed habitats typically act as a nursery for juvenile fish (Steneck et al. 2002; Smale et al. 2013; Seitz et al. 2013; Lotze et al. 2019) and seaweed farms may therefore replicate that function. Fish can be attracted to the physical structure acting as fish aggregation devices (FADs), as they do to mussel farms (Mascorda Cabre et al. 2021), and some studies have documented this effect, describing the impact of seaweed farming in fish assemblages (Bergman et al. 2001). Although the positive effects of

seaweed farms acting as nurseries for fish have been predicted by extrapolating the observations from natural kelp habitats as essential fish habitats, e.g. (Wood et al. 2017), there are few studies that have specifically focus on documenting these effect with some providing anecdotal evidence (e.g. (Visch et al. 2020a reporting the presence of lumpfish *Cyclopterus lumpus*). The surveys conducted at Ventry Harbour provides solid evidence of the use of a kelp farm as a nursery area by fish juveniles of at least two species frequently recorded in relatively high numbers, pollack *Pollachius pollachius* and two-spotted goby *Gobiusculus flavescens*, as well as lumpfish *C. lumpus*.

Two-spotted gobies *G. flavescens* are usually found in kelp and seagrass habitats all-year round although are more common in the summer months when additional recruits arrive from deeper waters (Fosså and Helge 1991; Furness and Unsworth 2020). The presence of *G. flavescens* was noted in the longlines during at peak kelp biomass, and considering the abundance of these fish in the seagrass habitats underneath during the summer, it is possibly both habitats are connected, fish utilising the kelp farm opportunistically as a feeding ground, for reproductive purposes (Smith and Heemstra 1986), and as a refuge from predators such as *P. pollachius*, rather than as an FAD. The diet of *G. flavescens* consists predominantly on small invertebrates, such as amphipods and copepods (Utne-Palm 1999), the dominant fauna recorded in the cultivated kelp. There is other omnivorous fish that in addition to the fouling organisms include in their diets small fish and, therefore, would benefit from the fish species that are attracted by the macroinvertebrate assemblage. For example, the lumpsucker *C. lumpus* and pollack *P. pollachius* frequently recorded in the farm and the seagrass beds nearby during the summer and early autumn, feed both on small crustaceans, molluscs, and small fish (Fosså and Helge 1991; Imsland et al. 2015). The population dynamics of pollack, a species of commercial importance, is generally poorly known but evidence from the few studies that exist suggest it undergoes seasonal migrations from deeper water where they have their spawning and feeding grounds to shallower water where the juveniles are usually frequently found in the summer, associated with kelp and seagrass habitats (Armannsson et al. 2007; Heino et al. 2012; Pita et al. 2018; Furness and Unsworth 2020).

Various techniques are available to investigate marine biodiversity of mobile fauna. For a long time, studies have been based mainly on extractive methods, such as fishing or dredging. However, due to technical progress in video cameras, sensors, battery life and information storage, novel techniques have become accessible and affordable for most users during the last decades (reviewed by Murphy and Jenkins 2010). Camera imagery has thus become a powerful tool for studying marine biodiversity on different scales, from individuals to entire ecosystems. Despite these promising applications, as well as initial pilot tests at our study site (Tonk et al. 2019), attraction of fish was not confirmed with

baited camera systems in the Schelphoek farm. The test of the baited camera in the Ventry site was limited to one opportunistic deployment in July 2019 for a limited period (2-hour deployments under the farm and at the northern control station ca. 200 m from the farm). Although the diversity and abundance of fish recorded were not very high, possibly a result of the bait used, it did produce some results with shoals of sandeel *Ammodytes* spp. and two lesser-spotted catshark *Scyliorhinus canicula* recorded under the farm.

4.1.3 Effects on benthic habitats and biodiversity

4.1.3.1 Benthic autotrophic habitats: seagrass beds

One of the main concerns about the potential negative effects that seaweed aquaculture can have on benthic autotrophic habitats (e.g. kelp beds and seagrass meadows) is light limitation due to shading and nutrient competition (Wood et al. 2017; Campbell et al. 2019). Some studies have found seaweed aquaculture having significant negative effect on seagrass habitats, reducing seagrass biomass and shoot density (Eklöf et al. 2005, 2006b, a). Those studies, however, were carried out in tropical lagoons in Africa and Asia, where longlines and supporting structures sit in very shallow water only suspended 0.15 m above the seagrass causing a direct impact through shading and abrasion. Those conditions are not comparable to those generally found in the majority of offshore kelp farms in Europe and North America, where longlines usually float just under the surface, are spaced (in Ventry approximately 15m between each other) and placed in deeper water.

The test farm in Ventry Harbour was unique in being located in a seagrass bed, a habitat of conservation importance and high biodiversity (Dale et al. 2007) sensitive to anthropogenic stressors, including physical abrasion (Duarte 2002; Dale et al. 2007), providing an excellent opportunity to assess the potential impacts of seaweed aquaculture on these habitats and the potential ramifications in the granting of licenses and site selection process. Previous studies in the area, limited in extent and replication, found no effect from the farm on seagrass biomass and thus concluded that the seaweed growing operations were not a threat to the conservation status of the seagrass beds from the seaweed farm operations (Walls et al. 2017a).

The surveys conducted in the area as part of the GENIALG project (this report) were more extensive and ambitious in scope and included measurements of seagrass coverage as well as biomass and irradiance. The results showed seagrass coverage and biomass under the farm were as high, if not higher, compared to distant stations where seagrass became sparser and patchier. The distribution is likely affected by factors known to limit the extent of seagrass e.g. light attenuation with increasing

depth turbidity caused by more exposed conditions away from the protection of the farm, with bottom swell increasing sediment resuspension (Van Der Heide et al. 2007).

While Walls et al. (2017a) found no effect in biomass from the seaweed farm the authors did not investigate the potential factors, for example by monitoring light availability and turbidity. While Visch et al. (2020a) found reductions of up to 40% in light availability under a small seaweed farm in Sweden, this attenuation had no effect on oxygen uptake from primary producers and no apparent effects on mobile epifauna or infauna, which were richer compared to the controls. The monitoring we carried out in Ventry Harbour found the opposite effect, with higher irradiance (as PAR) values under the farm compared to the controls, even those in shallower waters. It is likely that the seaweed farm acts as a sediment trap based on the lower turbidity and sediment deposition levels recorded under it compared to the highly dynamic areas outwith the farm. These effects are likely caused by the observed reductions in current flows within its boundaries, an effect reported in kelp farms elsewhere (Liu et al. 2016). These observations indicate the farm promotes water clarity, confirming its role as a sediment trap hypothesised by others (Wood et al. 2017 and references therein) increasing light availability (PAR) for primary producers underneath, which benefit for enhanced water clarity, less siltation and potential smothering in an embayment known for its turbidity and sediment accretion (Author Pers. Obs.).

Offshore seaweed farms can have an additional, positive effect on local ecosystems, as deterrents to boat navigation, anchoring and bottom trawling, sources of physical impacts responsible for seagrass habitat declines worldwide (Riesen and Reise 1982; Neckles et al. 2005; La Manna et al. 2015; de los Santos et al. 2019). In the 1980s, seagrass beds were reported as the dominant habitat across the whole Ventry Harbour embayment (Whelan and Cullinane 1985). Although the present study did not attempt to map the current extent of the seagrass beds in the bay, based on the data collected, we believe the habitat has been substantially reduced, now limited to the south of the bay, roughly to the area where the farm is located and between the farm and the shoreline. Ventry harbour is a known boat anchoring point and potting and other forms of fishing due occur in the bay. It is possible that, in addition to its effect as a sediment trap, the farm is acting as de facto exclusion zone for other vessels promoting the protection and conservation of sensitive habitats as other studies have demonstrated, e.g. recovery of biodiversity and Polychaete reefs within windfarms ((Pearce et al. 2014; Coates et al. 2016)).

4.1.3.2 Benthic mobile fauna

In Ventry Harbour, mobile species recorded during the seabed habitats surveys were similar to those observed directly associated with the cultivated seaweed overhead and included predominantly juvenile pollock *P. pollachius*, and two-spotted gobies *G. flavescens*. Their utilisation of the habitat did not appear to be affected by the farm, but their appearance and distribution was dependent on the extent of the seagrass habitats and season. Most fish recorded on site first appear in April as isolated individuals or in small groups but as water temperature (and seagrass biomass) increased their numbers increased, especially *P. pollachius* which were recorded in large (e.g. over 100 individuals) shoals in the seagrass meadows (Figure 51). Flatfish, sand gobies, sand-eel *Ammodytes* sp. were limited to the sandy areas to the north and east of the farm while opportunistic scavengers such as lesser-spotted catshark *S. canicula* and the near threaten thornback ray *Raja clavata* (Ellis 2016) were more ubiquitous, but more commonly encountered under the farm and its vicinity (50 m radius) during most dive surveys. Goldsinny *Ctenolabrus rupestris*, and Corkwing wrasse *Symphodus melops* were common in the area but limited to the kelp dominated habitats outside the boundaries of the farm.



Figure 51 Pollack *Pollachius pollachius* juveniles recorded in seagrass meadows ca. 25 m from the southern edge of the Ventry Harbour seaweed farm. Snakelock anemones *Anemonia viridis*. July 2018.

Other mobile fauna recorded under the farm and its vicinity included spider crab *Maja brachydactyla* and edible crab *C. pagurus* with their distribution either influenced by the increased feeding opportunities provided by the seagrass meadows and the occasional longline or by seasonality, while the masked crab *Corystes cassivelaunus* presence was seasonal, and again, habitat driven, preferring the sandy areas to the east of the farm. The presence of fish and mobile fauna under and near the

seaweed farm with no apparent negative effect but rather an attraction effect which has been reported in other seaweed farms of similar size by Visch et al. (2020a).

The effect of seaweed farms as fish nurseries has been documented before (Bergman et al. 2001; Zemke-White and Smith 2006). It is possible that the nearby presence of seagrass and kelp habitats represent donor sites of fish that found the farm suitable habitat and feeding ground. While the fate of these fish after the harvest is potentially a negative effect it is plausible that the nearby habitats can absorb the fish once the seaweed is removed. The effect of the harvest needs to be considered with attention given to seaweed farms located in areas that lack suitable habitats to absorb displaced fish populations after the seaweed biomass is removed.

4.1.3.3 *Benthic infaunal communities*

It was not possible to quantify spatial and temporal variability in benthic infaunal communities, and the potential effects of the seaweed farms studied, due to logistical and time limitations. While replicates for sediment indicated that the organic component as POM suggest that the farm might have a neutral effect on benthic communities with organic matter pathways having a variety of origins, not solely from the farm, but likely from seagrass and kelp beds that dominate adjacent areas. Judging by visual observations obtained during the surveys and the results of the sedimentology and organic matter analysis showing homogeneity across the studied area, we can infer the benthic infaunal communities largely influenced by those environmental factors, are unlikely to be subject to negative effects from the seaweed farm. Existing quantitative studies indicate the effect by seaweed aquaculture on benthic communities is not only neutral but positive, with significant increases in benthic quality indices compared to control stations (Visch et al. 2020a) and therefore we can hypothesise that is also the case in Ventry Harbour.

No evidence of drift or decaying kelp that could have originated from the seaweed aquaculture activities was reported during the surveys in Ventry Harbour but numerous fragments of kelp *Laminaria* spp., *Sacchoriza polyschides* and *Sachharina latissima* along with drift seagrass blades were commonly recorded in the autumn and winter surveys following storm events. The farm nonetheless provided organic inputs from the epifauna growing on the longlines (anemones, red seaweeds, mussels and associated taxa) which also attracted scavengers and opportunistic predators and other pelagic fauna in agreement with the results reported from Sweden by Visch et al. (2020a). However, it should be noted that mussels present on the longlines in both test farms and nearby mussel farms in the Eastern Schelde site, are also a source of organic matter. Mussels produce pseudo faeces that might alter benthic communities (McKindsey et al. 2011; Mascorda Cabre et al. 2021) although these

interactions and potential cumulative effects are likely to be site specific, with impacts depending on the area and volume occupied by those activities and local environmental conditions.

Impact on abiotic benthic parameters

4.1.4 Seabed sediment size and organic matter

Aquaculture installations interfere with local hydrodynamic patterns increasing drag and altering current speed and direction, generally reducing water flow (Grant and Bacher 2001; Fan et al. 2009; Aldridge et al. 2012). As a result of these altered current patterns, some studies have found that seaweed farms can potentially alter local sediment dynamics, and thus the sediment structure and composition of the seafloor in the surroundings (Bos et al. 2007; Petersen et al. 2008; Liu et al. 2016). The potential knock-on effects of such changes could be an increase in turbidity that can affect benthic primary producers or changes in the biodiversity and structure of infaunal assemblages.

This study, however, demonstrated that sediment under and near medium sized seaweed farms can remain unaffected by their presence. In Ventry Harbour, no significant temporal variation was found in sediment textural parameters (mean grain size, skewness, and kurtosis) in agreement with similar studies from China where sediment did not vary during seeding, harvesting or after harvesting (Liu et al. 2016).

There was some indication of localised increases in organic matter content in seabed sediments under and near the Ventry farm at peak biomass and post-harvest surveys. Others (Eklöf et al. 2006a) have also recorded organic enrichment in shallow seaweed farms in the tropics but the majority of studies have found no effects on organic matter content in seabed samples as a result of seaweed farming (Buschmann et al. 2014; Liu et al. 2019). It is difficult to ascribe these spikes in organic matter content to the farm, and the most likely source are the biogenic habitats (e.g. seagrass) that dominate the area, or the abundance of drift kelp recorded after autumn and winter storms, which are the most likely sources of organic carbon and detrital material. Organic matter in and around the test farm site (4%) were nonetheless low compared with those recorded in areas with finfish aquaculture, and similar to those recorded in other seaweed aquaculture sites (Liu et al. 2016). In comparison, the organic fraction from suspended solids in seawater samples was on average 13% at the centre of the site with values generally under 28% but as high as 48%. Ventry Harbour is a highly dynamic environment, the seafloor being subject to frequent bottom swell and turbid conditions as demonstrated by the readings obtained from the current meters. These hydrodynamic conditions might result in the quick removal of any organic material deposited by the currents and resuspension of sand. Based on the higher values of irradiance recorded under the farm and higher visibility records

from Secchi disk, and significantly lower sediment amounts captured by the sediment traps under the farm and low turbidity readings, it indicates that the seaweed is acting as a sediment trap retaining most of the suspended material observed in the water samples (Zhou et al. 2006; Wood et al. 2017).

4.1.5 Effects on tidal flow, water clarity and siltation

Most studies suggest cultivated seaweed are expected to act as a wave dampening device promoting coastal protection and minimising erosion of tidal flats, replicating the effects reported from natural kelp beds (Jackson 1997; Gaylord et al. 2007; Barbier et al. 2019). Similar to other studies that report a reduction in water flow within the boundaries of seaweed farms (Grant and Bacher 2001; Zeng et al. 2015; Liu et al. 2016) field data collected from both tests sites also indicated substantial reductions of current speeds within the seaweed farms. In Ventry Harbour, field data and hydrodynamic models predicted up to 31% reductions within the farm. However, current flow was diverted around the farm structure leading to current speed increases of up to 34% which aligns with the results of other models developed for seaweed farms in China showing an increase in current velocities around the farm and under the farm area (Grant and Bacher 2001; Zeng et al. 2015). In the case of the Ventry Harbour studies (this report) turbidity, suspended matter content and sediment deposition around the farm was higher compared to areas within the farm. Ventry Harbour is an embayment subjected to natural instability and the presence of the farm could further exacerbate these conditions in intertidal and shallow subtidal areas between the farm and the shoreline. Thus, these results could indicate that in fact an increase in coastal erosion is possible and should prompt to approach with caution claims of coastal protection as a de-facto ecosystem service provided by seaweed aquaculture farms, reinforcing the importance of site-specific studies and hydrodynamic models to inform the site selection and farm design process.

5 Conclusions

This study improves our understanding of seaweed farming impacts on surrounding ecosystems by using two farm sites to gather evidence. The surveys conducted at the test farms in Ireland and The Netherlands provide confirmation of the significant positive effect seaweed aquaculture has in local ecosystems, providing habitat for a diverse faunal community, and its role to support populations of commercially important fish and shellfish species. The findings represent evidence of the role of seaweed farming can play in the sustainable development of coastal economies and communities, by highlighting its role in supporting economies and maintaining healthy ecosystems and providing valuable services. This role aligns with Aichi Target 11 on marine biodiversity protection and UN Sustainable Development Goals (SDG) 2 food security and 14 on oceans as well (Le Gouvello et al. 2017), potentially helping to achieve EU targets for Good Environmental Status (GES; (Fariñas-Franco et al. 2014; Hasselström et al. 2018) and could play a role in the European Green Deal Strategy boosting the efficient use of resources by moving to a clean, circular economy, restore biodiversity and cut pollution (European Commission 2019).

Confirmation was provided on the increase in water clarity by the test kelp farm, the reduction of siltation and enhancement of biodiversity, directly through the provision of artificial structures for settlement and habitat and food by the cultivation of seaweed, and indirectly by facilitating the presence of autotrophic habitats of high conservation importance, i.e. seagrass beds. Further studies are encouraged in new and prospective seaweed farms, including the potential for integrated multi-trophic aquaculture (IMTA), to facilitate long-term ecological data representative of site.

Environmental monitoring as a requirement of seaweed-aquaculture specific licensing process could be a very valuable exercise in providing evidence to use as a benchmark to establish impacts against appropriate controls. Having appropriate temporal and spatial controls will help differentiate anthropogenic-driven change from seasonal and natural spatial variability. Every new site would be an opportunity to collate data to further increase the ability to detect impacts and effects and improve guidance. Monitoring and, especially, the development of full environmental impact assessments, could be necessary where the proposed sites are within marine protected areas and should follow robust Before After Control Impact (BACI) methodologies (Underwood 1992; Gray et al. 2006).

The recommendation is to establish the scale of the monitoring depending on the conservation value of the area. The establishment of distant control stations is crucial. A robust, future proof, baseline monitoring program can be established at benthic and water column levels. The main aim would be

to tackle known environmental interactions with 1) the benthic environment, focusing on biotic and abiotic elements; 2) water quality; 3) description of the biofouling communities including the detection of invasive non-native species; 4) genetic and pathogen interactions.

Although the deployment of infrastructure on the seabed would nonetheless require a license it would be necessary to establish if the moorings and boating activities (e.g. through anchoring) might affect fragile benthic communities, for example seagrass or other biogenic habitats of high conservation importance, in addition to any cumulative effects caused by the growth of seaweed itself. Besides direct physical effects, which should be avoided, the overall impact from the farm is expected to be minimal, or even positive, as observed during the GENIALG project. Monitoring can include the deployment of instrumentation to determine nutrient levels, temperature, PAR and currents. The position of sediment traps at regular intervals could help establish organic matter inputs into the sediment.

Follow up surveys along with seaweed growth monitoring surveys would help detect impacts following ecosystem models. The intensity of the survey protocols can be determined by the scale and dimensions of the farm and established following the first period of sampling. BACI approach and detection of spatial gradients can be accomplished by following survey design with sampling stations regularly spaced from the source of impact, with transects deployed along the direction of the prevalent currents and perpendicular to them. These surveys would incorporate sediment sampling for granulometric and organic carbon content analysis (1 replicate) and faunistic samples replicates. The latter would be used to define faunal assemblage composition and biodiversity indices and impact and natural variability investigated using univariate (GLMMS, GLAM or Permanova) and multivariate (PCA, MDA, PCO, LDA), to explain shifts in faunal assemblage composition caused by the farm.

In line with the recommendations provided by Tett et al. (2021) in GENIALG, we recommend pursuing research partnership between academia and industry to develop site specific environmental monitoring programs to inform ecosystem based management of existing and new aquaculture licenses. By incorporating environmental baseline and operational monitoring surveys as part of the licensing conditions risk can be minimised, ensuring provision of ecosystem services by the farm that can be monetised and claimed by the owner or operator. Supporting biodiversity as part of the farm management (e.g. coppicing, fallowing), promoting multi-species aquaculture and IMTA could be

facilitated by payment for biodiversity similar to the EU ‘Farming for Natura 2000’ programs² run in land-based agriculture. Seeking opportunities with other aquaculture activities, for example collocation of native oyster *Ostrea edulis* aquaculture in seaweed farms, could provide opportunities to maximise biodiversity and environmental restoration (Fariñas-Franco et al. 2018a; Pogoda et al. 2019). Following up on the results from GENIALG, continued BACI monitoring programs could help detect and quantify these ecosystem services, e.g. biodiversity enhancement, protection of natural habitats, role as carbon sinks and nursery habitats for commercial species, that could be used to highlight the environmental benefits of these activities, increase social acceptance, and support the expansion of the industry.

2

<https://ec.europa.eu/environment/nature/natura2000/management/docs/FARMING%20FOR%20NATURA%202000-final%20guidance.pdf>

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