

*Unlocking the potential of
macroalgae for a thriving
European blue
bioeconomy*



An assessment of ecosystem services provided by seaweed cultivation

SEAMARK DELIVERABLE 9.3

Sjókovin – Blue Resource



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SEAMARK DELIVERABLE 9.3: AN ASSESSMENT OF ECOSYSTEM SERVICES PROVIDED BY SEAWEED CULTIVATION

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Abstract

Seaweed cultivation provides ecosystem services, but their extent remains uncertain. SeaMark contributes to this knowledge by assessing three cultivation sites: one in the Faroe Islands, one in Norway, and one in France. Nutrient bioextraction and carbon sequestration were evaluated at all sites, while biodiversity impacts were studied in the Faroe Islands.

Data collection combined site-specific nutrient and carbon content with literature reviews. In the Faroe Islands, dislodgement rates were monitored over eight months and supplemented with 2015–2016 data. Biodiversity assessments were conducted through two large sampling events and literature comparisons.

The cultivated brown macroalgae *Saccharina latissima* had an estimated nutrient bioextraction potential of 2.5 kg nitrogen (N) per ton (t) wet weight (WW) in the Faroe Islands, 1.2 kg N t⁻¹ WW in Norway, and 1.9 kg N t⁻¹ WW in France. Additionally, the brown macroalgae *Alaria esculenta* in Norway removed 1.7 kg N t⁻¹ WW. Differences stemmed from varying nutrient and dry matter content.

The biodiversity assessment showed that continuous habitats (multiple years with same grow lines) had higher species richness and abundance (biodiversity) than cultivation structures that were removed annually, as it allows further development. A partially harvest method, where the holdfast and stem is left on the grow lines, allows for a continued habitat for mobile species even after harvest. More research is needed to understand the impact of harvest or partial harvest on biodiversity to confirm the benefits seen here. The impact of site exposure on biodiversity was tested, but no significant differences were found between species presence or abundance at semi-exposed and exposed sites.

Our carbon sequestration potential assessment estimated an uptake of 27 kg C (99 kg CO₂e) t⁻¹ WW harvested by Ocean Rainforest in the Faroe Islands, 12 kg C (45 kg CO₂e) t⁻¹ WW harvested by Algolesko in France, and 18 kg C (66 kg CO₂e) t⁻¹ WW harvested by Ocean Forest in Norway. Greater biomass loss was observed during winter storms in the Faroe Islands which increased sequestration values. Dislodgement data indicated biomass loss up to 129% of harvested amounts. Further research is needed to confirm these findings. This study did not account for emissions from deployment, harvesting, and air-sea exchange, and other remaining unknowns regarding carbon sequestration which are crucial for assessing net carbon balance in seaweed aquaculture.

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LIST OF ABBREVIATIONS

Abbreviation	Description
ALO	Algolesko
OF	Ocean Forest
DOC	Dissolved organic carbon
mCDR	Marine carbon dioxide removal
NPP	Net Primary Production
POC	Particulate organic carbon
P	Phosphate
SL	<i>Saccharina latissima</i>
AE	<i>Alaria esculenta</i>
ORF	Ocean Rainforest
ES	Ecosystem services
MC	Monte Carlo
N	Nitrate
P	Permanova
RDOC	Refractory Dissolved Organic Carbon
WW	Wet Weight

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INTRODUCTION

Our planet is facing major crisis with declining biodiversity, global warming due to rising atmospheric greenhouse gases (e.g. CO₂), and degrading ecosystems. Climate change can also aggravate eutrophication, where the increased temperature and altered precipitation patterns lead to higher nutrient runoff into the sea, which can further disrupt aquatic ecosystems through excessive algal blooms and oxygen depletion (Dokulil & Teubner, 2011; Pavlidou et al., 2024). The impacts of climate-change have raised concerns widely, from scientists to policymakers. The 2015 Paris Agreement has stated the urgency of keeping global warming below 2 °C, with aspirations towards 1.5 °C (United Nations, 2015). For this to be achievable, a reduction in CO₂ emissions is needed, which is a commitment many nations have pledged to follow through on. The European commission has set the target of reducing at least 55% net greenhouse gas emissions by 2030 (European Commission, 2020) and to become the first climate neutral continent by 2050 (Commission, 2019).

Seaweed has been termed a “climate champion” by the United Nations Global Compact and is being studied as a key to restoring our oceans thanks to its ecosystem services (Duarte et al., 2020). The concept of ecosystem services describes the benefits humans gain from ecosystems. There are four types of ecosystem services (IPBES, 2019) grouped by

1. Provisioning services i.e. cultivating food and feed
2. Regulatory services i.e. carbon sequestration and nutrient mitigation
3. Supportive services i.e. Habitat creation
4. Cultural services i.e. Recreational purposes such as tourism

Ecosystem services (ES) are a key component of human life and provide us with clean water, nutritious food, contribute to regulating climate and diseases, among many other elements.

The project SeaMark will demonstrate scalable and sustainable cultivation, processing and market applications based on seaweed. WP 9 within SeaMark aims to (1) assess risks and benefits to the environment by performing Life Cycle Analyses (LCA) on at least three of the seaweed products developed and (2) document and quantify the ES provided by seaweed cultivation in an ecosystem service assessment (ESA). In prior deliverables to this report, D9.1 and D9.2, the research plan and the details of the ecosystem services (ES) globally, for the marine ecosystem and for seaweed ecosystems specifically are described and compared. Then, based on collaborative choice of the SeaMark consortium, D9.2 describes it in further detail nutrient (N/P) removal, biodiversity and carbon sequestration. These three are most discussed in scientific literature and have potential to be valued and monetized. Valuation of ecosystem services is done in the follow-up deliverable (D9.4) which describes the quantification and valuation methodologies and their importance to policy makers.

The goal of this follow-up deliverable, D9.3, is to assess the three chosen ES quantitatively:

- 1) Nutrient bioextraction in chapter 1
- 2) Biodiversity in chapter 2 and
- 3) Carbon sequestration in chapter 3

Chapter 1 and 3 are assessing all three cultivation sites, one in Norway, one in France and one in the Faroe Islands, while chapter 2 focuses solely on a detailed biodiversity assessment done in the site in the Faroe Islands.

1. Context and overview of the three ecosystem services

1.1 Nutrient bioextraction

Excessive nutrients, particularly nitrogen (N) and phosphorus (P), can lead to eutrophication in aquatic ecosystems, triggering a series of environmental issues such as harmful algal blooms, reduced water quality, and the formation of “dead zones” with low oxygen levels (IUCN, 2019; Racine et al., 2021). Globally, in 2014, approximately 15-30% of fertilizers used on land ended up in the ocean (FAO, 2016; Kim et al., 2017). According to the European Environment Agency, 23% of Europe's coastal waters still face severe eutrophication (Agency European Environment, 2019). Extractive aquacultures, such as seaweed or bivalve aquaculture, could be supported in such areas as nature-based solutions. Seaweed aquaculture can have an impact on local water quality via increased oxygen (G. Gao et al., 2022; Jiang et al., 2020) and nutrient uptake (Barrett et al., 2022; Chopin & Tacon, 2020; Kim et al., 2014, 2017; Racine et al., 2021; J. Rose et al., 2015). As primary producers, seaweed absorbs N and P from the water when it grows and reproduces. This process is called *nutrient bioextraction* (J. Rose et al., 2015). While some of the nutrients are immediately used during growth of seaweed, others are stored as pigments, proteins and amino acid (Martínez et al., 2012). When the biomass is harvested, the absorbed nutrients are also removed from the environment.

1.2 Biodiversity

Ecosystems are complex networks of interacting species and the non-living environment they inhabit. One of the ES provided by an ecosystem with seaweed concerns biodiversity. It is in the category of a ‘supporting ES’, as it provides habitat and supports the biodiversity within. Biodiversity refers to the variety of species and individuals present at a particular location with dynamic and constantly changing relationships (World Resources Institute, 2003) and it is vital for the proper functioning of marine ecosystems and the provision of ecosystem services (IPBES, 2019). Seaweed aquaculture is said to support biodiversity through acting as a shelter and nursery ground for many species (Berger et al., 2023; Corrigan et al., 2024; Schutt et al., 2023; Theuerkauf et al., 2022), and is even called an “engineer” or habitat forming species (Smale et al., 2013), as seaweed beds, both natural or cultivated, can provide habitats for a range of species (Pereira et al., 2021; Radulovich et al., 2015; Teagle et al., 2017; Walls et al., 2016) and could provide a source of food (Hehre & Meeuwig, 2016; Theuerkauf et al., 2022).

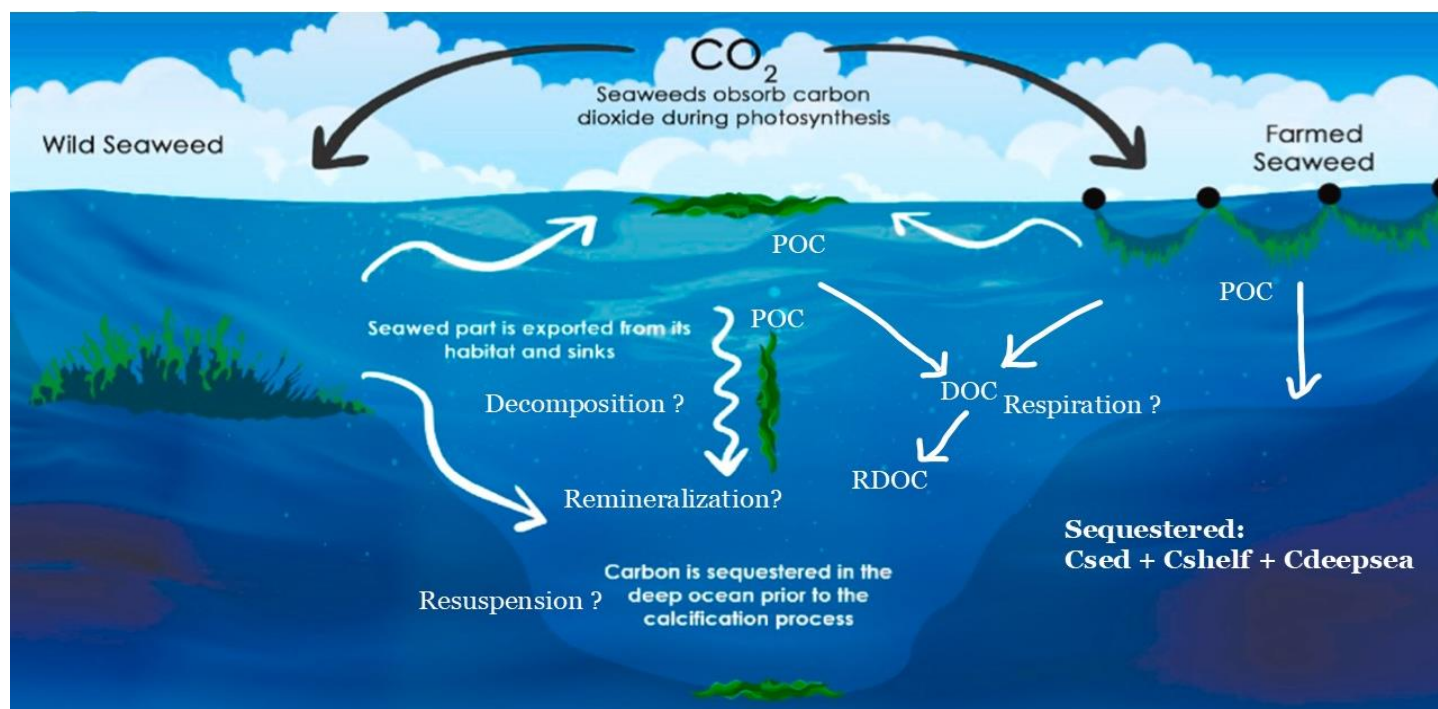


Figure 1: Carbon fluxes in the marine ecosystem. Figure adapted from Yong et al. (2022).

Quantitative studies on the contribution of seaweed farming to biodiversity are lacking (Corrigan et al., 2022, 2024; Forbes et al., 2022), and more is known about natural kelp forests. Some studies suggest that a single kelp forest can support over 7,000 individuals of different species (Bernard et al., 2019). It currently seems unlikely that seaweed cultivation can achieve the same biodiversity outcomes (Forbes et al., 2022). The approach to harvesting plays a role in this: seaweed lines could be harvested partially for several years or completely, once a year, taking away the created habitat. The design of the cultivation units can also influence the habitat provided to marine species, such as vertical line structures offering a three-dimensional habitat or horizontal longline systems creating a two-dimensional structure (Bak et al., 2020).

1.3 Carbon sequestration

Seaweed has gained increasing attention globally for its ability to capture carbon during its growth phase. Numerous studies have shown that seaweed can absorb anthropogenic CO₂ and incorporate it into organic matter (Chung et al., 2013; Duarte et al., 2017; N'Yeurt et al., 2012; D. J. Rose & Hemery, 2023; UN Global Compact, 2021). The term sequestration can only be used when the seaweed biomass is settled in the sediment and left undisturbed for over 100 years (Duarte et al., 2025; Krause-Jensen & Duarte, 2016; D. J. Rose & Hemery, 2023). Studies suggest that seaweed forest contribute significantly to carbon sequestration (Filbee-Dexter & Wernberg, 2020; Krause-Jensen & Duarte, 2016; Ortega et al., 2019; Pessarrodona et al., 2023; Yong et al., 2022). This process, however, is more complex than absorbing carbon during seaweed growth. Due to challenges and uncertainties when tracing, accounting and verifying the carbon sequestration (Pessarrodona et al., 2023), natural seaweed forests and thus also seaweed cultivation have not yet been included in the Blue Carbon ecosystems, such as

mangroves, seagrass and salt marshes (Krause-Jensen et al., 2018; Macreadie et al., 2019; Yong et al., 2022), but studies now call for its inclusion (Duarte et al., 2017; Froehlich et al., 2019; Macreadie et al., 2019; Ortega et al., 2019; Sondak et al., 2017). To quantify marine carbon sequestration from seaweed cultivation, the different fluxes and forms carbon can take, need to be considered, but are often overlooked and thus wrongfully simplified (G. Gao et al., 2022).

During the growth phase, seaweed biomass in the form of particulate organic carbon (POC) and dissolved organic carbon (DOC) becomes detached and/or erodes. Most of it will be decomposed and demineralized by microbial respiration and end up as CO₂ (M. Zhang et al., 2023). If the remaining carbon is deposited in sediments, either directly beneath the seaweed or in deeper parts of the sea where it is potentially not exposed to decomposition and remineralization (the breakdown of biomass) (Siegel et al., 2021), it can be considered sequestered (Filbee-Dexter et al., 2024). A portion of POC and DOC can be converted into stable refractory DOC (RDOC) and stay dissolved in the water (G. Gao et al., 2022; Y. Gao et al., 2021; Pessarrodona et al., 2024) (Figure 1).

Ongoing research is investigating the proportion of seaweed fragments or dissolved carbon that enters the long-term carbon cycle and becomes sequestered, versus the amount that is remineralized and remains in the short-term carbon cycle (Filbee-Dexter et al., 2022). The scientific community has yet to reach a consensus on the potential of carbon drifting into the deep sea (Dolliver & O'Connor, 2022; Hurd et al., 2022, 2023; Williamson & Gattuso, 2022), Figure 2.

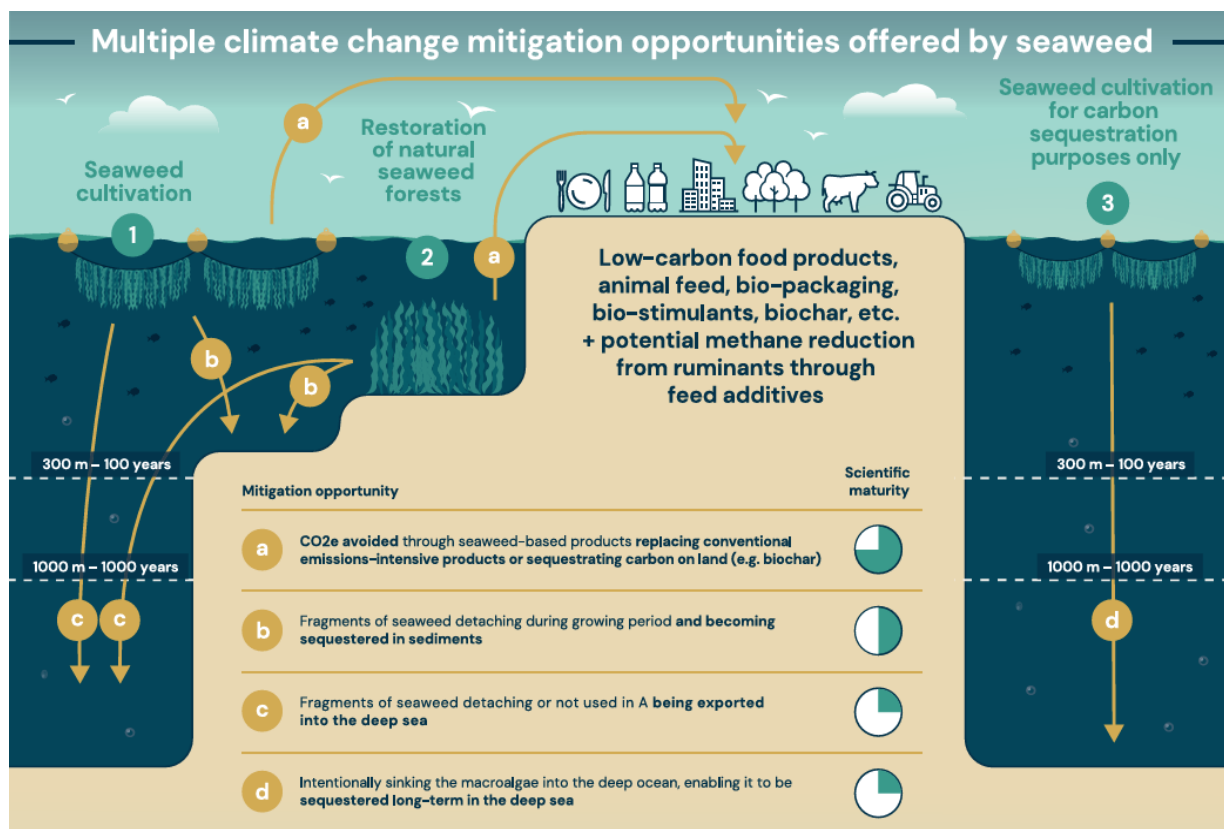


Figure 3: Climate change mitigation pathways with seaweed and their scientific maturity (UN Global Compact, 2021).

2. Site descriptions

2.1 The Faroe Islands seaweed aquaculture site of Ocean Rainforest (ORF)

The company ORF is cultivating seaweed in two fjords in the Faroe Islands: Funningsfjørður (A in Figure 3) and Gøta (B in Figure 3). In Funningsfjørður there are two sites located; a semi-exposed in the central part of the fjord, called A02 or Funning (62° 265385', -6° 959076'), and a nearshore exposed site on the outer part of the fjord (green) called AA01 or Barmur (62° 283594', -6° 928520) shown in Figure 3. In Gøta there is one exposed site called AA03 (62° 162915', -6° 657133').

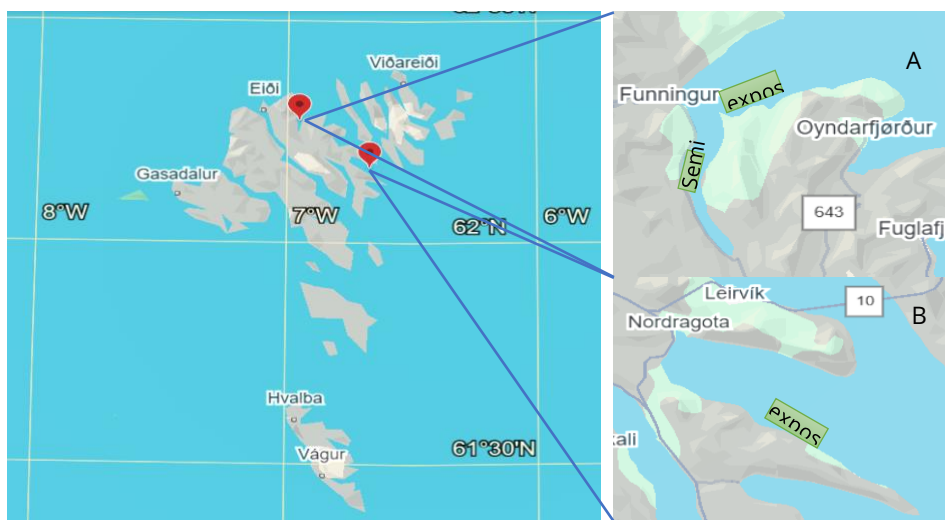


Figure 4: Geographical map of sites from Ocean Rainforest in the Faroe Islands.

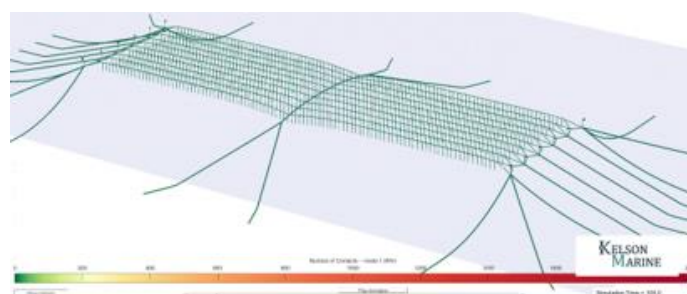


Figure 2: Cultivation structure used in the Faroe Islands.

The semi-exposed site had a water depth of 20-30 m, little to no current and occasional wave height of up to 3 m. The exposed site has a depth of 50 to 70 m, water current of 15-25 cm s⁻¹ and wave height of up to 6 m (Bruntse et al., 1999; Norði & Patursson, 2012) with a muddy substrate below the farms. The temperature of the water is stable between 7 to 11 degrees °C and salinity is stable at around 35 ppt (Bruntse et al., 1999; Larsen et al., 2008).

The seaweed species cultivated at both fjords is *S. latissima*, which is a large kelp species naturally found within the fjord. It is cultivated using a unique 3D structure with vertical dropper lines of 10 meters length shown in Figure 4. The lines are left in the



Figure 5: Geographical position of Ocean Forest.

water for 3 years with a partial harvest once or twice a year from April to October.

2.2 French seaweed aquaculture site of Algolesko (ALO)

The company ALO site is based in Lesconil, South Finistère, Brittany, France (47° 47' 00.23" N, 4° 10' 47.71" W) (Figure 5). The cultivation site is 150 hectares with a typical long line cultivation structure, in the heart of a natura 2000 classified area of the coast. *S. latissima* seaweed is seeded in September/October and harvested in March / April. All ropes are taken out of the water, only buoys and anchors remain until the next seeding period. The average depth is around 20 m, with a sandy rocky bottom. The site is exposed to the South and South-West wind, with a maximum wave height of 10 m. The

average current is around 0.26 – 0.5 m s⁻¹ and annual water temperature varying between 9 and 19°C.

2.3 Norwegian seaweed aquaculture site of Ocean Forest (OF)

The company OF site is based in Sandtorr, South-Western Norway in a sheltered bay (60° 8.902', 5° 13.635') (Figure 6).

The cultivation site is divided into 6 smaller plots. *S. latissima* and *Alaria esculenta* are cultivated in a typical longline cultivation structure. The seaweed is seeded in August on land, set out at sea in October and harvested in April or early May. All ropes are taken out of the water, only buoys and ankers remain until the next seeding period. The average depth is around 40 m, with a sandy rocky bottom, sheltered and a water temperature varying between 5-14.5 °C.

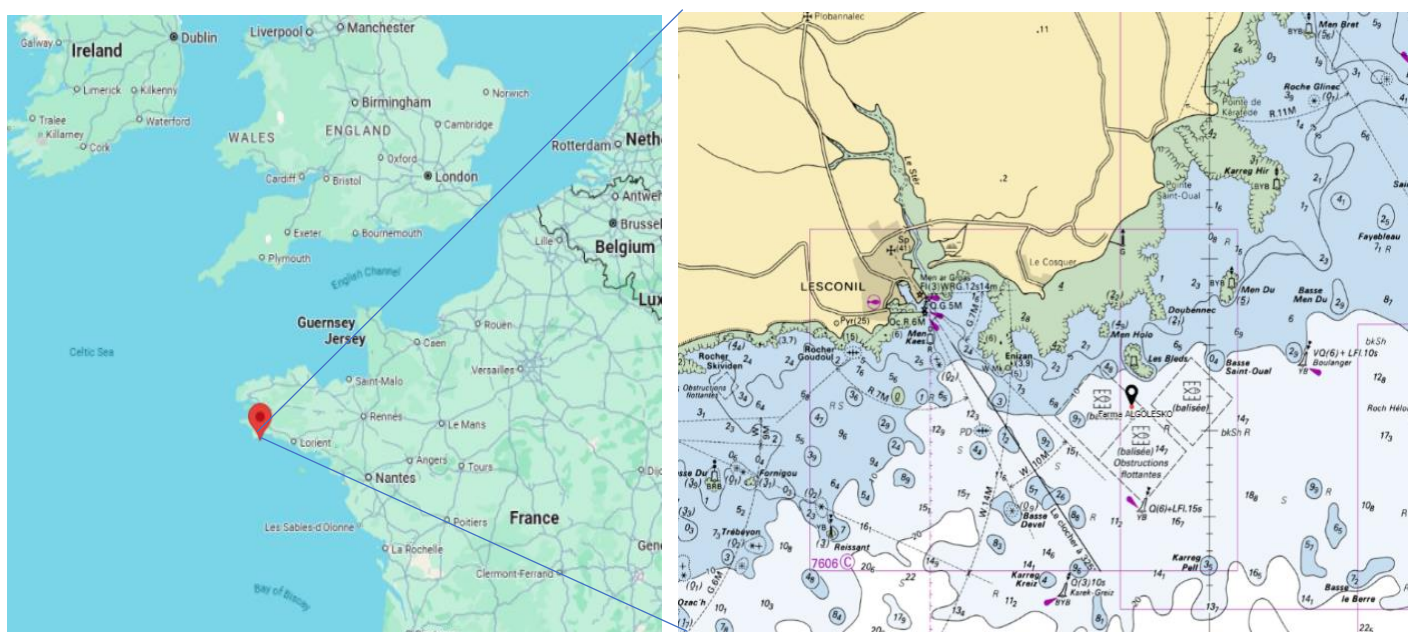


Figure 6: Geographical position of the Algolesko site.

CHAPTER I: NUTRIENT BIOEXTRACTION

In this chapter we assess the nutrient bioextraction potential for the 3 sites, including *S. latissima* and *A. esculenta* cultivation.

1. Methodology

To quantify the amount of nutrients absorbed by seaweed, for ALO three samples from 2023 harvested seaweed wet weight (WW) were taken and analyzed in the Eurofins laboratory for the content of N and P. For the two other sites, the average values from their database were taken. The amount of nutrients removed, the nutrient bioextraction potential, can be calculated via the following equation (Grebe et al., 2021; Kim et al., 2014, 2015; Neori et al., 2004):

$$\text{Bioextraction potential} = \frac{gN}{gDW} * \frac{gDW}{gWW} * \frac{gWW}{m \text{ or } ha}$$

With N being the nutrient content, DW being the dry weight and WW being the wet weight harvested. In simple words, the nutrient content is multiplied by the kg of harvest or yield. Note that the nutrient cycling, which would be the amount of nutrients taken up and subtracting the nutrients released (by degradation of the seaweed blades, erosion etc.) was not calculated. Furthermore, no additional nutrient inputs such as feed or fertilizers have to be considered in our case study, as these are not used in the three sites.

The average nutrient content and DW/WW ratios from each site are presented disregarding potential fluctuations over the seasons. These fluctuations are relevant if harvesting times would be adjusted to maximize the nutrient uptake per biomass. This is not the case with our three sites. Harvest is timed to maximize biomass and the bioextraction might not be at its highest potential.

2. Results

N and P contents of *S. latissima* and *A. esculenta* of the 3 sites are shown in Table 1.

With a harvest of 127,674 kg WW ORF removed 321.2 kg N and 62.4 kg P, which is equivalent to 2.5 kg N and 0.5 kg of P per ton of WW *S. latissima* harvested. OF removed with their harvest of 79,200 kg WW *S. latissima* 95 kg of N and 8 kg of P,

which is equivalent to 1.2 kg N and 0,1 kg P per ton of *S. latissima* harvested. With their harvest of 62,100 kg of *A. esculenta*, OF removed 106 kg N and 11 kg P, which is 1.7 kg and 0.18 kg P per ton of *A. esculenta*. ALO harvested 200 000 kg WW of *S. latissima* and thus removed 381.7 kg of N and 160.667 kg P, equivalent to 1.9 kg of N and 0.8 kg P per ton of *S. latissima* harvested.

3. Discussion

3.1 Comparison to other studies

Aquavita, a prior project to SeaMark had also assessed the nutrient bioextraction potential for ORF being 2.46 to 2.57 kg of N and 0.47 kg P t⁻¹ of harvested WW biomass, which is comparable to our results.

It has been shown that N is the limiting factor for further nutrient uptake and thus growth (Roleda & Hurd, 2019; Barrett et al., 2022 and references therein), therefore some studies priorities its assessment over P. From the literature we can see studies have large variations in their N removal results for with values ranging from 1.2 kg N ha⁻¹ for *S. latissima* (X. Zhang et al., 2022) and values going as high as 678 kg N ha⁻¹ based on values from several species (Barrett et al., 2022). Several factors can be responsible for this variation and looking at them helps us to understand differences from our results to the literature.

The nutrient content in the seaweed biomass changes over the seasons and site (Bak et al., 2019; Grebe et al., 2021; Kim et al., 2015, 2019; Racine et al., 2021). Kim et al. (2015) even conclude, that if seaweed were to be cultivated for nutrient bioextraction, it should be harvested at a different moment than harvesting for maximum biomass (Kim et al., 2015). Grebe et al. (2021) describe differences in N content for *S. latissima* between 1-4% and that the higher N content could be due to higher dissolved inorganic nitrogen (DIN) from fish- or anthropogenic waste in the water (Grebe et al., 2021; Handå et al., 2013; Kim et al., 2015). Another factor influencing the reported removal per area significantly is yield. Grebe et al. (2021) studying nutrient bioextraction from *S. latissima* in the Gulf of Maine compared their removal potential to other similar study setups and are with their 19.2–46 kg ha⁻¹ higher than other studies, but also have higher yield (10–15 kg WW m⁻¹) than their comparative studies (Table 3 in Grebe et al., (2021)). The importance of yield

Table 1: Data on nutrient contents, yield, total harvest and total nutrients removed by the three case studies for *S. latissima* (SL) and *A. esculenta* (AE).

Site	Species	Area (ha)	Harvest (t WW)	Yield kg m ⁻¹	Yield kg m ²	Content (%)		Removal					
								Kg per total WW harvest		Kg per ton WW		Kg per ha	
						N	P	N	P	N	P	N	P
ORF	SL	17.85	127.7	1.51	0.72	2.15	0.42	321.2	62.74	2.52	0.49	18.0	3.51
OF	SL	N.A.	79.2	4.02	N.A.	1.41	0.119	95.04	8.02	1.2	0.101	N.A.	N.A.
	AE	N.A.	62.1	2.38	N.A.	1.3	0.139	105.6	11.27	1.7	0.18	N.A.	N.A.
ALO	SL	150	200	N.A.	0.13	1.91	0.801	381.7	160.667	1.91	0.801	2.5	1.1

becomes particularly obvious when looking at Barrett et al. (2022) having a very high nutrient bioextraction rate per area with on average 275 kg N ha⁻¹ while ALO has 2.5 kg N ha⁻¹ and ORF has 18 kg N ha⁻¹, but the study's N removal per harvested kg WW is 3.8 kg N t⁻¹ WW harvested, similar to our values (Barrett et al., 2022). A recent study from Denmark tested different cultivation techniques to achieve higher yields and also higher bioextraction potential. Their optimized yield for *S. latissima* results are 5.0 - 6.8 kg WW m⁻² yr⁻¹, and 10.9 - 30.4 kg WW m⁻² for 1.5 years of growth (Boderskov et al., 2023), which is still lower than the study from the Gulf of Maine showing how even optimized yield can be lower than other regions. Other examples of high N removal due to high yield are reaching a removal up to 82 kg N ha⁻¹ calculated with a yield of 1.8 kg WW m⁻² (18,350 kg WW ha⁻¹) and an N content of 2.3% (X. Zhang et al., 2022) or modeling even 576 kg N ha⁻¹ removal calculated with 12 kg WW m⁻² (18 t DW ha⁻¹) and with a high N content of 3.2 % (Holdt & Edwards, 2014). Further, these studies calculate a removal potential, therefore choosing the highest N content (Holdt & Edwards, 2014) and using yields from test cultivation sites (Zhang et al., 2022). This contributes to the incomparability to operational seaweed cultivation companies which do not plan the harvest times for the highest nutrient bioextraction potential.

3.2 Seaweed's nutrient bioextraction potential as an aspect of regenerative aquaculture

Considering that 60% of our coast lines are estimated to suffer from eutrophication (Theuerkauf et al., 2019), its mitigation is of great importance and seaweed aquaculture can play its part in this process. A study projected that if seaweed were farmed in just 0.03% of the ocean, it could remove around 30% of the nitrogen entering the ocean (Kim et al., 2017; Yarish et al., 2016). Moreover, seaweed biomass could be used to create agricultural biostimulants (Arias et al., 2024; Garcia-Caparrós et al., 2025; Makonya et al., 2025; Tariq et al., 2025), reducing the need for fertilizers. This could help decrease the excess nutrients from agriculture that make their way into the water. Seaweed aquaculture, with its extractive quality is regenerative to an ecosystem (Alleway et al., 2023). Seaweed aquaculture or other extractive aquaculture could thus be implemented in targeted eutrophic areas (Barrett et al., 2022; Grebe et al., 2021; Hughes et al., 2022). It can be used in specific nutrient rich locations, or in combination with other fed aquacultures (e.g., fish or shrimp) and mitigating their excess nutrient discharge. This system is called Integrated MultiTrophic Aquaculture (IMTA) (Carras et al., 2020; Chopin & Tacon, 2020; Kim et al., 2017).

3.3 Consequences and limitations to nutrient bioextraction potential

In other context and locations, and at another scale of a cultivation, this nutrient bioextraction potential may not always have positive effects; in fact, it could have negative consequences for surrounding individuals, populations, or ecosystems. For example, cultivating seaweed in a nutrient poor environment may lead to competition of the nutrients or even depletion of nutrients (Jiang et al., 2020; Kotta et al., 2022;

van der Meer, 2020). Understanding the extractive potential and the condition of the environment is essential for evaluating the acceptable impacts of activities like aquaculture. The concept of carrying capacity, an interdisciplinary approach, defines the threshold beyond which changes are considered unacceptable (Kluger & Filgueira, 2020; McKindsey et al., 2006; Weitzman et al., 2021; Weitzman & Filgueira, 2020), and it is adaptable to assess the capacity of different ecosystems and contexts (preprint Koch et al., 2024; Weitzman et al., 2021).

CHAPTER II: BIODIVERSITY

The objective of this chapter is to understand firstly in general what species and how many find shelter in the Faroese cultivation site. The goal is to clarify if the created habitat is having any negative or unwanted impacts, such as uncommon or invasive species, and/or to confirm that the created habitat supports the local species populations, such as commercially interesting fish. Secondly, we aim to understand if the three-year duration of leaving the cultivation structures in the water has an added value for biodiversity or might be more beneficial than common one-seasonal cultivation. For this we assess if and how the species abundance and richness differ within the seasons and over the years. A third objective is to assess how biodiversity varies with different exposures. This might play a role when expanding and looking for new sites.

To answer these questions, quantitative data was collected from different seasons, sites with different exposures and from lines being in the water between 1 and 3 years.

1. Methodology

1.1 Sampling

Sampling was done in Funningsfjørður, which is a fjord located in the Faroe Islands (view figure 5 in site description). The sampling was done at two sites (see Figure 7), one semi-exposed site (Funning or A02) and one exposed site (Barmur or AA01), allowing to compare the effect of exposure on biodiversity.

Sampling was done twice, once in early spring (April), the beginning of the seaweed growth, and once in the summer (July), where seaweed is fully grown, to capture the full life cycle of the different species that can vary throughout the year.



Figure 7: Placement of the two sites with their exposures.

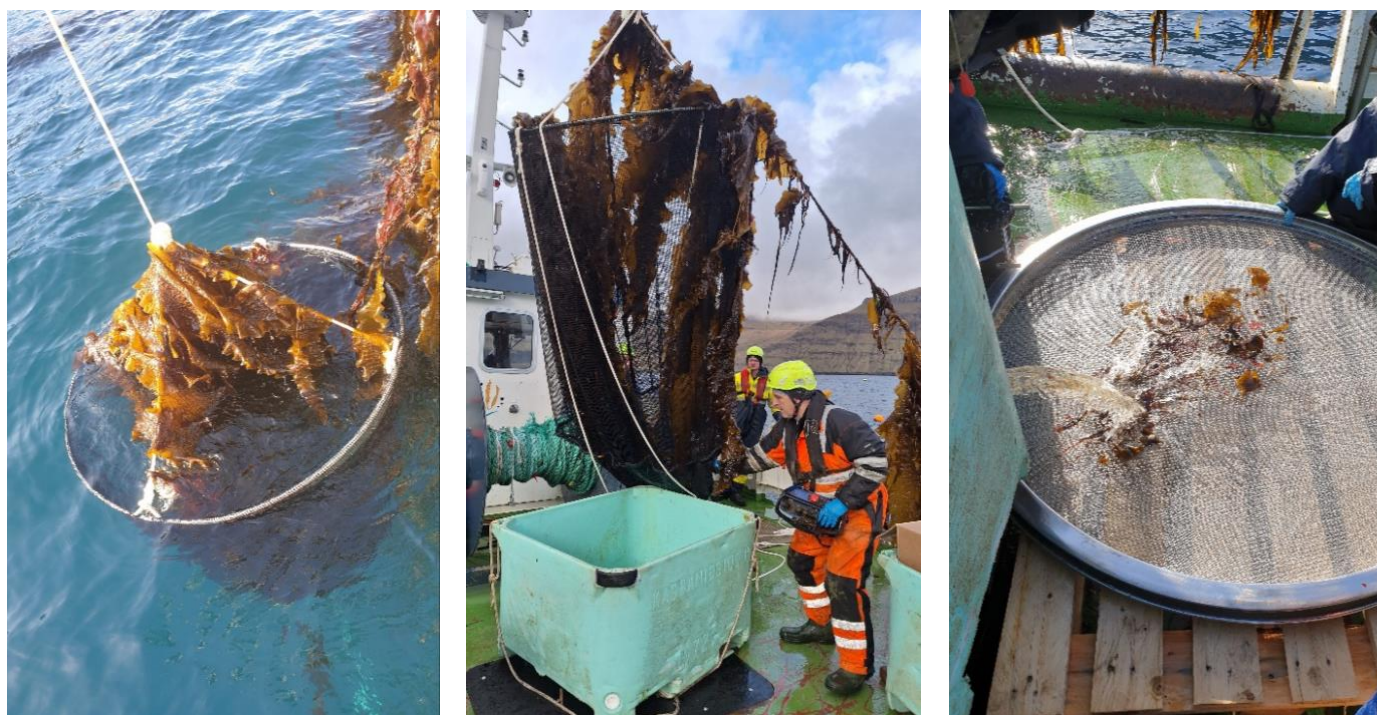


Figure 8: Sampling process. Left: Vertical line being lifted out of the water. Middle: vertical line being placed in the fish tub. Right mobile species being filtered out.

Periods of expected low and high biodiversity could thus be captured.

ORF cultivates seaweed using vertical dropper lines, and the cultivation system allows for the seaweed to be left out for 3 to 4 years with partial harvest once or twice a year. 3 replicate lines were taken from first-, second- and third-year lines, providing samples of biodiversity developing over the years. This correlates to the biodiversity measured in three continuous years, but more effectively and in the same water conditions. The catamaran workboat Breiðblöðka was used for the sampling. On the seaweed cultivation site, a net (diameter 1.5m, length 2m and mesh size 1 cm) was lowered below the 10 m vertical line and positioned below, with help of an anker, to pull up the entire single vertical line and the organisms living in and on it (Figure 8, left). The line and net were brought onboard the boat and weighed, before the line was placed into a fish box full of seawater (Figure 8, middle). The net was rinsed thoroughly above the fish box so as to not lose any organisms bigger than 1 cm (Figure 8, right).

The line was carefully washed in the fish box and removed in segments to release as many attached organisms as possible, and segments of 30 cm at the 4th and 7th meter of line were taken and placed in a bag for further analysis (line samples). The water and organisms in the fish box were rinsed through sieves (1 mm mesh size) to catch all the mobile organisms (Figure 8, right), which were placed in bags for further analysis (mobile samples). This was done for all 12 lines, in both sampling events.

These replicates of lines for each season (12 per season, 24 replicates in total) were taken:

- 3 x 3-year-old lines at semi-exposed site

- 3 x 1 year old lines at exposed site
- 3 x 2 years old lines at exposed site
- 3 x 3 years old lines at exposed site

The bags with the sample were brought back to shore and sorted the day after sampling. For the line samples, all organisms were picked off the blades, holdfast and line, and placed in a jar with 70% ethanol for further analysis. For the mobile samples, all individuals above 1cm were placed in a jar with 70% ethanol and placed in a cool room (10 °C). Since the smaller individuals could have theoretically been washed out of the net, we did not quantify them but took a small representative sample of each organism for identification. The organisms were identified to the lowest taxonomic level possible and counted (July and September 2024).

1.2 Data analysis

1.2.1 Quantifying biomass samples

To combine the two data sets from the two different sampling methods, line and mobile samples, 4 steps were conducted:

1. In case sessile species found and counted within the mobile samples, for example fallen off shells and clams (in Table 1: not considered), were deleted. The ones that were too small to be quantified (smaller than the mesh size of the net) were marked as 'present but too small' in Table 1.
2. Two columns for species found in both samples were made: one column for the individuals found in the line samples (all sizes were counted) and another column for the species found in the mobile sample (only individuals above 1 cm were counted).

3. The rope samples were extrapolated from 30 cm to 5 meters. The total length of vertical growth line is 10 m, the upper and lower samples of each line representing 5 m.
4. The line samples were added together.

Table 2 shows which species are from which sampling method and which species have 2 columns for the different sizes.

Table 2: Overview of species present in the sample methods and which species were divided into two columns.

Species name	Common name	Line samples (mostly sessile)	Mobile samples (mostly mobile)	Two columns
<i>Mytilus edulis</i>	Blue mussels	yes	Not considered	No
<i>Hiatella arctica</i>	White square-ish bivalve	yes	Not considered	No
<i>Aequipecten opercularis</i>	Fan-like bivalve	yes	Not considered	No
<i>Hyas araneus</i>	Crab with pointy head	yes	yes	yes
<i>Cyclopterus lumpus</i>	Lumpfish	No, doesn't appear	yes	No
<i>Caprella mutica</i>	"skeleton shrimp"	yes	yes	yes
<i>Tectura virginea</i>	Small limpet-like, top not in center	yes	Not considered	No
<i>Idotea balthica</i>	Large isopod	No, doesn't appear	yes	No
<i>Styela rustica</i>	Sea squirt	yes	Not considered	No
<i>Galathea</i>	Lobster	yes	yes	yes
<i>Anomiidae</i>	Saltwater clams	yes	Not considered	No
<i>Gammaridae</i>	Small amphipods	yes	Present, but too small	No
<i>Caridea</i>	Shrimp	No, doesn't appear	yes	No
<i>Isopoda</i>	Unknown isopods	No, doesn't appear	yes	No
<i>Nudibranchia</i>	"Naked" snails	yes	yes	yes
<i>Cirripedia</i>	Barnacles	yes	Not considered	No
<i>Ascidacea</i>	Sea squirt	yes	Not considered	No
<i>Polychaeta</i>	Brittle worms	yes	yes	yes
<i>Gastropoda</i>	Sea snails with shell	yes	Present, but too small	No
<i>Echinoidea</i>	Sea urchin	No, doesn't appear	yes	No
<i>Ichthyoplankton</i>	Fish larvae	No, doesn't appear	yes	No

1.2.2 Statistical testing

We formally examined statistical differences between both seasons, within the three years and between the two sites using univariate permutational analysis of variance (PERMANOVA), in PRIMER v7 (Version 7.0.13) using the PERMANOVA add-on. We used Bray-Curtis similarities with 99 999 permutations under an unrestricted model. When a factor showed a significant effect ($p < 0.05$), we conducted pairwise comparisons to identify where differences occurred. For Shannon indexes, shading plots and cluster graphs PRIMER v7 was used, while the boxplots were made using R software (version 2024.09.1 +394) was used. The data for the shading plots was log-transformed to optimize the visualization of the large differences in abundance of species. In non-metric Multidimensional Scaling (MDS) plots and Permanova analysis, dummy species were added to better differentiate the sample with zero or just one species. Additionally, we excluded the 1st year line April samples (3 replicates) for the non-metric MDS graphs, because only one sample contained one species, the others none. These samples were statistically too distant from the others, making visualization difficult. To identify which species contribute most to differences between samples, we performed Canonical Analysis of Principal Coordinates (CAPs) based on Bray-Curtis similarities. This was complemented with Permanova tests with Monte Carlo simulations based on D2 average distances for each species until reaching a statistical significance threshold of $p = 0.05$.

1.3 Literature

The results of this study were compared to current literature. We did not start a review, rather based the reading of publications on recent very comprehensive literature reviews (Corrigan et al., 2022; Forbes et al., 2022; Theuerkauf et al., 2022), read further studies using the snowballing method, as well as more recent studies up until February 2025. Due to a limited number of studies available on quantifying biodiversity in and around seaweed farms, we discussed results from seaweed farms in general, always stating the region and species, and whenever possible compare our results to the regional ecosystem of the Faroe Islands or the cultivated species *S. latissima*.

2. Results

2.1 Abundance of species

2.1.1 Total number of species and individuals in Funningsfjørður

Our results differentiate between 21 faunae, not all identified to the species level. Our results represent at least 10 classes (95% identified), at least 17 orders (76% identified), at least 21 families, genus and species with respectively 67%, 48% and 43% identified to this level. A total list of all species and their abundance is found in the supplementary material. Figure 9 shows the most dominant ones.

Saltwater clams (*Anomiidae*) were the most dominant species found overall, followed by scuds (*Gammaridae*) and sea snails (*Gastropoda*). However, any bivalves only start appearing from the 2nd year samples onwards, thus the most dominant species in the first year are scuds (*Gammaridae*), followed by sea snails

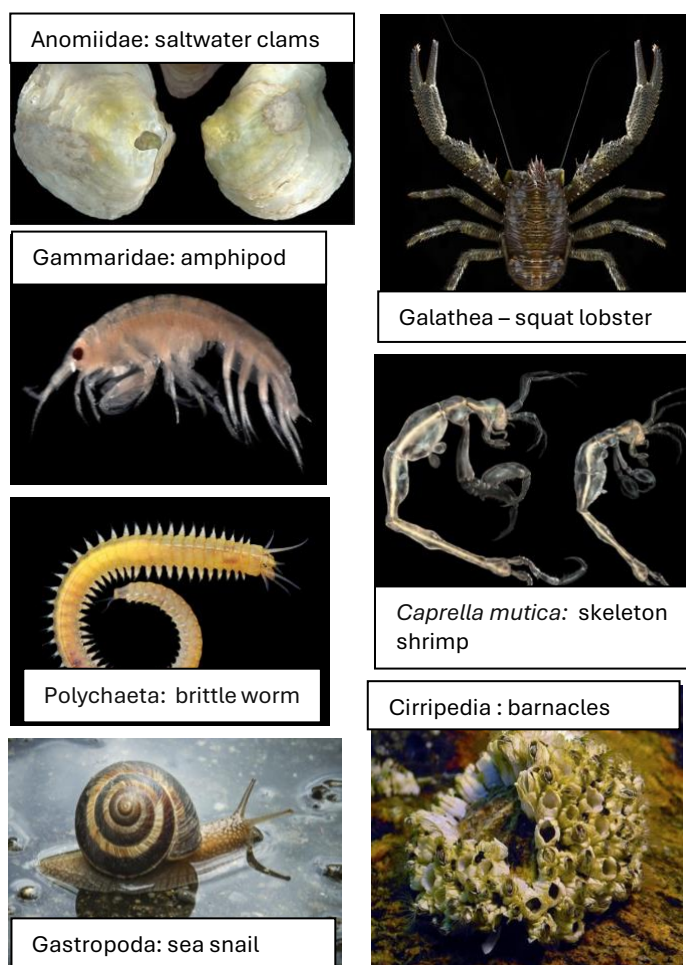


Figure 9: Most dominant organisms found in Funningsfjørður.

(*Gastropoda*) and skeleton shrimps (*Caprella mutica*). When differentiating between the seasons, in April saltwater clams (*Anomiidae*) are by far the most dominant species, followed by scuds (*Gammaridae*) and brittle worms (*Polychaeta*). In July the abundance of species is greater and even though saltwater clams (*Anomiidae*) are still the most dominant, scuds (*Gammaridae*) and sea snails (*Gastropoda*) are in the same range of magnitude. While most species increase in abundance in July, for example, sea snails (*Gastropoda*) multiplied by factor 650, barnacles (*Cirripedia*) multiplied by factor 27 and scuds (*Gammaridae*) multiplied by factor 15, some species also decrease from April to July, such as saltwater clams (*Anomiidae*) and squat lobster (*Galathea*), Table 3. Table 3 present the overall number of organisms found in Funningsfjørður.

3.1.2 Abundance of these species visualized

The abundance of the species is visualized in Figure 10 with a Log transformation of the data to normalize the large differences in abundance.

Figure 10 shows that very few species are present in April in the first-year lines (*Isopoda*), and even though there are more in July, there's still a difference visible to the abundance and evenness of species to the other years. Most species are not very abundant in the first year, only appear in the second-year lines, other species like *Styela rustica* and *Idotea balthica* only appear in the 3rd year. There's a bigger difference from the first

year to the second year than from the second to the third year, which is later confirmed by Permanovas of the similarities between the years (section 3.2.2).

Figure 10 furthermore shows that some species are more present in the July samples (right half of the plot) than in the April samples (left half of the pot), like *Mytilus edulis*, *Hyas araneus*, *Cyclopterus lumpus*, *Gastropoda*, *Nudibranchia*,

Caprella mutica. Some only arrive in July, for example, *Caprella Mutica*, *Stella rustica*, *Idotea balthica*, *Echinoidea*, and *Ichthyoplankton* and some are more present in the springtime, like *Galathea* and *Isopoda*. Some species, like *Anomiidae* and *Gammaridae* have a very high number of individuals per sample (some samples >1000), while for other species only single individuals were found, like *Galathea*, *Cyclopterus lumpus*, *Isopoda*, *Nudibranchia*, *Caridea*.

Table 3: Overall number of organisms found within the whole cultivation unit of Funningsfjørður, based on extrapolations of our individual samples (view section 2.2.1).

Species	Exposed site						Semi-exposed site	
	April			July			April	July
	1st year	2nd year	3rd year	1st year	2nd year	3rd year	2nd and 3rd year	2nd and 3rd year
<i>Mytilus edulis</i>	-	44.653	9.424	-	234.430	122.513	297.259	363.317
<i>Hiatella arctica</i>	-	1.250.294	508.902	-	1.641.011	848.170	677.091	3.352.426
<i>Aequipecten opercularis</i>	-	11.163	-	-	-	-	16.514	-
<i>Hyas araneus</i>	-	82.831	33.926	38.763	1.253.643	175.853	1.981	228.557
<i>Cyclopterus lumpus</i>	-	670	-	581	670	2.261	3.963	1.981
<i>Caprella mutica</i>	-	11.163	-	165.517	245.594	750.158	-	909.283
<i>Tectura virginea</i>	-	-	-	-	-	-	16.514	-
<i>Idotea balthica</i>	-	-	-	-	-	-	-	1.981
<i>Styela rustica</i>	-	-	-	-	-	18.848	-	544.976
<i>Galathea</i>	-	61.844	12.816	-	34.160	565	991	1.981
<i>Anomiidae</i>	-	23.643.961	19.762.352	-	11.040.547	4.994.777	6.093.819	6.457.136
<i>Gammaridae</i>	-	312.574	320.420	1.880.020	3.549.943	8.896.357	660.577	313.774
<i>Caridea</i>	-	670	1.131	-	670	-	7.925	2.972
<i>Isopoda</i>	523	1.339	1.696	581	-	-	1.981	1.981
<i>Nudibranchia</i>	-	-	-	1.744	1.339	9.989	991	49.543
<i>Cirripedia</i>	-	111.633	122.513	38.763	960.048	556.022	148.630	11.659.176
<i>Ascidacea</i>	-	44.653	18.848	-	33.490	-	49.543	330.288
<i>Polychaeta</i>	-	1.691.009	346.238	12.597	389.599	678.157	1.314.862	1.989.972
<i>Gastropoda</i>	-	11.163	-	1.327.643	4.041.130	3.147.652	16.514	7.844.346
<i>Echinoidea</i>	-	-	-	-	670	-	-	-
<i>Ichthyoplankton</i>	-	-	-	581	4.018	-	-	991
Total individuals	523	27 279 622	21 138 266	3 466 793	23 430 960	20 201 322	9 309 156	34 054 682

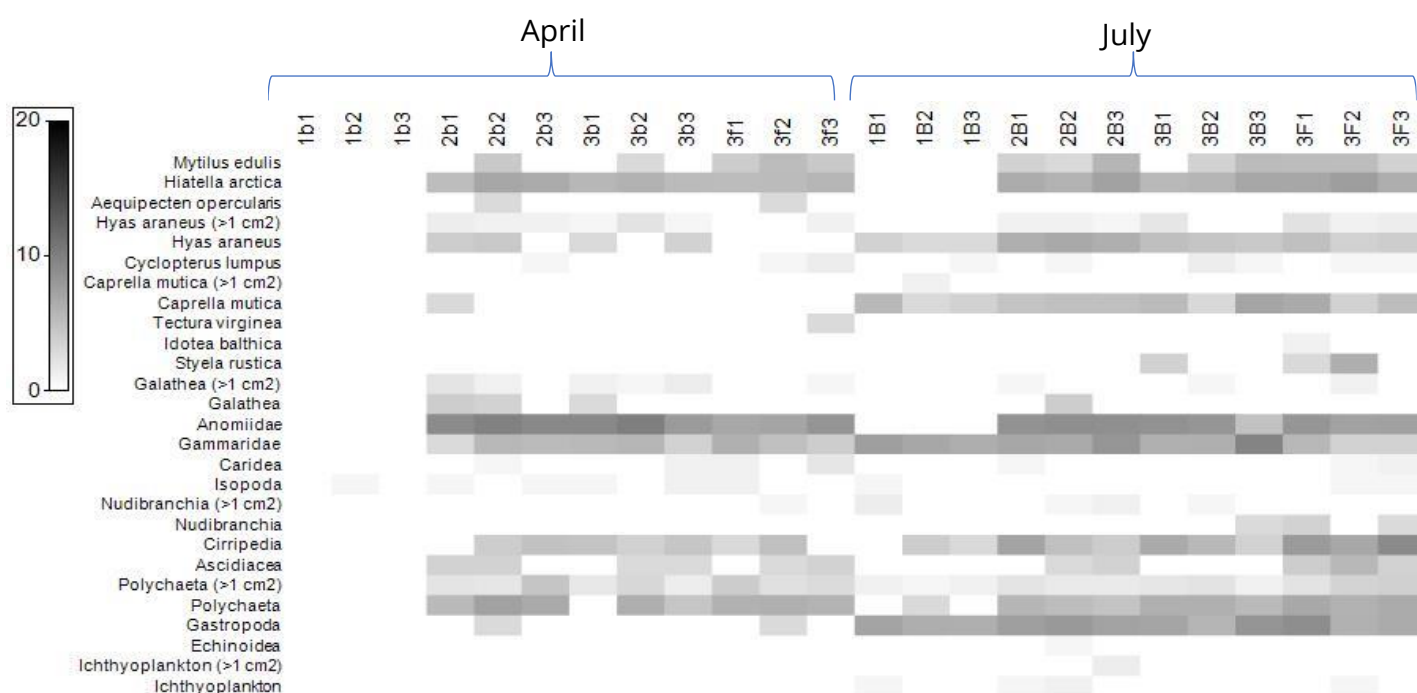


Figure 10: Shade plot with log data transformation showing the relative abundance of each species. The sample labels in the x axis signify which growth year (1, 2 or 3), which site (b, B for Barmur, the exposed site, and f, F for Funning, the sei-exposed site, and f, F for Funning, the sei-exposed site, and f F for Funning, the semi-exposed site, and which season (b, f for April and B, F for July).

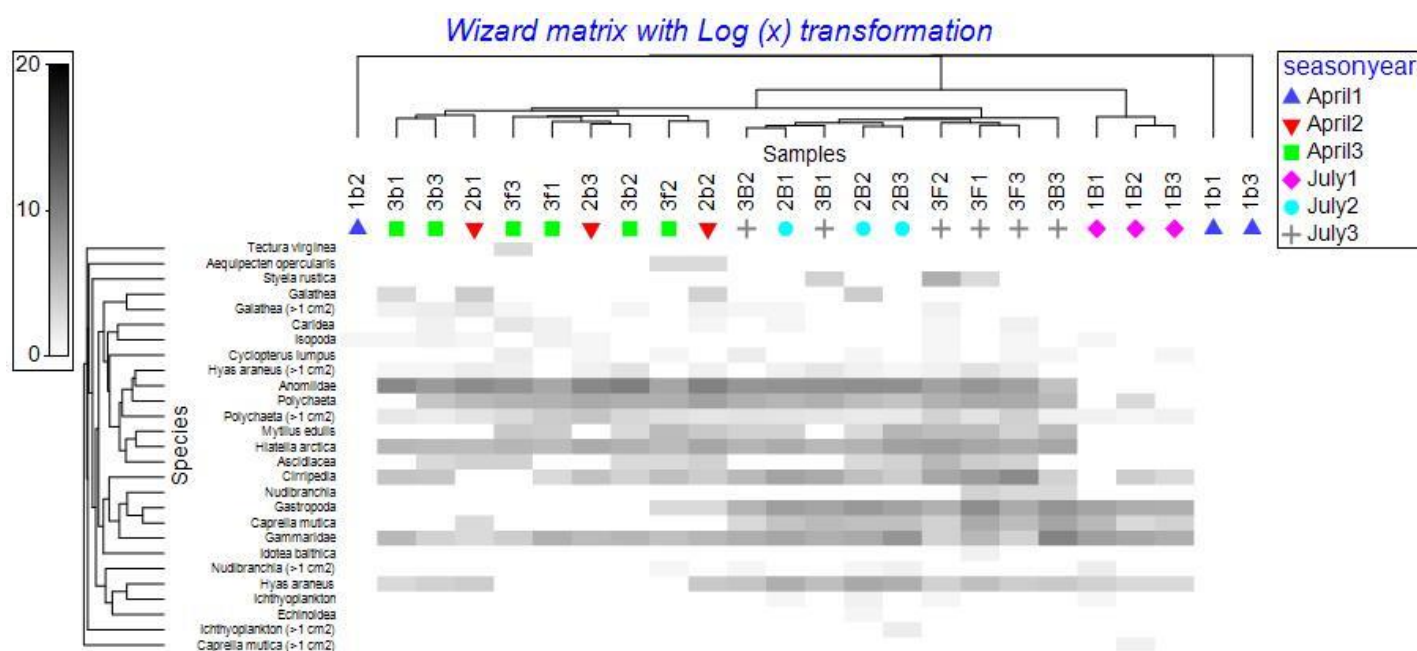


Figure 11: Wizard matrix based on Bray-Curtis similarities showing a grouping of the samples.

A wizard matrix, based on a Bray-Curtis Similarity, grouped similar samples together, helping to identify where the biggest differences lie within the samples, Figure 11.

Figure 11 shows that April samples are in separate groups from the July samples, reiterating that there's a difference in these groups of samples. The first-year samples are also not mixed within the 2nd and 3rd year samples. The biggest group of samples is a group made from only April samples, 2nd and 3rd year ones, including both sites (exposed and semi-exposed). The second biggest group consists of July samples, also both sites mixed and 2nd and 3rd year lines. There are two 3rd largest groups consist of only 3rd year semi-exposed July samples and only 1st year samples from the exposed site in July.

2.2 Effect of season, year and site on biodiversity

The first indices on the importance of the factors season, site or year play for biodiversity are statistically tested. Permanova was used to test the resemblance in the samples for three

factors: season, year, site and timeline and is respectively 5e-5, 1e-5, 0.0143 and 1e-5, all statistically different. When testing the interactions of the factors, 'season and site' and 'timeline and site' are not statistically different, and no test was possible for 'year and site', since only one site covers all three years, but 'season and year' was statistically different ($P=0.0006$) and thus pairwise tests were conducted.

2.2.1 Factor 'season'

2.2.1.1 Only 'season'

The non-metric MDS graph with factor season (Figure 12) visually supports the statistical difference in season detected by the Permanova test. A boxplot graph visually compares the species abundance within the seasons, Figure 13. We statistically tested which species are relevant for the separation of the clusters, in other words, which species create the biggest difference in the samples. Permanovas, including a The Monte Carlo test showed that *Gastropoda* (MC=0.0035), *Caprella mutica* (MC=0.0205), *Hyas araneus* (MC=0.0212) and *Ichthyoplankton* (MC=0.0422) are key responsible for the distinction between the samples in season, which statistically confirms what is seen in Figure 13.

2.2.1.2 Interaction of 'season and year'

When looking at the interaction of the factors 'season and year', the seasonal distinction and the distinction of the 1st year samples is visible in Figure 14. Statistically there's only a difference in seasons within the 3rd year samples ($P=0.0117$), in the 1st and 2nd year line samples there's no statistical difference (respectively $P=0.0998$ and $P=0.1$; note that the boxplot doesn't have the 1st year samples from April). The statistical difference between the seasons grows with the years. When looking at

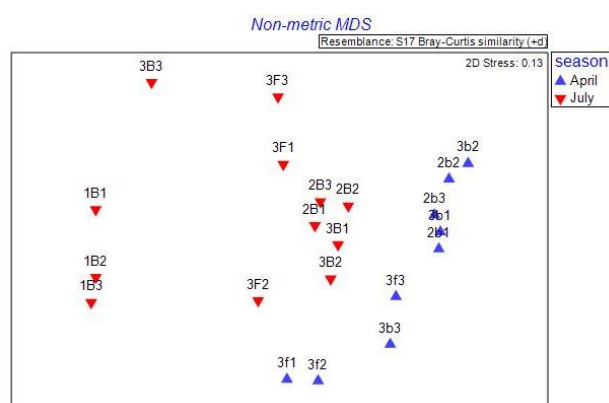


Figure 12: Non-metric MDS plot of the factor 'season'. Data does not include 1st year line April samples.

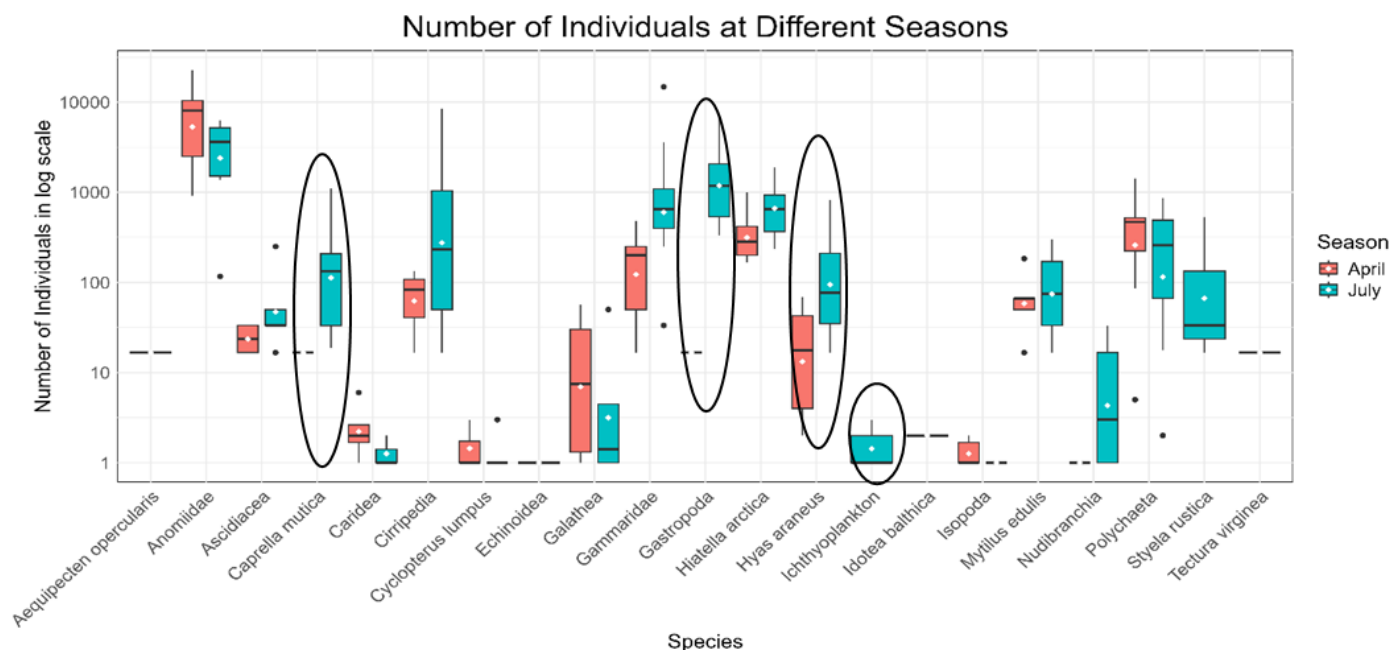


Figure 14: Boxplot of data from both sites and all years comparing the seasons (April and July samples), with the median, mean (white dots), upper and lower quartile and black dots for minimum and maximum values. Circles show the species responsible for creating the biggest difference in the samples.

how the years change within the season April, the biggest and statistical difference is between the 1st and 3rd year line samples ($P=0.0048$), 1st and 2nd year line samples are not statistically significantly enough ($P=0.0997$), as aren't 2nd and 3rd year sample ($P=0.5698$). In July, the similarity ranking is the same with samples from the 1st and 3rd year lines are significantly different ($P=0.0226$), but samples from the 1st and

2.2.2 Factor 'year'

2.2.2.1 Year including both sites

he samples. The non-metric MDS graph with factor year (Figure 15) visually supports the results from Permanova stating that the years are statistically different. It is visible that 1st year samples are distinct from the other samples, and that 2nd and 3rd year samples are mixed up, yet 2nd year line samples are grouped closer together. The Permanova pairwise test confirms that 1st and 3rd year line samples are most different ($P=1e-6$), 1st and 2nd year line samples are also very different ($P=1e-5$), while 2nd and 3rd year line samples are quite similar

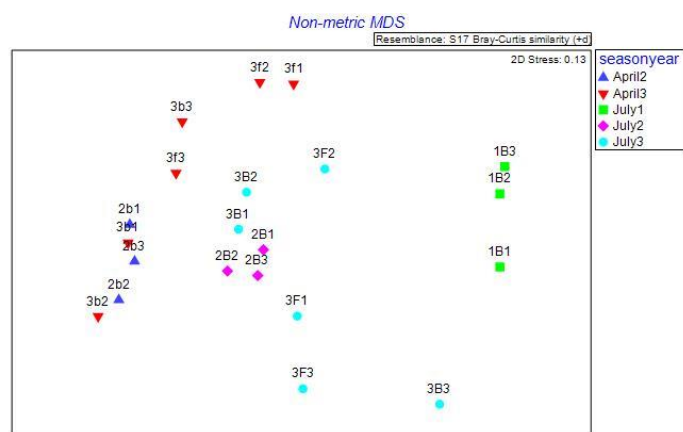


Figure 13: Non-metric MDS plot with combined factor 'season and year'. Data does not include 1st year line April samples.

2nd year lines are also not statistically different enough ($P=0.1005$), as well as samples from the 2nd and 3rd year lines ($P= 0.8788$). In general years are more different in April than they are in July; the similarity within the samples grows from April to July.

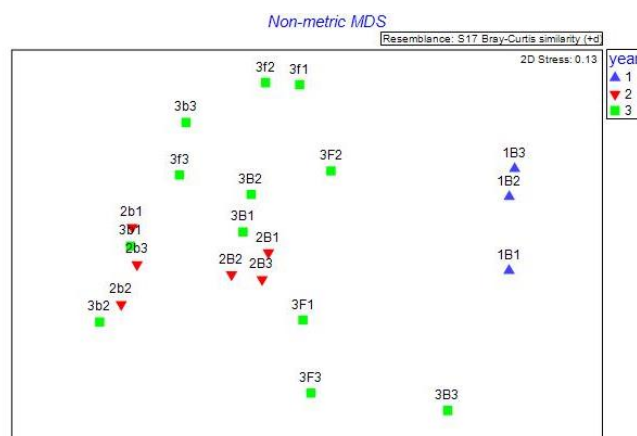


Figure 15: Non-metric MDS plot of the factor 'year'. Data does not include 1st year line April samples.

and not statistically different ($P=0.5692$); a similar pattern to the difference in years within the seasons, see section 2.2.1.2.

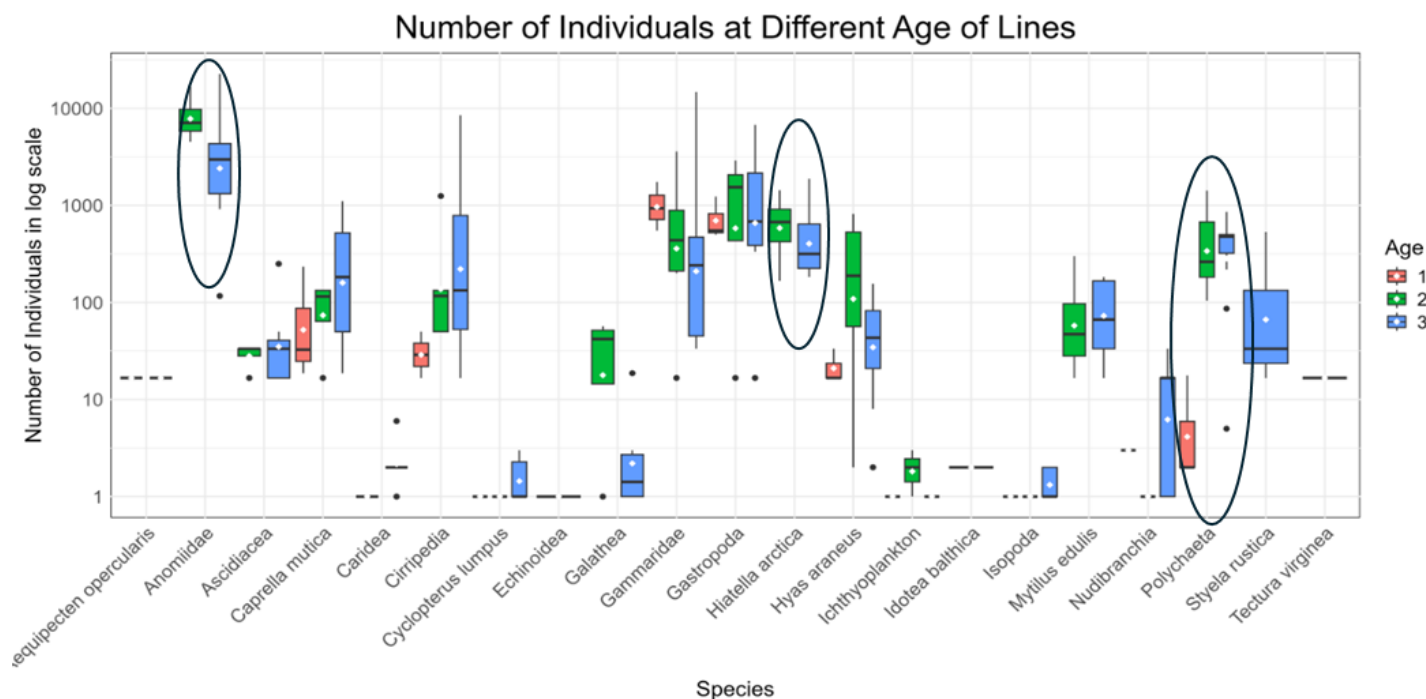


Figure 17: Box plot of data from both seasons and both sites compared by year, with the median, mean (white dot), upper and lower quartile and black dots for minimum and maximum values. Circles show the species responsible for creating the biggest difference in the samples.

The boxplot graph in Figure 16 offers a visual comparison of the difference of species abundance over the three years. Using CAP, we statistically tested which species play a main role in the difference in samples (Figure 17).

In Figure 17, CAP separated the 1st year line samples from those of the following years. A Permanova, including a Monte Carlo test, statistically confirmed that *Polychaeta* ($P=0.017$) contributed most to this separation of groups, with a very low number of individuals in the 1st year compared to the 2 other years (Figure 16). A pairwise test, looking into the year, confirms that *Polychaeta* are most different between 1st and 3rd year line samples ($P=0.0008$), and 1st and 2nd year line samples are more similar, but still statistically different ($P=0.0395$) while 2nd and 3rd are not statistically different ($P=0.6$).

Another species creating a difference in the samples is *Anomiidae* ($P=0.0242$), also visible in Figure 14. The pairwise tests show that for *Anomiidae*, 1st and 2nd ($P=0.0007$) are different, while 1st and 3rd ($P=0.086$) and 2nd and 3rd year are not statistically different anymore ($P=0.19$).

Another statistically relevant species is *Hiatella arctica* ($P=0.0175$). The pairwise test shows that the biggest difference is between 1st and 2nd year line samples ($P=0.0031$), then 1st and 3rd year line samples ($P=0.0194$) but 2nd and 3rd year line samples are statistically not different ($P=0.46$).

As can be seen in Figure 16, these bivalves are not present in the 1st year line samples, thus creating this difference.

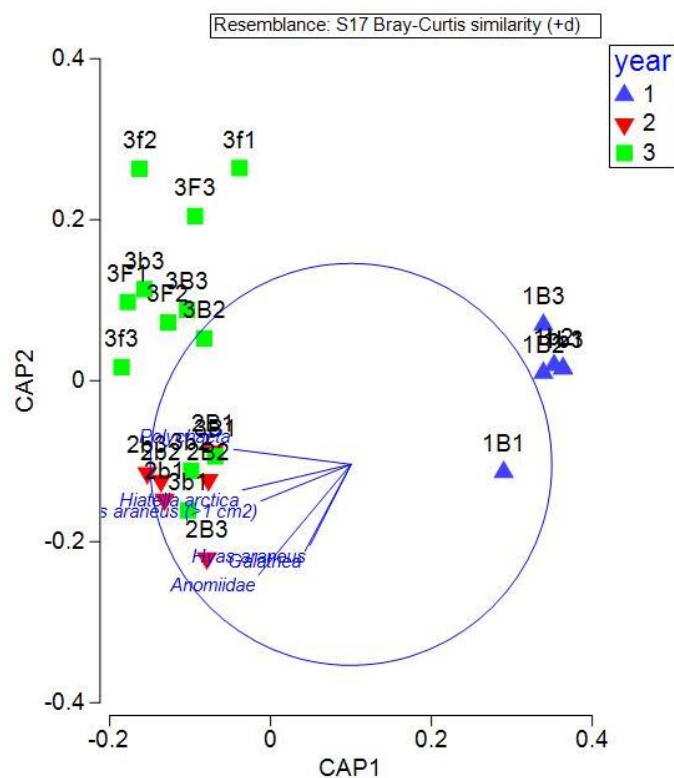


Figure 16: Cap graph of factor year displaying species with $R \geq 0.45$. Data does not include 1st year line April samples.

2.2.2.2 Factor 'year' only for the exposed site

The non-metric MDS graph with factor 'year' only for the exposed site (Figure 18) shows similar clusters of years like the data from both sites (Figure 15). A Permanova main test of only the exposed site data, states that the seasons and years are statistically different ($P=4e-5$ and $P=1e-5$ respectively), as is the interactions of both factors ($P=0.0023$).

In the pairwise test the similarity ranking between the different years, is again similar to the tests from both sites, section 2.2.2.1, and to the tests for year with in the seasons, section 2.2.1.2: 1st and 3rd year line samples are most different ($P=9e-5$), then 1st and 2nd year line samples ($P=2e-5$) and 2nd and 3rd year line samples are not statistically different anymore ($P=0.5348$). When it comes to comparing years, the samples (only 3rd year) from the semi-exposed site, did not change the outcome significantly.

A box plot Graph (Figure 19) visually compares the species' abundance within the years for only the exposed site. We visualized using CAP which species play a main role in the difference in samples (Figure 20).

Figure 20 shows that with the factor 'year', the 1st year line samples are again clearly distinct from the samples of the other two-year line samples. A Permanova with Monte Carlo test confirmed *Hiatella arctica* to play the most relevant role in sample differentiation ($P=0.004$), with pairwise tests detailing that the biggest difference is between 1st and 2nd year line samples ($P=0.0029$), and 1st and 3rd ($P=0.0047$), while 2nd and 3rd don't have a statistical difference ($P=0.1701$). *Anomiidae* also

Factor 'year' only data from exposed site

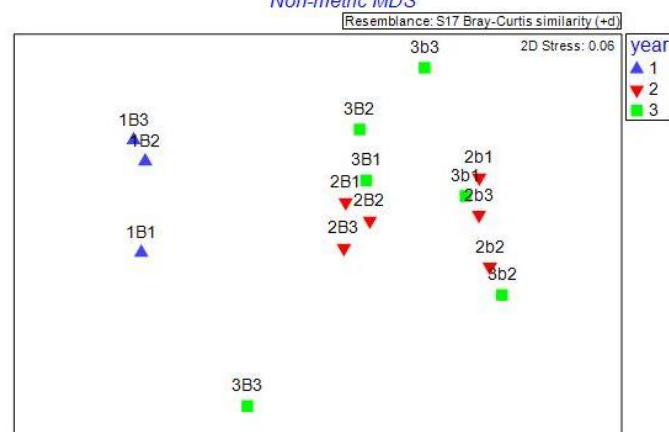


Figure 18: Non-metric MDS plot of only data from the exposed site (Barmur) for the factor 'year'. Data does not include 1st year line April samples.

play a role in sample differentiation ($P=0.0297$), with a statistical difference in between 1st and 2nd year line samples ($P=0.0009$), but no statistical difference in the other combination of years ($P=0.054$ and $P=0.7$ for 1st and 3rd year lines samples and 2nd and 3rd year line sample respectively). Again, two bivalves, not present in the first year.

The species-specific results between the data sets of both sites data and only the exposed site data are comparable. Only *Polychaeta* played a bigger role when looking at factor year in both sites, while *Nudibranchia* played a significant role for the

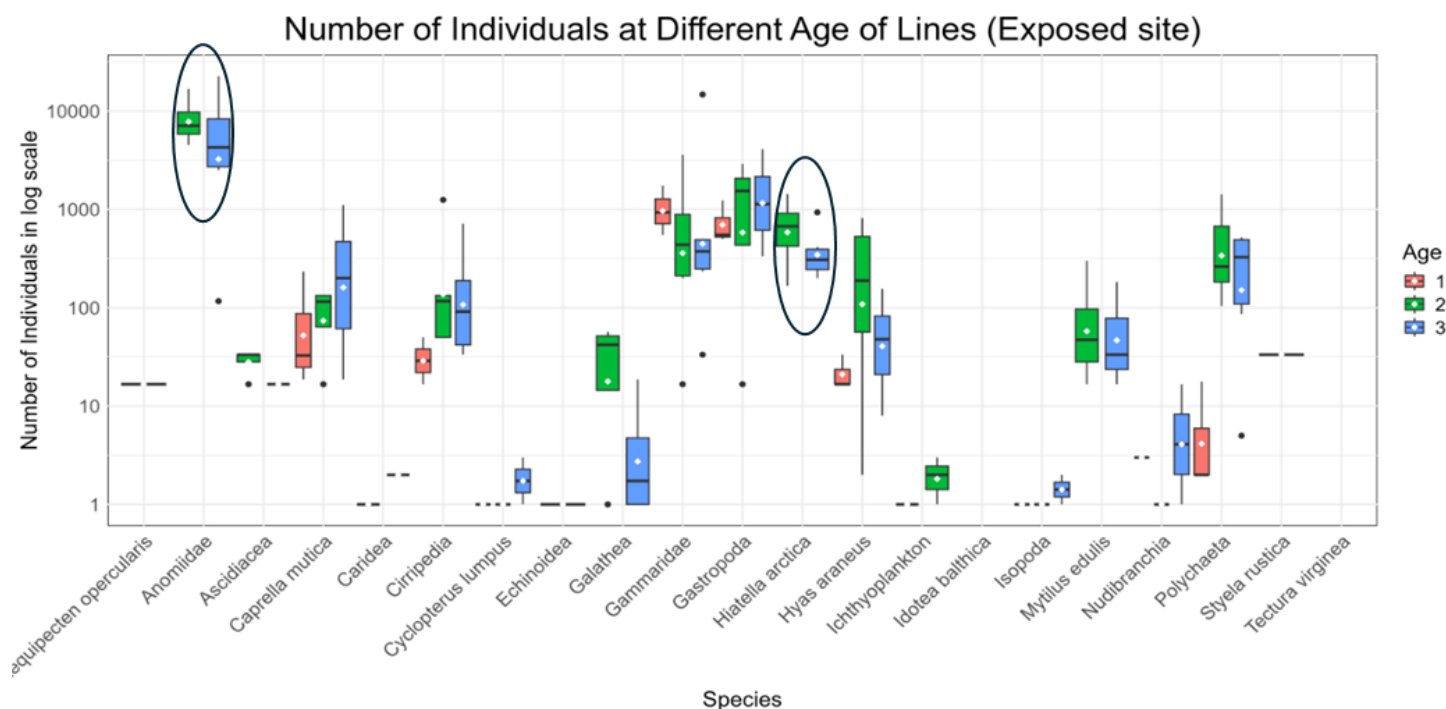


Figure 19: Box plot of only data from the exposed site (Barmur) with both seasons and all three years, with the median, mean (white dot), upper and lower quartile and black dots for minimum and maximum values. Circles show the species responsible for creating the difference in the samples.

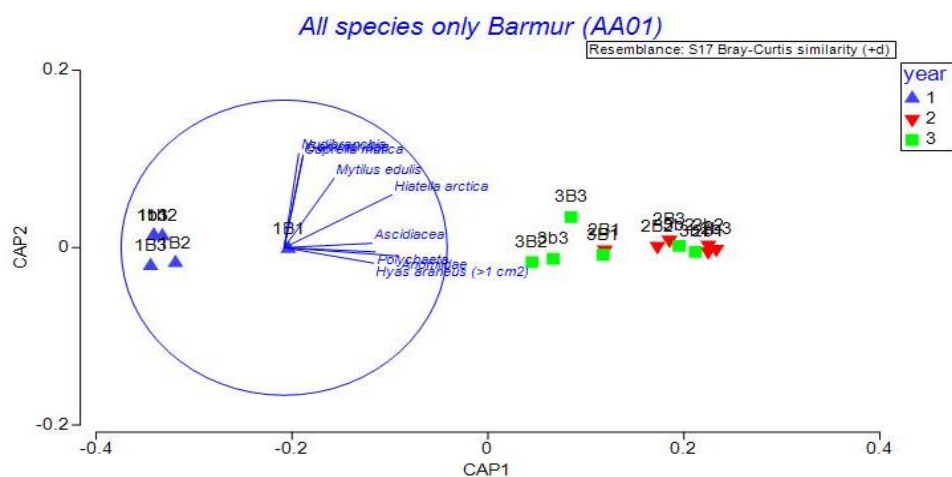


Figure 20: Bray-Curtis similarity of only the exposed site, Barmur (AA01) data displaying the relevant species ($R \geq 0.05$).

data from only the exposed site, but not for data from both sites. However, the data set representing only the exposed site is reduced by a quarter.

2.2.3 Factor 'site'

2.2.3.1 Site including all years

The non-metric MDS graph with factor site (Figure 21) shows a less distinct clustering, supporting the results from Permanova, that factors site ($P=0.0143$) is statistically less different by roughly factor 300 for season ($P=5e-5$) and roughly factor 1500 to the factors year and timeline ($P=1e-5$). Figure 22 shows the difference in abundance of each species in the different sites (exposures). The species driving the differences in samples when looking at the factor 'site' (exposure) were tested with Permanova including Monte Carlo tests and are *Caridea* ($MC=0.0077$), *Ascidiacea* ($MC=0.0175$), *Cirripedia* ($MC=0.0245$) and *Nudibranchia* ($MC=0.0476$), as can be seen in Figure 22.

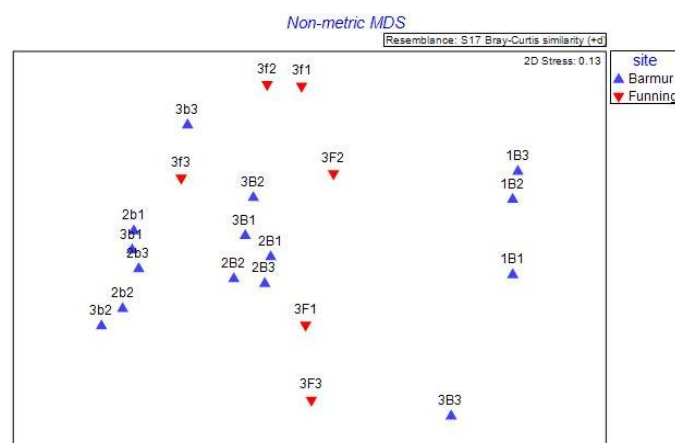


Figure 21: Non-metric MDS plot of the factor site. Data does not include 1st year line April samples.

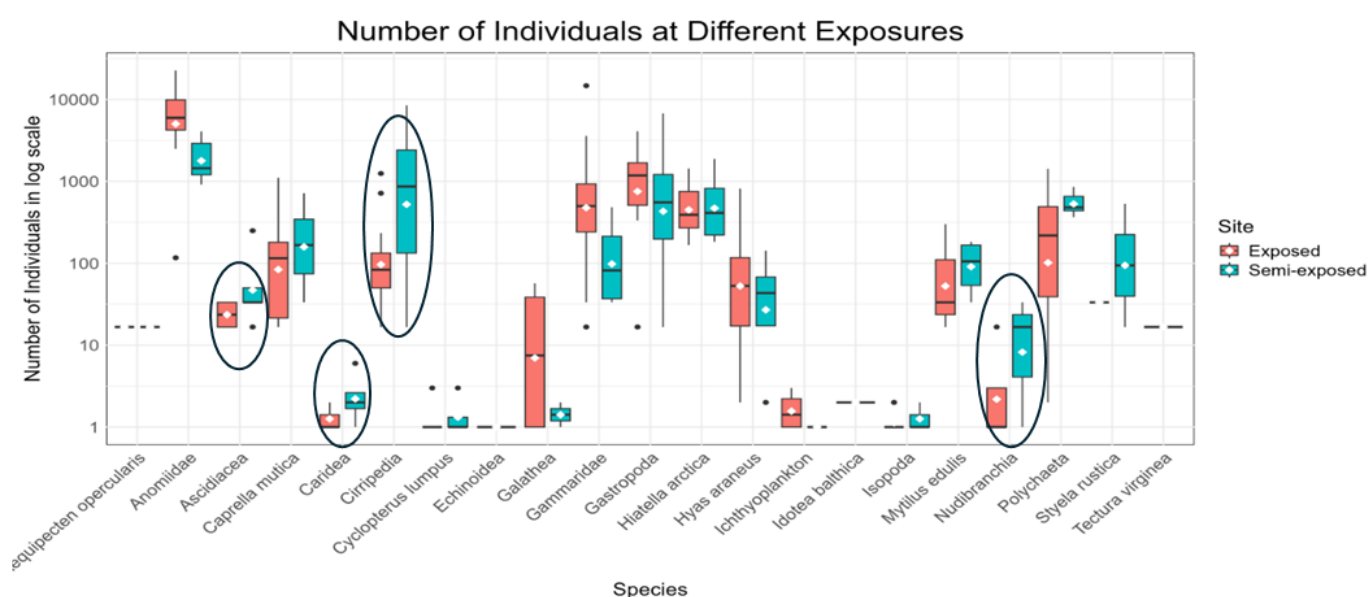


Figure 22: Box plot of all data comparing the sites with the median, mean (white dot), upper and lower quartile and black dots for minimum and maximum values. Circles show the species responsible for creating the biggest difference in the samples.

2.2.3.2 Factor 'site' for only 3rd year line samples

The non-metric MDS graph with factor 'site' for only 3rd year line samples (Figure 23) shows an even less distinct clustering than with data from all years, which is confirmed by the Permanova not being statistically different ($P=0.1149$). It explains that the statistical difference in site in the man test with all years (section 2.2.3.1) occurs due to the 1st and 2nd year line samples from the exposed site being different to the 3rd year line samples from the semi-exposed site, the site with only 3rd year line samples.

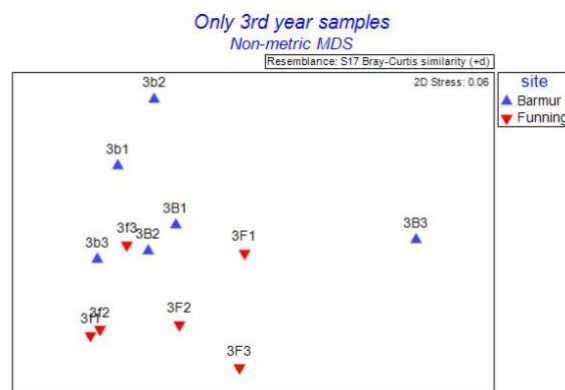


Figure 23: Non-metric MDS graph of only 3rd year data with factor site.

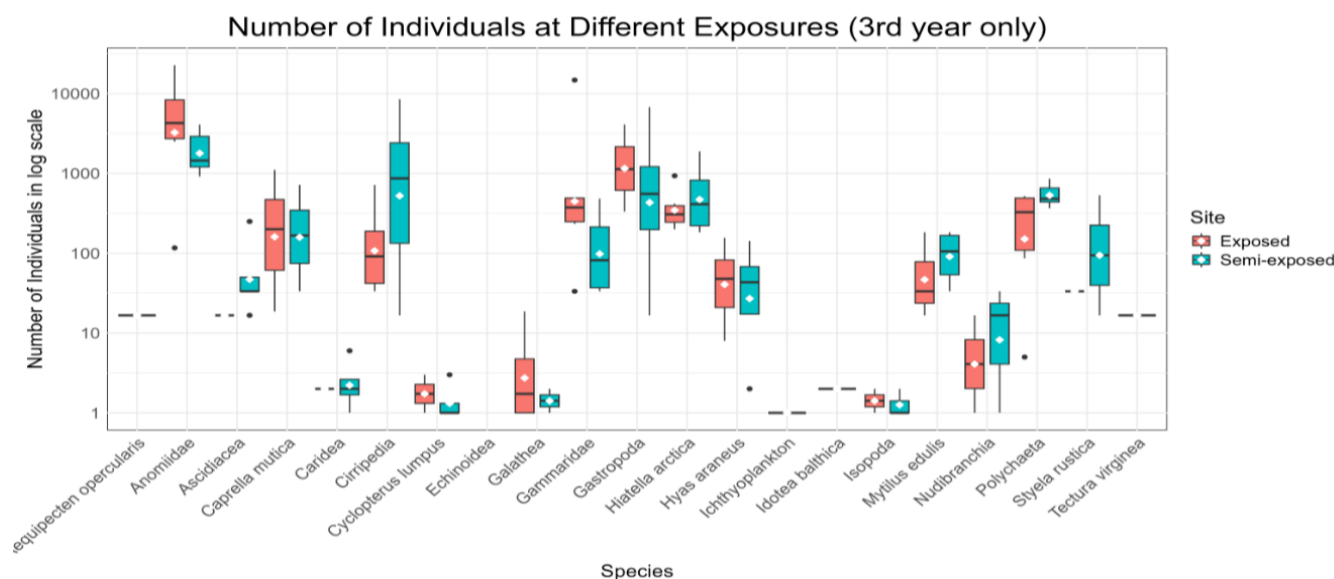


Figure 24: Only 3rd year line data comparing the exposure on the species with the median, upper and lower quartile and dots for minimum and maximum values. Circles showing the species creating the biggest difference in the samples.

2.2.4 Timeline

To understand if there is a significant change to the biodiversity with increasing months of seaweed lines in the water, factor 'timeline' was tested (replacing factor season and year) for our samples.

2.2.4.1 Timeline including data from both sites

The non-metric MDS graph (Figure 25) shows clusters of the samples from 7 growth months (1st year July samples), 19 growth months (2nd year July samples) and 16 growth months (2nd year April samples), while the 27 and 31 growth months samples (April and July of the 3rd year line samples) are wider spread apart. The Permanova shows a statistical difference for the factor 'timeline' ($P=1e-6$). Pairwise test on the resemblance in timeline (combination of each month) shows the biggest difference (lowest P value) when the 1st year line samples are in interaction with the 3rd year line samples (month 4 and 7 to month 27 and 31), Table 4 and Figure 26, see circles in Figure 26.

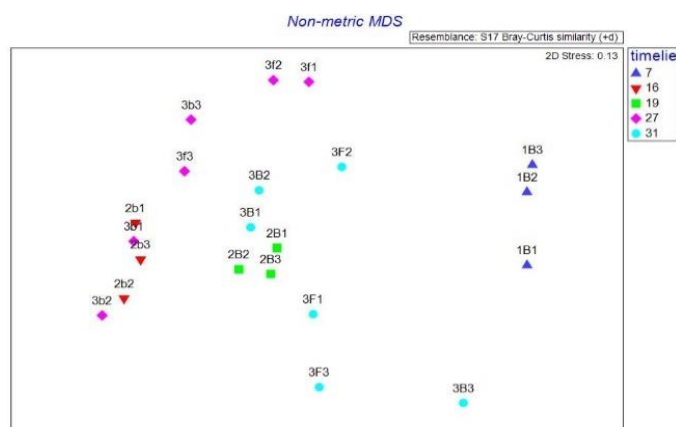


Figure 25: Non-metric MDS plot of the factor 'timeline'. Data does not include 1st year line April samples.

Table 4: Pair wise test of factor timeline for both sites

Growth months	P(pern)
4 => 7	0.0998
4 => 16	0.1003
4 => 19	0.1004
7 => 16	0.1000
7 => 19	0.0997
16 => 19	0.0996
16 => 27	0.5708
16 => 31	0.0474
19 => 27	0.0760
19 => 31	0.8787
27 => 31	0.0118

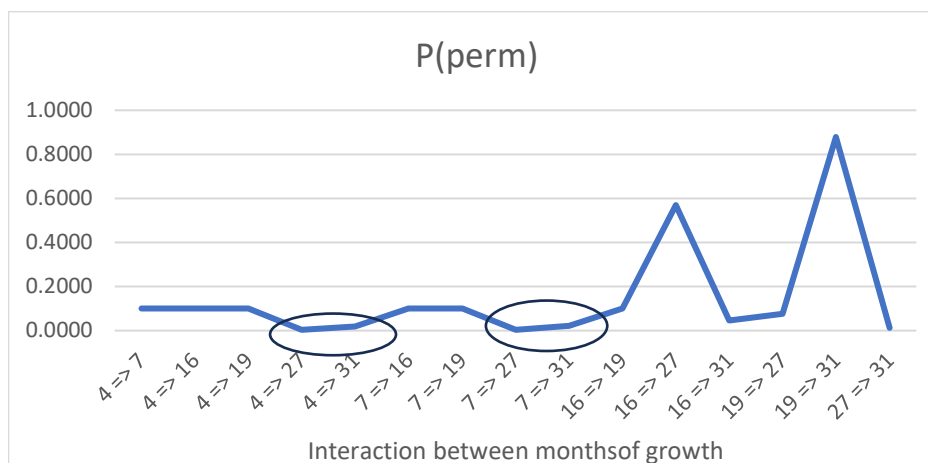


Figure 26: Permanova values of selected timeline interactions, only consecutive ones of data from both sites.

The biggest similarities are April 2nd year line samples to 3rd year line samples and July 2nd year line samples to 3rd year line samples, peaks in Figure 26. When comparing the Permanova values of the consecutive timeline interactions (Figure 27), it can be seen that in the second year, from April to July (months 16=> 19), the interactions start being more significant (lower values). But only in the third year do they start being statistically different ($P < 0.05$). Thus, the development in biodiversity is limited in the beginning, not being statistically different within the steps of 3-5 months, but in the 3rd year changes are significant and statistically relevant and even within 3-5 months significant changes occur. These findings are in line with results from section 2.2.1.2 showing that the

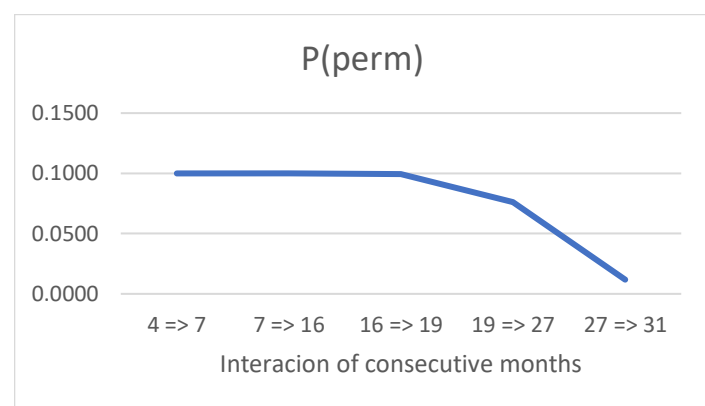


Figure 27: P values of post hoc timeline data of both sites. Months 4 and 7 are within the first year, month 16 and 19 in the second year and month 27 and 31 in the third year.

statistical difference between the seasons grows over the years.

2.2.4.2 Factor 'timeline' only for the exposed site

When considering all the data, including data from the semi-exposed site, it might be adding an additional difference, since only in the exposed site we have data from all three years.

Permanova main test shows a statistical difference in the data from only the exposed site ($P = 1e-5$). The pairwise test shows none of the interactions of the months are statistically different anymore.

2.2.5 Summary of role the factors play for biodiversity

The statistical tests confirm that the biggest difference in sample is between the years, less so for season and the different exposures in the sites showed no difference. The Permanova pairwise test confirms that 1st and 3rd year line samples are most different ($P = 1e-6$), 1st and 2nd year line samples are also very different ($P = 1e-5$), while 2nd and 3rd year line samples are quite similar and not statistically different ($P = 0.5692$). Factor years within the seasons and years only for the data of the exposed site show a similar pattern.

In general years are more different in April than they are in July. The same pattern is there when looking into only one site, the exposed site, yet here 2nd and 3rd year line samples are not significantly different anymore.

The statistical difference between the seasons grows over the years, only becoming statistically relevant in year 3. Within both seasons only 1st and 3rd year line samples are statistically different. In general, the years are more different in April than they are in July.

When looking at consecutive interactions of the growth months, only in the third year they start being statistically different. Thus, the development in biodiversity is limited in the beginning, not being statistically different within 3-5 months, but in the third-year changes are significant and statistically relevant within 3-5 months.

For the factor 'year' *Polychaeta* are most relevant, then *Anomiidae* and *Hiatella arctica*. The last two species are also the most relevant ones when looking at data only from the exposed site having all year line samples. For factor 'season' *Gastropoda*, *Caprella mutica*, *Hyas araneus* and *Ichthyoplankton* are key responsible for the distinction between the samples in season.

2.3 Shannon index

The Shannon index, considering the species richness and species evenness, was determined for each sub sample set, comparing the biodiversity development within the three factors 'season', 'year' and 'site', Table 5.

The Shannon index is 0 for the 1st year line April samples, as only 1 species was quantified (*Isopoda*), further species were found (*Gammaridae*) but were not able to be included in the quantification as they were part of the mobile sampling and were not bigger than 1cm. The comparison of the Shannon index for site, season and year shows that July (H=1.32) had in general a bigger index than April (H=0.55) and the 3rd year line July samples in the semi-exposed site had the biggest index (H=1.57), followed by the 2nd year line July samples in the exposed site (H=1.52).

3. Discussion

3.1 Our findings

In our samples, organisms were determined down to 21 different faunas. As there was no expert taxonomist on the team, large and difficult groups such as the *Polychaetas* were not determined in detail, and other groups such as *Bryozoa* or *Cnidarians* were not quantified due to the difficulty in quantification. The general species found within this study are either mobile or have a pelagic larval stage, e.g. *Galathea* (Lovrich & Thiel, 2011). The only non-native species found was the *Caprella mutica*. It is native to Japan but has spread widely, likely as fouling on ship hulls (Ashton et al., 2007). It does not have a planktonic larvae stage, limiting dispersal (Ashton, 2006), but has been found in the Faroe Islands since the 90's (Pers. comment Jan Sørensen). It has a wide diet, ranging from detritus, macro- and microalgae and amphipods (Turcotte et al., 2009) and is prey to *Malacostracans* and fish (Woods, 2009).

The seasons influenced the abundance and diversity of the organisms. From April to July, there was a decrease in *Anomiidae* and *Galathea*, though both were still present in July. The decrease in *Galathea* may be due to the larval and juvenile stage being pelagic, and the adult stage is primarily benthic (Lovrich & Thiel, 2011). It would be expected to migrate to its natural habitat, where it is also important prey to top predators such as fish and marine mammals (Lovrich & Thiel, 2011).

The largest increase from April to July was in *Gastropoda*, *Cirripedia* and *Gammaridae*, all important prey animals to other invertebrates, fish and marine mammals (Cowden et al., 1984;

Ponder et al., 2020; Sheader, 1983). Other significant increases included *Caprella mutica*, *Hyas araneus* and fish larvae.

Both *Gammaridae* and *Gastropoda* were plentiful on all ages of line, but *Anomiidae* was the most common on second- and third-year lines. The organisms with the largest spread from first to third year lines were *Polychaeta*, *Hiatella arctica* and *Anomiidae*. These are sessile organisms that need time to settle and were usually found living among the holdfast of the seaweed. The holdfasts of older seaweed were larger and more entangled than the new seaweed blades, which offered plenty of substrate to attach to for bivalves or hide in between for *Polychaeta*. The *Polychaeta* larvae can travel long distances while the adult is relatively inert, and reproductive rates are generally high for this group (Dean, 2008). They are particularly interesting for their ability to act as bioindicators for the benthic environments as this group is high in species and abundance (Murugesan & Swain, 2025). They prey on and can act as control of the abundance of invertebrates and are preyed upon by *Malacostracans* and fish (Dean, 2008).

There was a small difference in site, and the semi-exposed had more *Caridea*, *Ascidacea*, *Cirripedia* and *Nudibranchia*. Compared to the season and age of lines, this was the parameter that showed the least significant difference of our tested parameters. This finding supports the development of new exposed seaweed cultivation sites.

Generally, no species is associated with a negative impact to the fjord ecosystem (personal communication with Petur Steingrund, University of the Faroe Islands), as these naturally migrate to and from seaweed cultivation within the fjord. As there is relatively low presence of predators inside the seaweed cultivation site, it can act as a refugium for millions of prey invertebrates and fish larvae. This could in turn make the fjord itself more reproductive, as these prey species will immigrate and emigrate from the cultivation throughout their lifespans.

3.2 Other studies on invertebrates in cultivated kelp

In the Gulf of Maine studying *S. latissima* and *S. angustissima* the most dominant invertebrates were amphipods (*Lembo websteri*), copepods (*Paracalanus sp.*) and the skeleton shrimp (*Caprella linearis*) (Schutt et al., 2023). These results compare to our first-year line sample results, where also amphipods, scuds (*Gammaridae*), are most dominant, followed by sea snails (*Gastropoda*) and skeleton shrimps (*Caprella mutica*). A study in Norway found Isopods (*Idotea pelagica*) as a dominant species in the younger forests (3 months after seeding) and amphipods

Table 5: Shannon index of averages of replicates.

	Exposed site					Semi-exposed site	both sites	
	April	July	1st	2nd	3rd	3rd	3rd	all data
April	0.321		0	0.536	0.429	1.238	0.833	0.550
July		1.235	0.951	1.516	1.240	1.567	1.403	1.318
Both seasons	0.7799		0.476	1.026	0.834	1.402	1.118	0.935

(*Caprella*) as a dominant species in the older forests (7 months after seeding) (Bekkby et al., 2023). We found those species also belonging rather to the April and July season respectively, even though they were not dominant in our samples. The species found in the other two studies are mobile species, while the most abundant organism in our study was *Anomiidae* bivalves, as well as other sessile fauna such as *Cirripedia* and *Ascidacea*, which are abundant on second- and third-year lines. This can be explained by the shorter time other seaweed cultivation structures are in the water, while ORF cultivation process provides enough time for bivalves to settle.

3.3 Comparing cultivated kelp to natural kelp forest

To understand how our quantified biodiversity could be compared to the natural surrounding ecosystem, results from other temperate seaweed studies on biodiversity abundance were reviewed. The 3 year-long deployment cycle of seaweed lines from ORF, allows for to leave most of the organisms intact, as many reside in the holdfast or are mobile species. The holdfast is usually the part of the seaweed with the highest abundance and diversity of invertebrates (Katharoyan et al., 2024). In Ireland, Walls et al.(2016) studied if the habitat created by the holdfast would be different on cultivated seaweed (*Laminaria digitata*) (suspended and growing on ropes), than from natural seaweed (benthic). The study found that the total abundance of holdfast epifauna was similar, while species richness was higher at the holdfasts from the cultivation site, however species assemblage composition was different, thus representing a novel habitat (Walls et al., 2016). Another study in Norway also tested the difference in communities hosted by a kelp farm (*S. latissima* and *A. esculenta*) and natural kelp forests (*S. latissima*, *A. esculenta* and *Laminaria hyperborea*) and found that the farm had a lower taxa abundance, but showed many kelp associated species (Bekkby et al., 2023).

A studies from the UK on *S. latissima* found an increase in fish abundance relative to their reference site (Corrigan et al., 2024), while another study from the Gulf of Main on *S. Latissima* and *S. angustissima* didn't find an impact on the species abundance (Schutt et al., 2023).

3.4 Food and habitat for fish

Seaweed cultivation sites could provide habitat, spawning grounds, or shelter for commercially or ecologically important fish (Bergman et al., 2001; Corrigan et al., 2024; Kelly et al., 2020; Theuerkauf et al., 2022). A recent study from the UK on *S. latissima*, which observed more fish in the cultivation than in the reference site, found that low trophic level fish had high proportions of amphipods in their stomachs, which was the most dominant species found within their seaweed study site (Corrigan et al., 2024). The researchers further analyzed stomach contents of higher level species fish and concluded that seaweed cultivation provide new foraging grounds and support secondary and tertiary production (Corrigan et al., 2024).

The only identified fish found in our samples was the lumpfish (*Cylopterus lumpus*), which is also found in other suspended

aquacultures in Sweden (Hasselström et al., 2018) and the Gulf of Maine (Schutt et al., 2023). The lumpfish is valued for being a biological control agent in salmon aquaculture and also valued for their roe (Imsland et al., 2014; Powell et al., 2018). The invertebrate species found in our samples have also been found in fish stomachs on the Faroe Shelf (Pers. comment Petur Steingrund), leading us to conclude that fish could potentially be feeding on invertebrates from our cultivation. Commercially interesting fish species such as cod are spawning from February to April in the Faroes (Gaard & Steingrund, 2001; Jacobsen et al., 2020), and food availability are among the key factors affecting the growth and survival of fish larvae (Anderson, 1988; Cushing, 1990; Leggett & DeBlois, 1994). The early life stages of cod are very opportunistic and adaptive in relation to variability of the abundance, species of composition and size of prey (Heath & Lough, 2007). Late juvenile cod fish prefer Malacostracans (Heath & Lough, 2007), which we found plenty of in our study, but at low prey abundances cod may predate on less preferred prey (Seljeset et al., 2010) such as *Cirripedia* larvae which also were present at our samples. The stomach contents from cod caught on seabed with a depth of less than 200 m at the Faroe Plateau show that the dominant diet of cod consists mainly of sand eels and benthic crustaceans including *Galathea* sp. and *Hyas* sp. (Rae, 1967), also found in our samples.

Looking at fish abundance around seaweed cultivation units compared to reference sites in other regions, studies differ in their outcomes. A review on studies assessing habitat creation using control-impact or before-after experimental design found that bivalve and seaweed aquaculture had higher abundance and species richness of wild, mobile macrofauna, compared to their reference sites (Theuerkauf et al., 2022). Another review reconfirmed these results and found that farm bivalve and seaweed habitats (weighted mean of 273 ind. ha⁻¹) had a higher abundance of targeted fish than the reference site (weighted mean of 130 ind. ha⁻¹) (Barrett et al., 2022). The study further detailed that when it comes to density of rather large and mobile fish species, it was found to be high around tropical seaweed farms (0,3–1,15 m⁻²), however stating that at the reference sites the densities were generally similar or higher (0,37–1,4 m⁻²) (Bergman et al., 2001; de Carvalho et al., 2017; Hehre & Meeuwig, 2016; Theuerkauf et al., 2022).

3.5 Longer duration of the created habitat

It has been questioned what the effect on biodiversity would be after harvest, where the created habitat as well as some of the marine life within is abruptly removed (Corrigan et al., 2022). Questions remain whether seaweed cultivation will still have a positive effect on biodiversity, as it temporarily helped juveniles find shelter and thus survive, or if it even created a biodiversity sink. Cultivation gear is suggested to be an important factor to increasing biodiversity (Forbes et al., 2022; Schutt et al., 2023). Forbes et al., (2022) furthermore detail, that seaweed cultivation is usually too short to create a meaningful habitat comparable to natural kelp forests (Forbes et al., 2022). Taxa richness has shown to be higher after a partial harvest, compared to a full harvest in a study in the UK on *S. latissima*

(Corrigan et al., 2023, 2024). ORF leaves their structure in the water, growing seaweed for up to three years, potentially creating a habitat closer to the natural seaweed ecosystem, than other common seasonal seaweed aquacultures. With a rotating system of only adding newly seeded 1st year lines in places where 3rd year lines had been harvested, leaves year-round roughly two thirds of the cultivation structure with partially harvested seaweed every year in place. This allows at least mobile species to have a continuous habitat. The increase in biodiversity over the years is also seen in the Shannon index, being bigger in the 2nd and 3rd year than the 1st year for our samples, which only have mobile species present. More research is needed to better understand the impact harvest or partial harvest can have on the biodiversity in and around cultivation to confirm the benefits seen on this site with the partial harvest and habitat created for a longer period for sessile species and continuous habitat created for mobile species.

3.6 Potential negative impacts

More fish or invertebrates aren't always inherently positive. The impact these species have on the ecosystem function needs to be assessed (Corrigan et al., 2022). A review even concluded that seaweed cultivation creates new habitats that support distinct communities, differing to natural kelp forests and are not a replacement for natural kelp (Corrigan et al., 2024; Forbes et al., 2022). Furthermore, depending on the perspective and context, seaweed cultivation units can also be seen as 'ecological traps' where the fish are more vulnerable to capture or natural predators like seals (Alleway et al., 2018; Kelly et al., 2020; Theuerkauf et al., 2022), essentially a trap for all animals not capable surviving the cycles of commercial harvesting and replanting (Corrigan et al., 2022; Forbes et al., 2022; Theuerkauf et al., 2022). Corrigan et al. (2024) also showed that fish abundance and richness decline after harvest, confirming that seaweed cultivation is only a temporary habitat, when harvesting fully and taking out the whole cultivation structure (Corrigan et al., 2024).

Seaweed cultivation can have several impacts on the benthic community, for example via shading, changes to hydrodynamic flow, increased sedimentation, organic enrichment, etc. (Campbell et al., 2019; Corrigan et al., 2022; Hasselström et al., 2018; Langton et al., 2019; Wood et al., 2017). For example, when seaweed cultivation units are above seagrass beds, it can have negative impacts on the macrofaunal abundance and seagrass density (Eklöf et al., 2005). In general, it has been shown that effects on the benthic ecosystem are less severe in temperate ecosystem with suspended cultivation (Visch et al., 2020; Walls et al., 2017). Negative impacts on the benthic community structure are very unlikely for the ORF site, as the water depth is minimum 20 m and there are no seagrass beds below.

3.7 Research needs

Questions remain, and more studies are needed to quantify the impact of seaweed cultivation on the surrounding biodiversity. , via invasive species (Bernard et al., 2019)? Questions on the additionality of habitat created need to be

investigated: Does seaweed cultivation only serve as a biodiversity aggregation or really create additional habitat and richness in species (Gentry et al., 2020; Hasselström et al., 2018; Radulovich et al., 2015; Theuerkauf et al., 2022)?

It is important to mention that seaweed cultivation can still have other negative impacts and thus pose risks such as invasive species, entanglement of megafauna, littering, competition for nutrients, etc. (Banach et al., 2022; Campbell et al., 2019; Forbes et al., 2022; Grebe et al., 2019). These impacts also need to be researched and quantified to avoid harm to the ecosystem. The intensity and management of the cultivation play an important role in the magnitude of these potential impacts (Campbell et al., 2019; Grebe et al., 2019), please also view D9.2 section 4 for more details on carrying capacity of an ecosystem.

CHAPTER III: CARBON SEQUESTRATION

This chapter assess the carbon sequestration potential for the three sites: ORF in the Faroe Islands, OF in Norway and ALO in France. In the Faroese site. Data on lost particulate organic carbon (POC) was also collected to gain an understanding of how much more carbon is potentially sequestered due to a year-round POC loss, compared to only a few months like in other seaweed cultivation units. The carbon sequestration potential for all three sites is calculated based on carbon content from the sites and values from literature.

1. Methodology

1.1 Sampling

1.1.1 Dislodgement rate at the ORF site

Carbon sequestration potential calculations consist of estimates on dissolved organic carbon (DOC), particulate organic carbon (POC), burial rates and percentages of DOC that contribute to sequestration as it is not being remineralized. The latter is so called refractory DOC (RDOC). Lost POC consists of three different processes for which data can be collected:

- 1) Erosion: when the tip of the blade erodes away, mostly towards the summer months)
- 2) Exudation: when parts of the blades break off
- 3) Dislodgement: where entire blades rip off the growing line

For the Faroese site, dislodgement rates data was collected, as this was expected to be the biggest contributor to POC loss for the Faroese site.

For dislodgement data, 30 cm samples at 1.5 m depth of three selected vertical lines (replicates) at the beginning, middle and end of the 750 m long horizontal growing lines were taken during the winter months and nine vertical lines (three selected replicates of different year lines) of the 750 m long horizontal growing lines during the summer months, covering both sites (exposed and semi exposed). Whenever 30 cm had already been collected on a cultivation line, another line was taken to

ensure independent data collection. The number of blades (blade density), length and width were measured. Overall, sampling took place over 10 months (Nov 2023 – Aug 2024), covering blades being in the water for up to 41 months.

In April, July and December 2024, 10-20 randomly selected blades were collected, weight and length were measured to graph the length to weight correlation of a growing blade.

Additionally, a data set from 2014 – 2017 was used to complement the blade density data, average growth, carbon content and dry weight to wet weight conversion.

1.1.2 Carbon content and DW/WW ratio

For the assessment of the potentially sequestered carbon, the carbon content and DW/WW ratio was used from the ORF, OF, and ALO seaweed cultivation sites.

1.2 Literature search

Based on a recent literature review on carbon sequestration potential of seaweed cultivation, data and equations for the calculation of the total carbon sequestered were taken (Pessarrodona et al., 2024). Other more recently published papers were added to complement the set of data from literature.

To calculate carbon sequestration potential in the marine environment the following equations were used based on (Krause-Jensen & Duarte, 2016; Li et al., 2024; Luo et al., 2024; Pessarrodona et al., 2024) and compared to (G. Gao et al., 2022; Song et al., 2023), representing the net carbon fixed by seaweed during cultivation, the amount that is harvested, and the fate of unharvested carbon. In our analysis carbon sequestration potential in the algal bed is considered negligible as strong tidal currents have shown that there is hardly any sedimentation below the farm (Pers. comm. with Grandorf-Bak, 2024).

C_{seq} was calculated as:

$$C_{seq} = C_{deep} + C_{sed} + C_{RDOC} \quad (1)$$

Where C_{seq} is the total kg of C sequestered, C_{deep} the carbon from Particulate Organic Carbon (POC) that is sequestered in the deep sea, C_{sed} the POC sequestered in the sediment and C_{RDOC} is the fraction of the carbon released as Dissolved Organic Carbon (DOC), which is resist degradation entering the refractory DOC pool.

C_{deep} (2) and C_{sed} (3) were calculated as:

$$C_{deep} = C_{POC} \cdot \%C_{POCloss} \cdot \%C_{POCburialdeep} \quad (2)$$

$$C_{sed} = C_{POC} \cdot \%C_{POCloss} \cdot \%C_{POCburialsed} \quad (3)$$

Where C_{POC} is the amount of carbon fixed by the potential amount of seaweed grown and calculated as

$$C_{POC} = \frac{\text{Harvested POC}}{1 - \%C_{POCloss}} \cdot \%C_{POCloss}$$

is the percentage of assimilated carbon lost before harvest as rip off, erosion or break off (exudation), $\cdot \%C_{POCburialdeep}$ is the percentage of POC that is buried in the sediment of the deep

sea and $\cdot \%C_{POCburialsed}$ is the percentage of POC that is buried in the sediment of the continental shelf.

C_{RDOC} was calculated as:

$$C_{RDOC} = C_{FIX} \cdot \%DOC \cdot \%RDOC + C_{POC} \cdot \%POCloss \cdot \%POC-RDOC \quad (4)$$

Where C_{FIX} is the amount of carbon fixed in farmed seaweed, $\%DOC$ is the percentage of carbon that is released as DOC, $\%RDOC$ is the percentage of DOC that turns into refractory DOC.

$$C_{FIX} \text{ was calculated as } C_{FIX} = \frac{C_{POC}}{1 - \%DOC}$$

The values $\%DOC$, $\%RDOC$, $\%POCloss$, $\%C_{POCburialdeep}$, $\%C_{POCburialsed}$, and $\%POC-RDOC$ were taken from literature (Table 3), while, Harvested POC, Carbon content, dry weight to wet weight conversions came from the sites. For $\%POCloss$, we screened the literature for POC (erosion, exudation and dislodgement) studies and narrowed down the data sets to our species *S. latissima* and studies taking place in the North Sea and Northern Atlantic. The studies for $\%DOC$, $\%RDOC$ and $\%POC-RDOC$ are numerous but vary a lot ranging from global studies covering several species to studies concerning one species. We focused on the studies cited in the recent literature review and chose, when possible, the ones on *Saccharina Japonica*, being close to *S. Latissima*. There are also minimum and maximum values for the burial rates, but most studies refer to one value, which was also used by us.

The overall carbon sequestration potential for each cultivation site was calculated based on equations 1-4. To convert this value into CO₂ equivalent, we multiplied the organic carbon content of the biomass by 3.67 (=44/12), i.e., the ratio between the CO₂ molecular and C atomic weights, to obtain the corresponding stored CO₂.

For the equations, the sampled data on carbon content and DW/WW ration were complemented with data from literature on DOC, POC, burial rates and RDOC, and RDOC.

1.3 Data analysis

Python and excel were used to plot the data and find the best fit functions. For the blade density loss (dislodgement rate), the best fit function, an exponential decay, was modeled with Python. Data was only taken from the first April after deployment onwards (4 months in the water), as the blades are too small before that, to have significant importance in biomass loss.

To calculate the biomass lost due to dislodgement, we used the following calculations:

Total carbon lost by dislodgement:

$$C_{POCdis} = \frac{DW}{WW} \cdot \frac{gC}{gC} \cdot POC_{dis}(WW) \quad (5)$$

Where C_{POCdis} is the total carbon lost by dislodgement, $\frac{DW}{WW}$ is the dry weight to wet weight ratio, $\frac{gC}{gC}$ is the grams of carbon in the dry weight of seaweed, $POC_{dis}(WW)$ is the wet weight of the biomass lost due to dislodgement.

$POC_{dis}(WW)$ and is calculated:

$$POC_{dis}(WW) = \sum_{n=1}^3 \# \text{ of } m \text{ of } n \text{ year line} \cdot \# \text{ of blades lost in } n \text{ year} \cdot \text{average } WW \text{ of blade in year } n \quad (6)$$

2. Results

2.1 Dislodgement of blades on the ORF site

2.1.1 Dislodgement rate

The blade density data for each site and both sites together is graphed and fitted with an exponential decay function (Figure 28). Most blades are lost during the first year after deployment.

Taking the changing blade loss over the years into consideration, dislodgement rates were calculated for the different years. The average number of blades loss per year is shown in Table 6.

2.1.2 Average blade length and weight

ORF used to harvest twice in the first year (May and August), now once a year (August), we calculated the blade length and weight for both scenarios.

The average blade length for both scenarios was calculated, Figure 31. The average length of a first-year blade, when

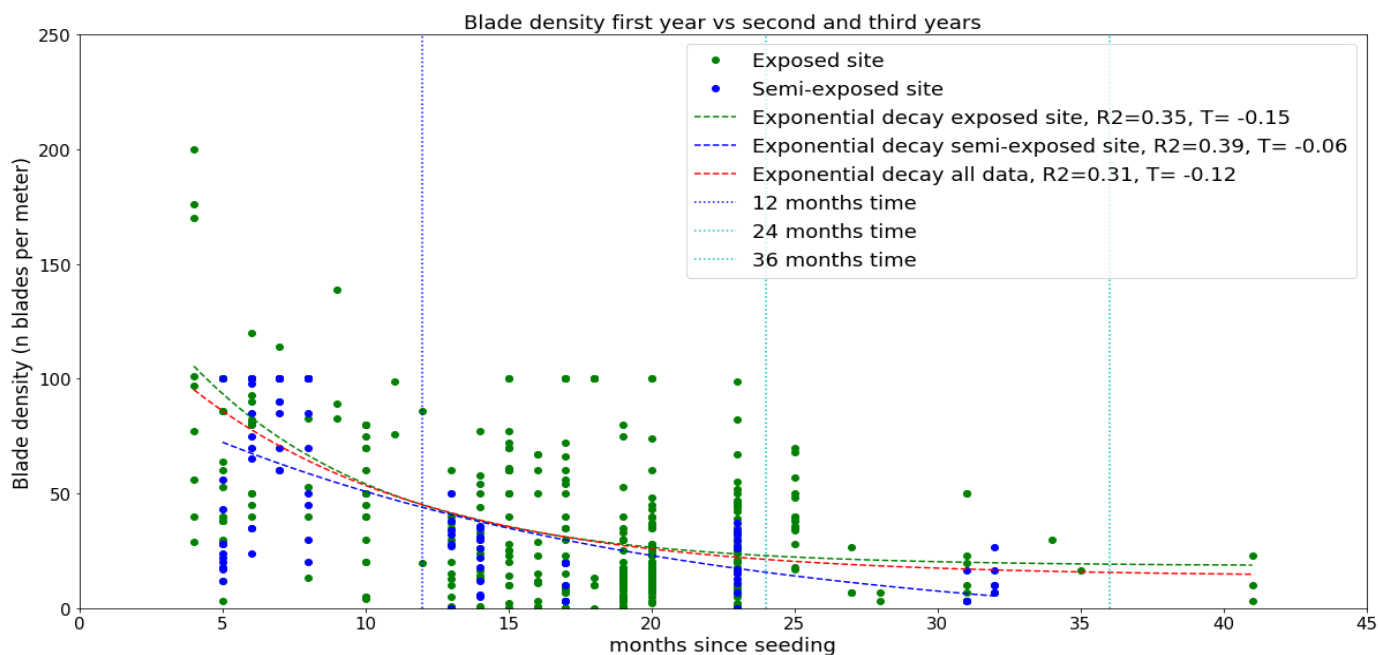


Figure 28: Blade density total numbers, showing the difference in the sites. Light green dots are data from the exposed site and red dots are data from the exposed site.

Table 6: Average number of blades loss per year for the exposed site, based on exponential decay fit function and average per month.

Method of calculation	Blade loss in year		
	1	2	3
Based on exponential decay fit	75.25	33.95	21.05
Average per month	76.4	30.5	24.3

Additionally, a boxplot was graphed showing the average blade density per month with standard deviation error bars of the exposed site, figure 29. The monthly variation and seasonal density fluctuations become visible. Drops in the blade density can be seen in the winter months (Month 11-13 and Month 25-27).

The blade loss over the winter months, as seen in Figure 29, can also be visualized when arranging the data per 12 months. Figure 30 shows the average blade density with standard deviation for the exposed site.

harvesting twice was 58.4 cm, when harvesting once was 74.6 cm. The average blade length of a second- and third-year blade was 86.1 cm.

To calculate the biomass lost due to dislodgement calculation, the average weight of an average blade within each year was taken from a weight length correlation graph, fitted with a power function (Figure 32). Zhang et al. (2012) also found this function to describe best this correlation. The average weight of a blade within the first year with two harvests is 30.2 g, with only one harvest is 52.3 g. The average weight of a 2nd and 3rd year blade is 72.1.

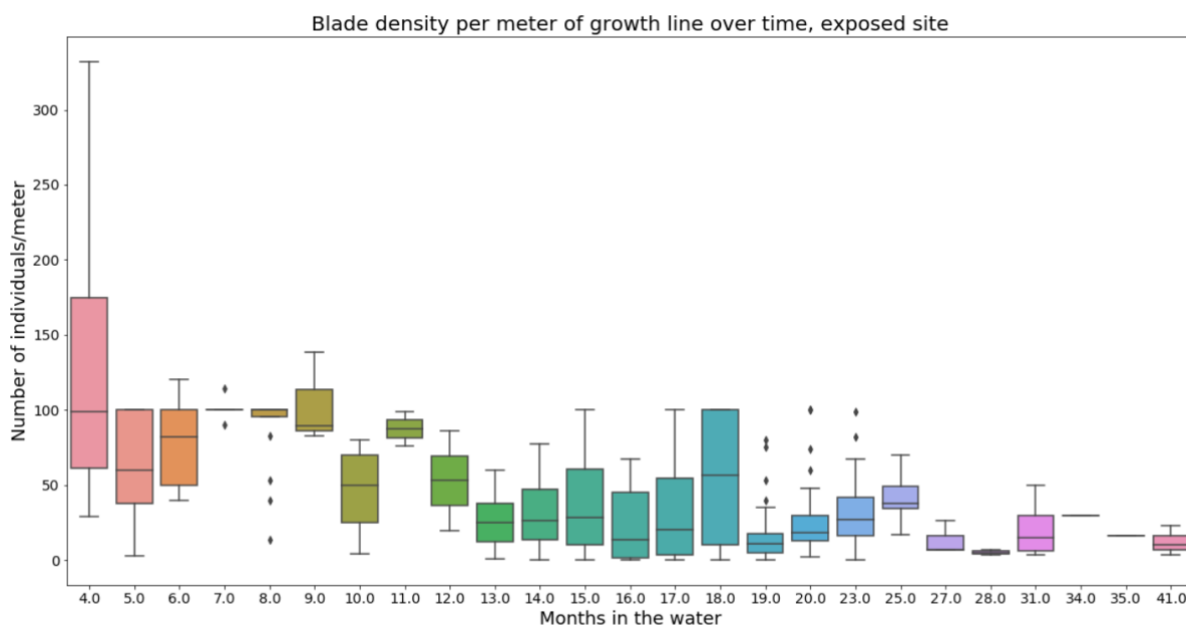


Figure 29: Boxplot of blade density at the exposed site.

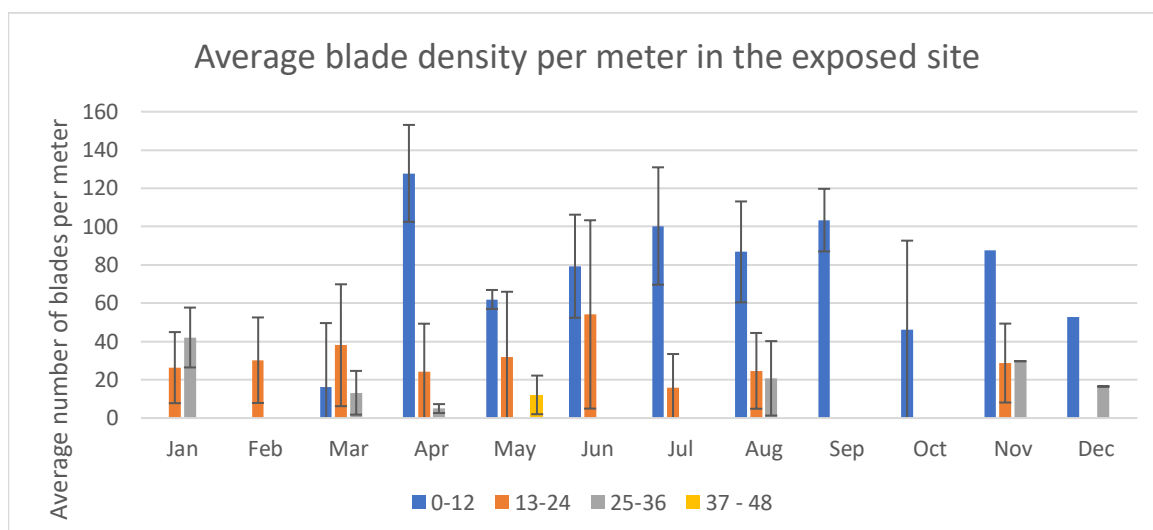


Figure 30: Blade density per meter in the exposed site with 4 different years of blades. 1st year blades are dark blue (in the water for 0-12 months), 2nd year blades are orange (in the water for 13-24 months), 3rd year blades are green (in the water for 25 – 36 months) and 4th year blades are light blue (in the water for 37 – 48 months).

2.1.3 Total POC lost due to dislodgement

The POC lost due to dislodgement in the exposed site is 2,900.2 kg C with the scenario of 2 harvests and 3,633.7 kg C with only one harvest. Assuming that the semi-exposed site dislodgement rate would be the same, that site would lose 991.6 kg C (no need for two scenarios, since there were no first-year lines available in 2024) and both sites would lose 3,891.8 kg C with two harvests and 4,625.3 kg C with one harvest. This POC loss from both sites equivalents to 153,286 kg WW lost when harvesting twice and 181,176 kg WW lost in the case of 1 harvest.

In literature the POC loss in the carbon sequestration potential calculations is expressed as a percentage of NPP. We did not estimate the NPP of the ORF production, neither did we have

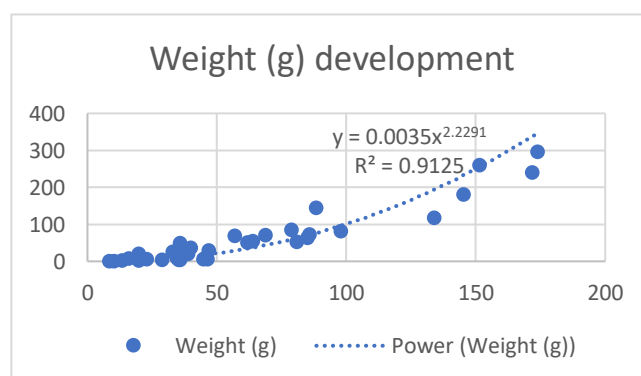


Figure 31: Weight length correlation of *S. latissima* blades.

erosion and exudation data and thus could not translate the dislodgement rates to a comparable unit, % of NPP lost due to dislodgement. We can thus only express it in comparison to the harvested biomass.

The calculated biomass loss equivalents to an average loss for the year 2024 of 1.93 kg WW m⁻¹, while the harvest yield for the same year was 1.51 kg WW m⁻¹. This could mean a potential biomass loss of 129% compared to the harvested biomass.

2.2 Lost POC from literature

Table 7 shows the values for burial rates in the dep sea (POC burial deep) and in the sediment (POC burial sediment), %DOC, %RDOC and % (POC – RDOC), based on the values from a recent literature. If there was no data available on *S. latissima*, values for *Saccharina Japonica* were used.

The literature was screened for POC (erosion, exudation and dislodgement) studies and narrowed down to data sets on *S. latissima* and studies taking place in the North Sea and Northern Atlantic. Three comparable studies for POC (erosion, exudation and dislodgement) on *S. latissima* were found (Table 8). Canvin et al. (2024) studied the dislodgement and erosion of *S. latissima* in the South-west of England, in Cornwall, in an exposed site, calculating that 35% of the assumed growth, also called net primary production (NPP) is lost due to dislodgement and 35% as well to erosion or exudation. Dolliver & O'Connor (2022) studied *S. latissima* in Northern Ireland at a relatively sheltered site (currents 0.3 m s⁻¹) and calculated dislodgement to be 41% of the NPP, exudation 43% and erosion only very little with 0.2% of NPP. A study done by Fieler et al. (2021) on two sites (mid Norway and north Norway) calculated the POC loss either at 8.1% or to be up to 49.4%. The latter is the case, when the lines stay in the water until August, which is closer to the case of the Faroe Islands. The estimated erosion was low, while dislodgement was the main part (around 40% in the high POC loss case).

2.3 Carbon sequestered

For consistency in the carbon sequestration potential calculation, dislodgement rates from literature were used. However, for the Faroese site, higher dislodgement rates from the study in Norway were taken displaying dislodgement rate when leaving blades in the water as compared to optimized

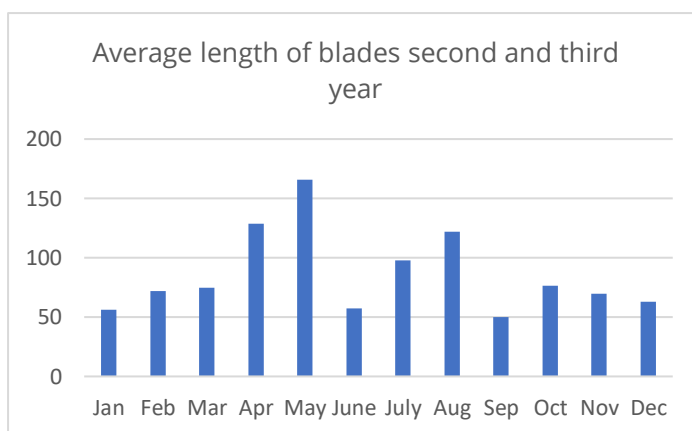
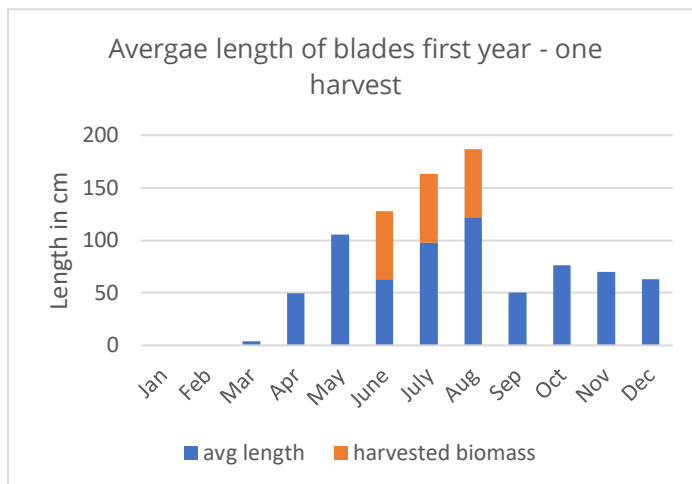
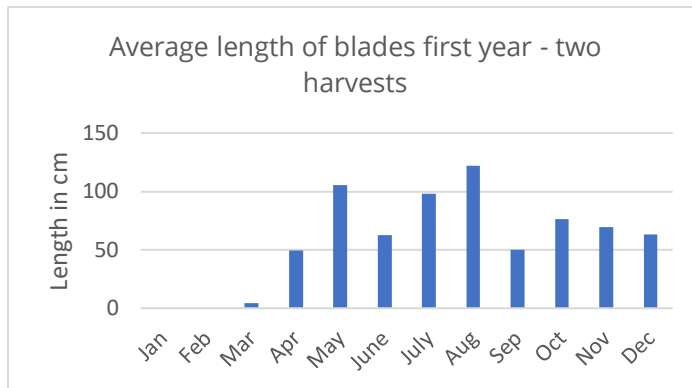


Figure 32: Average length of blades (cm) of first year blades with two harvests (top), only one harvest (middle) and for the 2nd, 3rd and 4th year blades.

Table 7: Values from literature for carbon sequestration equations.

Ratio needed	Average	Species	References
%POCburialdeep	0.11	Not specified	(G. Gao et al., 2022; Krause-Jensen & Duarte, 2016)
%POCburialsed	0.04	Not specified	(Canvin et al., 2024; Dolliver & O'Connor, 2022; G. Gao et al., 2022; Krause-Jensen & Duarte, 2016)
%DOC	0.314	Saccharina Japonica	(Y. Gao et al., 2021)
%RDOC	0.214	Saccharina Japonica	(Y. Gao et al., 2021; M. Zhang et al., 2022)
%(POC-RDOC)	0.21	Saccharina Japonica	(Feng et al., 2022; M. Zhang et al., 2023)

harvest dislodgement rates Fieler et al., (2021) in Table 8. For the two other case studies, who harvest much earlier and don't keep the biomass in the water over the winter, the average dislodgement rate included the low value of Fieler et al.,s (2021) results.

Our calculations conclude that the total carbon potentially sequestered from the ORF cultivation units in Funningsfjørður is 3,446 kg C yr⁻¹, which is equivalent to 12,636 kg CO₂e per 127,674 kg of WW harvest on 135,696 m of cultivation rope. The total carbon potentially sequestered from the French seaweed cultivation ALO is 2,456 kg C yr⁻¹, which is equivalent to 9,005 kg CO₂e per 200,000 kg WW harvest on 150 ha. In

Norway, the OF site potentially sequestered 2,560 kg C yr⁻¹, which is equivalent to 9,388 kg CO₂e for their 141,400 kg WW harvest on 36,893 m of growth line, view Tab 9.

ALO in France and OF Norway have an almost similar sequestration potential with more WW kg harvest on the French site. This is due to the lesser carbon content in their seaweed.

Other studies did not include the burial in the deep sea (Pessarrodona et al., 2024), considering it as negligible if the cultivation is in sheltered bays and thus can have reduced exchange with the open ocean (Liu et al., 2019). According to the Aquaculture Overview report on the Faroe Islands the

Table 8: Erosion, exudation (parts of the blade break off) and dislodgement (entire blade breaks off) rates from literature on *S. latissima* in % of NPP.

Location		Species	Erosion and exudation	Dislodgement	Total		Reference
Northern Ireland, sheltered site		<i>S. latissima</i>	43.2%	41%	84.2 %		(Dolliver & O'Connor, 2022)
South-west of England, exposed site		<i>S. latissima</i>	35%	35%	70%		(Canvin et al., 2024)
Mid- and North Norway	blades left in the water longer	<i>S. latissima</i>	Approx. 9.40%	Approx. 40%	49.4 %	N.A.	(Fieler et al., 2021)
	with optimized harvest	<i>S. latissima</i>	N.A.	N.A.	N.A.	8.1%	(Fieler et al., 2021)
Average for ORF					67.9 %		
Average for ALO and FO						54.1 %	

Table 9: Summary of total carbon sequestration potential for all three sites.

	ORF in the Faroe Islands		ALO in France		FO in Norway	
Total harvest	127,674 kg WW harvest on 135,696 m growth line and 17.8 ha		200,000 kg WW harvest on 150 ha		141,400 Kg WW on 36,893 m growth line	
yield	1.51 kg m ⁻¹ ; 0.715 kg m ⁻²		N.A. m ⁻¹ ; 0.133 kg m ⁻²		3.83 kg m ⁻¹ ; N.A m ⁻²	
	Carbon sequestered (kg C yr ⁻¹)	CO ₂ sequestered (kg CO ₂ e)	Carbon sequestered (kg C yr ⁻¹)	CO ₂ sequestered (kg CO ₂ e)	Carbon sequestered (kg C yr ⁻¹)	CO ₂ sequestered (kg CO ₂ e)
Per total harvest	3,446.24	12,636.2	2,456.00	9,005.34	2,560.41	9,388.16
Per t WW	27	99	12	45	19	68
Per kg C harvested	1.06	3.9	0.51	1.86	0.51	1.86
Per m	0.0254	0.09	N.A.	N.A.	0.07	0.25
Per m²	0.0334	0.12	0.0016	0.0060	N.A.	N.A.

water residence time in the Faroe Islands is between 1-2 months (ICES WGFaroesAO, 2023 and references therein), which is comparable to the model from Liu et al. (2019), stating that the sea residence time for the Faroe Islands is between 15 – 30 days (Liu et al., 2019). The French West coast (ALO) as well as the Norway south west coast (OF) have a water residence time of 1-3 years (Liu et al., 2019). According to Pessarrodona et al. (2024), a temperate and polar coastline seaweed cultivation situated close to coastal carbon sinks (other blue carbon habitats, fjords, canyons) or the deep sea (with a narrow continental shelf), will have greater sequestration potential (Pessarrodona et al., 2024 and references therein). The Faroe Islands have a narrow continental shelf and short water residence time compared to the other two sites; we thus only included the deep-sea burial rates for the Faroe Islands case, not for France and Norway.

3. Discussion

3.1 Blade density loss

Our results show a greater blade density loss in the first year than in the following years, which is comparable to results found in other studies stating that fall off occurs in the early stages of cultivation (Canvin et al., 2024; Dolliver & O'Connor, 2022; J. Zhang et al., 2012). In the first year after deployment the seedlings are very fragile and rip off easier, but their density also reduces due to competition for space and light (pers communication with ORF). This trend slows down in the summer months when their rhizoids are stronger and attach better (J. Zhang et al., 2012). However, our results show an increase in blade density in the late summer. A potential explanation could be that the empty spaces on the growth lines are taken by naturally settling seaweed. These blades are again fragile, and blade density drops again in the autumn and winter months with harsh wind and weather conditions (pers communication with ORF).

The calculated POC loss due to dislodgement on the Faroese site is an equivalent of 129 % to the harvested biomass when leaving the blades in the water year-round, so 12 months. The Norwegian study from Fieler et al., (2021), measured a loss rate¹ equivalent to the production rate in August, while in June only 9% of production equivalent was lost. The study furthermore estimates that of the total standing biomass, up to 75% of it can be lost within 3 months and even losses of 100% have been observed later in the season (Fieler et al., 2021). Our calculated biomass loss due to dislodgement on the Faroese site seems plausible, due to the seaweed staying in the water year-round, also over the winter months, in a site with high waves and currents, making it an exposed like environment. However, we expect our results to be a slight overestimation, as this calculation is based on an average blade mass loss with an average loss rate per year, yet more blades are lost during the winter, when they are shorter and have less biomass. Further data collection is needed to quantify the biomass lost monthly and additional data on

erosion and exudation would complement these results and compared to other case studies.

3.2 Carbon sequestration potential

Studies from literature use various methodologies and reach different values for carbon sequestration potential. Canvin et al, (2024) calculate that the assessed seaweed cultivation in Cornwall may sequester 5g CO₂e m⁻², or 1.36 g C m⁻², which is substantially less than ORF (33.4 g C m⁻²) and in line with ALO (1.64 g C m⁻²). With ORF's three-dimensional cultivation structure, the yield and sequestration potential per area is expected to be different than other studies assessing two-dimensional cultivation sites. Furthermore, the Cornwall study is not considering DOC and RDOC, only determining POC loss and the burial from it. Pessarrodona et al. (2024) estimated global sequestration potential from cultivated seaweed to be median 0.35 Tg C yr⁻¹ (range 0.01–3.18 Tg C yr⁻¹) calculated for a global cultivation area of up to 4000km² with a global average yield of 300 – 1400 kgC.m⁻² yr⁻¹ and with 1.5 C Tg POC globally harvested in 2018. This equates to a carbon sequestration potential ratio of 87.5 g C m⁻², 0.0625 - 0.29 g C sequestered for each kg of carbon harvested and sequestered carbon ratio to POC harvested being 0.23. The carbon sequestration potential per area is over 60 times larger than the study from Canvin et al. (2024), over 50 times larger than our results for ALO and OF and twice as much than the value from ORF. It is not meaningful to compare the per area results, as Pessarrodona et al. (2024) results are based on global seaweed production including all kinds of seaweed growing in all kinds of cultivation techniques. When looking at the POC harvested to carbon sequestered ratio, ORF is 4,6 times higher (1.063), and ALO and OF roughly twice as high (0.51). Pessarrodona et al. (2024) explain that values of carbon sequestration potential vary with different carbon contents for each species and burial rates for each coastal condition. Interestingly, in their global model, fixing the carbon content didn't change the outcome of the analysis too much. In fact, ca. 85 % of their calculated carbon sequestered was achieved through the DOC pathway. Their analysis was highly sensitive to the amount of DOC and the fraction that eventually is stored as RDOC, both pathways with large, reported difference in literature. An explanation for our higher values in carbon sequestration potential could thus be our *S. japonica* specific DOC and RDOC rates, which were on the higher end of the global averages from Pessarrodona et al. (2024).

Furthermore, Pessarrodona et al. (2024) chose to use minimum POC loss rates (assuming optimized harvesting time), which offers an additional explanation to the higher values for ORF, leaving the cultivation structures in the water. Fieler et al. (2021) even argued, that an industry with early harvest will have low carbon exports, while four to six times more carbon could be exported (with an potential to be sequestered) when leaving the blades in the water and harvesting late (Fieler et al., 2021). A calculation from Gao et al.

¹ of which 80-90% is due to dislodgement

(2022) states that the ratio of harvested POC to Net Primary Production is 0.294 and furthermore that the ratio of carbon sequestration potential to harvested POC is 0.568, meaning that the equivalent of 57% of the amount of carbon of harvested seaweed would also be sequestered. For each kg C, 568 g C would be sequestered. Our ratios for ALO and OF are similar, with 51%, while ORF is almost double with 106%.

The ratios from Gao et al. (2022) are taken from Krause-Jensen & Duarte (2016), who originally calculated global carbon sequestration potential from NPP of wild seaweed, growing from a two-dimensional surface, while for the case of ORF seaweed grows three-dimensionally. Furthermore, the ratios are for all kinds of seaweed grown globally, and the original authors further state that the ratios of grazing and remineralization per NPP don't add up to a total, as they are based on separate calculations. Similarly, also Song et al. (2023) based their calculations on the Krause-Jensen & Duarte (2016) ratios, concluding a mean carbon sequestration potential of $174 \pm 6 \text{ g.m}^{-2}.\text{yr}^{-1}$ of seven different kinds of macroalgae with a ratio between carbon sequestered and harvested POC of 0.41 (41%). The mean carbon sequestration potential is comparable to the ALO value (1.68 gC.m^{-2}), and the ratio in the vicinity of the ALO and OF value (51%), while again, the ORF values are higher.

These few examples of comparing carbon sequestration potentials and their vastly different methods used, and results obtained, show that this is a fast-advancing field of science with no one solution. The results also need to be explained in their context, as carbon content per species- cultivation technique- and DOC, RDOC and POC percentages used, all alter the results.

3.3 Further considerations

As introduced in the beginning of this chapter, uncertainties on the magnitude and burial rates of the carbon fluxes, and climate driven shifts in this carbon cycling, still need to be further constrained to gain a better understanding of the sequestration potential. Additionally, other important questions remain and should be considered when calculating the carbon sequestration from seaweed cultivation.

3.3.1 Emissions to consider

In this deliverable, we did not take the CO₂ emissions needed to build and maintain a cultivation site, the infrastructure and the CO₂ emitted during harvest and processing, into account (Canvin et al., 2024; Pessarrodona et al., 2024). Deliverable 9.7 within task 5 of WP 9 models the LCA of three seaweed flagship products, and thus also accounts for the emissions of seaweed production. Pessarrodona et al. (2024) estimates that currently seaweed farming is rather a net global emitter with $-0.77 \text{ Tg C yr}^{-1}$ (range -5.65 – $1.80 \text{ Tg C yr}^{-1}$) offsetting the sequestered carbon with the farming emissions (median $1.14 \text{ Tg C yr}^{-1}$; range 0.03 – $6.37 \text{ Tg C yr}^{-1}$) (Pessarrodona et al., 2024).

Roth et al. (2023) point out the understudied air-sea gas exchange and therefore often neglected CH₄ emissions from coastal ecosystems in sequestration potential calculations. The authors calculate that potential CO₂-equivalent CH₄ emissions

from seaweed POC deposit in the sediment, could even offset 28% of the carbon sequestered (Roth et al., 2023).

To a smaller magnitude, theoretically also the CO₂ released from the additional calcifying organisms, such as bryozoans, inhabiting the potential additionally habitat from seaweed, should also be considered in the net sequestration potential (Bach et al., 2021; Pestana, 1985).

3.3.2 Air-sea exchange

When aiming for carbon sequestration accounting or carbon trading, it is atmospheric carbon that needs to be stored away and accounted for (Bach et al., 2021). Hurd et al. (2023) state that for marine carbon sequestration potential from seaweed growth, the air-sea CO₂ equilibrium needs to be considered. Seaweeds absorb CO₂ from seawater rather than the atmosphere, resulting in a CO₂ deficit in the waterbody, which is not immediately equilibrated but may take weeks to years to equilibrate, depending on the movement and mixing of the waterbody. This quantification of the CO₂ flux from the atmosphere to the ocean increases the complexity considerably and may even question the actual removal of CO₂ from the atmosphere (Bach et al., 2021; Hurd et al., 2023). Hurd et al. (2022) attempted to gain an overview over this complexity with an Carbon Forensic Accounting framework (Hurd et al., 2022), and Bach et al. (2023) proposed a marine Carbon Dioxide Removal (mCDR) risk assessment to not lose the potential carbon sequestration out of sight (Bach et al., 2023). This risk assessment works with incomplete near-term equilibration, essentially assessing a likelihood for several years of sequestration.

3.3.3 Replacement ecosystems

When upscaling or expanding seaweed production, the availability of nutrients may become an issue. Seaweed may not immediately be limited in growth, but it may compete for these nutrients with other primary producers such as photosynthesizing phytoplankton (Bach et al., 2021). When it comes to carbon sequestration potential, due to the reallocation of nutrients from phytoplankton to seaweed, theoretically the carbon sequestration potential from the plankton would need to be subtracted from the seaweed carbon sequestration potential (Bach et al., 2021). A question is even if the phytoplankton POC formed and the natural carbon sequestration potential thereof would even be more efficient than carbon sequestration potential from seaweed (Bach et al., 2021).

3.3.4 Challenges for carbon accounting in financial frameworks

Besides these uncertainties in quantification of carbon sequestered, there are additional challenges with requirements, such as additionality and governability, when carbon credits will be sold from seaweed cultivation (Pessarrodona et al., 2023). Further details will be discussed in D9.4.

3.3.5 Other sequestration pathways from seaweed production

The marine carbon sequestration path is just one of the potential ways carbon from seaweed can be sequestered. As presented in D9.2 and in the introduction to this chapter, it is

possible to sequester carbon via the terrestrial pathway, for example with biochar from seaweed, or placing seaweed in construction materials, to reduce emissions, resulting when seaweed products replace products with a higher carbon footprint, or to avoid carbon emission, for example when seaweed products are replacing other more emission intensive products. A study from British Columbia, Canada modeling the carbon removal pathways concluded, that the marine sequestration was rather inefficient, compared to seaweed based products (Bullen et al., 2024). In our results, we have not yet included the sequestration, avoidance or reduction potential from the seaweed products and can thus expect, that a substantially bigger part of climate benefit potential is still to come with further analysis on the products (Bullen et al., 2024; Canvin et al., 2024; Pessarrodona et al., 2023; Spillias et al., 2023; Thomas et al., 2024; van den Burg et al., 2023). This argumentation is in line with other researchers suggesting that marine carbon sequestration potential from seaweed could be seen as a nice co-benefit, but not the driver behind the cultivation. Alternative products, and other co-benefits of seaweed cultivation expansion, such as job creation and potential habitat creation, should rather be focused on (Canvin et al., 2024).

CONCLUSION

Seaweed aquaculture has been praised for providing several ecosystem services. Yet, quantitative data on the delivery of these ecosystem services is scarce. This report assessed three ecosystem services for three seaweed aquaculture sites: Ocean Rainforest (ORF) in the Faroe Islands, Algolesko (ALO) in France and Ocean Forest (OF) in Norway. The goal was to quantitatively report on the nutrient bioextraction potential for each site and what impact ORF's cultivation activity has on biodiversity, and carbon sequestration potential for each site.

The potential for nutrient bioextraction by *Saccharina latissima* ranged from 1.2 kg N t⁻¹ WW harvested (OF) to 1.9 kg N t⁻¹ WW harvested (ALO) to 2.5 kg N t⁻¹ WW harvested (ORF). The nutrient bioextraction is highly dependent on yield and nitrogen content of the dry weight seaweed. Our results are comparable to prior assessments (Aquavita project also on ORF site), but often less than literature results. A key factor identified for this is the yield, oftentimes stemming from studies analyzing extraction potentials from experimental sites using maximum potential yield, rather than real life operations with average yields. Furthermore, yields are expressed per area, which are highly cultivation structure dependent, and not per kg of harvest.

Our biodiversity assessment for the ORF site in the Faroe Islands showed that the cultivation lines harbored many prey species, such as bivalves and crustaceans, which are also found to be food for commercially important fish species e.g. cod. It was found that the exposure of the site did not influence biodiversity development and that abundance and richness in species increased both from April to July and in the age of lines from first to third. With ORF's cultivation structure staying in the water year-round, the impact of creating habitat and

providing food can be concluded to be greater than other seasonal cultivation systems, providing a three-year habitat at the holdfasts and allowing for biodiversity to develop there and continuous for mobile species.

The carbon sequestration potential for the ORF site is higher than the other two sites, with potentially 27 kg C or 99 kg CO₂e per t WW harvested compared to 12 kg C or 45 kg CO₂e per t WW harvested (ALO) and 18 C or 66 kg CO₂e per t WW harvested (OF). This is due to ORF leaving their lines in the water over the winter, which leads to additional particulate organic carbon (POC) loss, and thus higher burial potential. This loss potential was calculated to be as high or even higher (>129%) than the harvested biomass. In order to assess the net sequestration of the cultivation, the emissions of the production and processing have to be added, which will only be done in a later deliverable. Other open questions remain, for instance the air-sea exchange, which is important for carbon accounting, or if other carbon sequestering organisms, such as phytoplankton, have been outcompeted over nutrients.

Our findings reduce the knowledge gaps related to the very complex assessment of biodiversity and carbon sequestration potential by providing data on a cultivation technique, which differs from the most common one seasonal cultivation systems. This cultivation technique leaves the structure and partially harvested seaweed in the sea, allowing a comparison across cultivation strategies.

Our conclusion is that partially harvested and multiple seasonal cultivation systems are more beneficial in terms of biodiversity and carbon sequestration potential. To improve the potential of the activity to be beneficial for the ecosystem, proportion of the cultivation lines could be left in the water to support biodiversity. Furthermore, the high loss of biomass due to dislodgement is also an argument for carbon climate change mitigating activity, as this sequesters more carbon, although this is not desirable from an economic point of view, as more biomass is lost per seeded line.

These findings presented here are food for thought on the regenerative nature of seaweed cultivation and provide input into how the potential of seaweed cultivation's ecosystem services can be increased.

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DATA SETS

Carbon sequestration data for ORF (2024)

Biodiversity data for ORF (2024)