Synthesis of ReMEDIES actions and assessments of seagrass beds within the SACs of Plymouth Sound & Estuaries, and Fal & Helford

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Report details

Author(s)

Luke Edwyn Marsh, Marine Ecology Specialist, Natural England
luke.marsh@naturalengland.org.uk

Natural England Project Manager

Fiona Tibbitt, Senior Project Manager- EU LIFE Recreation ReMEDIES, Natural England

fiona.tibbitt@naturalengland.org.uk

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Contents

Executive Summary

The EU LIFE Recreation ReMEDIES: ‘Reducing and Mitigating Erosion and Disturbance Impacts affecting the Seabed’ project (LIFE 18 NAT/UK/000039), ran from July 2019 to October 2024 with aims to improve the condition of seagrass beds and maerl in five Special Areas of Conservation (SACs) between Essex and Isles of Scilly. This was achieved by trialling restoration, management options, and reducing recreational pressures. Promoting awareness, communicating, and inspiring better care of sensitive seabed habitats was key. Natural England (lead partner) worked with the Marine Conservation Society (MCS), the Ocean Conservation Trust (OCT), Plymouth City Council (PCC)/Tamar Estuaries Consultative Forum (TECF), and the Royal Yachting Association (RYA). The project is financially supported by LIFE, a financial instrument of the European Commission.

Additionally, as a nature and environmental asset providing ecosystem services, the UK Government in its 25-Year Environmental Plan for ‘securing clean, healthy, productive, and biologically diverse seas and oceans’, has seagrass beds as a priority habitat under section 41 of NERC Act 2006, with a legal duty to conserve and enhance such habitats.

The initial aim of this report was to follow Action D1’ within project on Monitoring and Evaluation; whereby the report will consider, and assess, the outcomes of the ReMEDIES project’s restoration achievements for the protected seagrass beds within the SACs of Plymouth Sound & Estuaries, and Fal & Helford. Using the best available evidence from baselines of seagrass surveys at the start of the project in 2018, to the most complete and recent data towards the end of the project in 2024, the report will then make a before and after comparison of the seagrass within these two SACs following set criteria linked to quality measurements of seagrass beds. The overall aim of the project was, by using this point at time (none-temporal) comparison, identity links between restoration efforts and the periodic changes in the seagrass beds to indicate restoration efforts that could be both valuable for seagrass conservation and bring the best results in future seagrass restoration.

Data was provided by Natural England (NE) dive surveys and Environment Agency (EA) (boat-based) drop-down video surveys, and the report models the differences in seagrass bed coverage; estimated extent including estimated extent by coverage; possible infection burden by ranked browning scale; and longest leaf-length as a proxy for biomass. By quantification of these changes for the two sites where ReMEDIES project and its partners have implemented a range of management interventions, the report provides a grounding for the project achievements and a platform towards identifying lessons learned and similar project improvements for the future. Study design and error analysis are discussed, alongside interpretation of conclusions and implications from results.

Certain changes in the seagrass bed were identified, possibly from the modification of pressures, but natural expansion and contraction of seagrass beds may also have contributed. Seagrass extent was shown to have increased in Fal & Helford, with Helford

Passage and the St. Mawes areas showing measurable changes, and Plymouth Sound & Estuaries also showed slight increases at Cawsand Bay and Firestone Bay. Though Cawsand Bay was not definitive, the links between seagrass extent improvements at Cawsand Bay and Helford Passage, where the use of a combination of Advanced Mooring Systems (AMS) and Voluntary No-Anchor Zones (VNAZ), were noted. Challenges during the project led to variation in the time of year of monitoring and limited direct evidence at the active restoration site. This was particularly evident with the assessment of infection burden data for the two SACs. The results indicated low levels of infection burden for most beds, but a consistent volume across surveys, and certain beds like Drakes Island in Plymouth Sound & Estuaries being specifically problematic. However, probable temporal changes could not be ignored but could not be assessed. As a result, whilst certain beds have seen positive changes, it is not fully clear with information incomplete to identify the drivers for this, and the success of active restoration is not considered within the scope of this report due to restoration timelines and evidence gaps.

The report concludes that further annual long-term monitoring of active restoration of subtidal seagrass beds is required, including ground-truthing that extends over a period of 5-10 years, before any conclusions can be drawn with confidence towards singling out the most the most effective active restoration technique for success. The report highlights the importance of pressure identification, and modification, for improving seagrass condition and makes recommendations towards conservation management of seagrass for the future.

Acknowledgements

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Working in Partnership with:

Introduction

The functional benefits of common seagrass

Seagrasses are the only angiosperms (flowering plants) which grow in marine environments. They are monocotyledons (commonly called monocots), herbaceous (soft stemmed) annuals and perennials, with sexual reproduction by seed, but also asexual cloning from their projecting rhizomes which can also increase distribution and bed size. For reproducing sexually, seagrasses follow two strategies, 1; the dispersal of seeds on the substrate surface by currents or other hydrodynamic processes, and/or 2; the formation of seedbanks by depositing of dormant seeds by sedimentation. Therefore, substrate type and localised hydrodynamic processes characterise seagrass distribution. Additionally, being plants, limiting factors to growth for seagrasses are light, the correct nutrient availability, and nutrient balance; so, depth and localised water quality conditions are also factors that can influence survival and control seagrass bed distribution (Greve & Binzer, 2004).

Common seagrass (*Zostera marina*), which is also referred to as eelgrass, is a Northern Hemisphere seagrass species, widespread in the northern Atlantic and north-eastern Pacific. In the UK, common seagrass beds occur mostly in the subtidal zone of shallow, sheltered, coastal, and estuarine environments, with a wide but patchy distribution across southern and eastern coasts of England. The plant forms a biogenic habitat. This extends habitat area and complexity in marine sediments as it forms a three-dimensional space, creating benthic structure that facilitates wider biological and ecological processes such as refuge for marine life, and foraging habitat including predator–prey relationships. It is a key habitat for important marine vertebrate species, such as long-snouted seahorse (*Hippocampus guttulatus*) and European eel (*Anguilla anguilla*), as well as invertebrate species like stalked jellyfish (*Calvadosia* spp.). It also provides important habitat for commercial fish species, such as pollack (*Pollachius pollachius*), European seabass (*Dicentrarchus labrax*), and common cuttlefish (*Sepia officinalis*). The plants support a microbiome for grazers such as common periwinkle (*Littorina littorea*), and the plants themselves are a food source for herbivorous birds such as brent geese (*Branta bernicla*).

Figure 1: Seagrass with nudibranch.
Image © Natural England.

The common seagrass's biological structure above and below the surface help to stabilise the sediment and influence local environmental conditions. The plant creates hydrodynamic drag which dissipates wave energy. This process reduces suspension of particulate organic matter and inorganic sediment in the water column, which then settles and is bound into the seagrass meadow (Lefebvre and others, 2010). This creates a positive feedback loop that reduces turbidity increasing the light available to the benthos and encouraging primary productivity and growth (Carr and others, 2010), further stabilising the bed. Through photosynthesis, seagrass will transfer dissolved CO_2 from seawater to plant matter, storing and sequestering this CO_2 (aq) by eventual burial in substrates. Therefore, common seagrass functions as a blue carbon sink. Additionally, common seagrass beds will also store carbon from terrestrial as well as marine sources, by trapping carbon matter transported by land-water run-off. From these processes, seagrass beds in the UK are estimated to have stored up to 11.5 mega-tonnes of carbon historically (Gregg and others, 2021) and are a future 'nature-based solution' for absorbing carbon.

The goals of Recreation ReMEDIES and the conservation techniques applied.

EU LIFE Recreation ReMEDIES was focusing on how sensitive seabed habitats are impacted by recreational activities. The project focussed on seagrass beds and maerl beds. Together, the project partnership aimed to:

- Reduce recreational pressures on sensitive habitats.
- Restore and protect sensitive habitats.
- Promote awareness of these habitats and their importance

ReMEDIES was funded by the EU LIFE programme and led by Natural England in partnership with The Royal Yachting Association, Marine Conservation Society, Ocean Conservation Trust and Plymouth City Council/Tamar Estuaries Consultative Forum. See our Partners and Funders page for more information. This report summarises the work on subtidal seagrass beds during the scope of the project.

Natural England monitoring shows that seagrass beds have declined in extent and health nationally (Natural England, 2023), with research indicating common seagrass is increasingly subjected to both natural and anthropogenic stresses (Lee and others, 2004: d'Avack and others, 2014: Jackson and others, 2016: Green and others 2021).

This has been attributed in part to elevated levels of anthropogenic activity pressures in addition to seagrass parasites that cause 'wasting disease' such as *Labyrinthula*. As a result, some beds were in 'unfavourable condition' in several protected sites, such as The Solent when assessed in 2018 (Natural England, 2018[a]: Natural England, 2018[b]), The Isles of Scilly in 2020 (Natural England, 2020[c]), and Plymouth Sound and Estuaries in 2021 (Natural England, 2021[d]: Natural England, 2021[e]). The ReMEDIES Project sought to change this condition and move the beds towards a more 'favourable condition' status. Five sites make up the project suite within ReMEDIES, with two being targeted for active interventional seagrass restoration. Each of the five sites have their own combination of restoration techniques under a variety of indirect and broader educational actions, and proactive, targeted intervention measures.

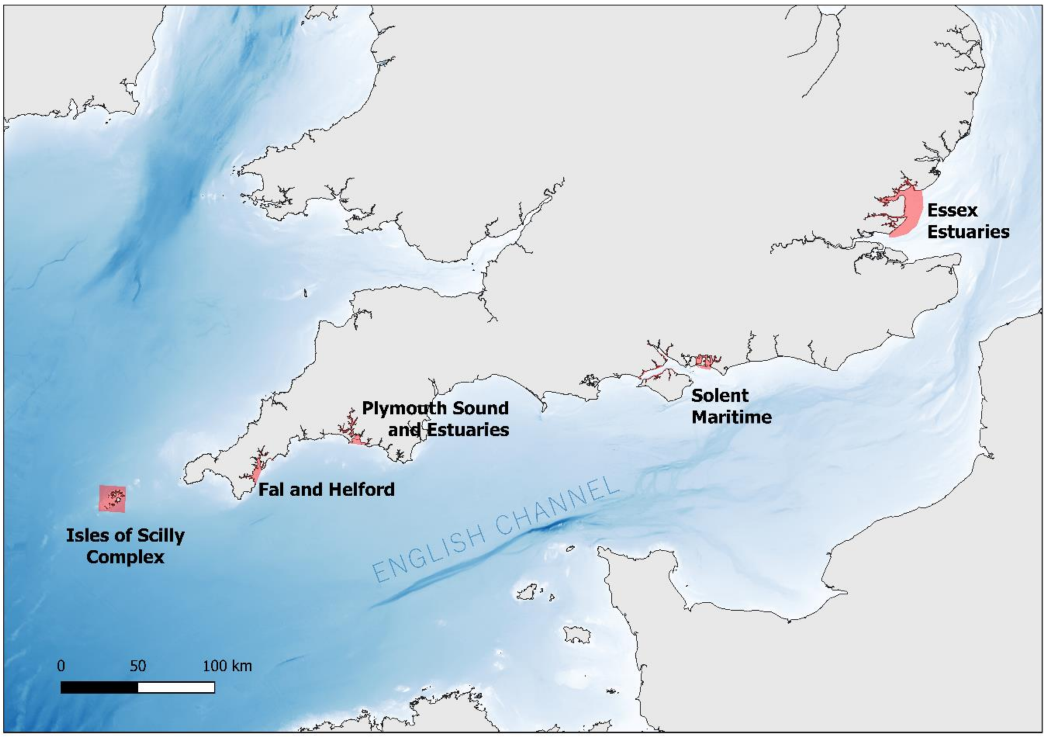

Figure 2: The distribution of ReMEDIES Sites across the south-west of Britain.

The proactive measures were applied in Marine Management Organisation (MMO) licenced restoration areas of seabed and involved the addition of seagrass material in either seed or plant form into the present substrate. This was to trial methods of active restoration to play a key role in moving sites toward 'favourable condition', as planting common seagrass in these locations could increase their extent and help to rebuild connectivity between seagrass beds improving their ability for sexual reproduction. These created seagrass areas would then be managed to encourage the new seagrass beds to be established. Additionally, a combination of Advanced Mooring Systems (AMS), Voluntary No-Anchor Zones (VNAZ), and swim markers were used to help change behaviour and raise awareness. VNAZs used marker buoys to notify boat users of risk of anchoring within seagrass.

There were targeted public engagement which sought to educate the recreational boating community and industry about their impacts to common seagrass habitats. Lastly, AMS were used to prevent and reduce direct mooring-based impacts to current beds. AMS technology reduces seabed damage that can occur from the anchoring and mooring of recreational boats. These techniques for restoration were carried out in ReMEDIES sites where seagrass beds had been lost, fragmented, or degraded by historic harm from either wasting disease, or anthropogenic activity such as mooring, anchoring, trampling, and certain demersal fishing practices such as shellfish potting, bottom-set gill and tangle nets within the SACs. A wider methodology for applied techniques in restoration can be found in the Seagrass Restoration Handbook (Gamble and others, 2021).

All the five ReMEDIES sites are designated as SACs under Directive 92/43/EEC on the Conservation of Natural Habitats and of Wild Fauna and Flora, and the Conservation of Habitats and Species Regulations 2017 (as amended) in England and Wales and included as an Annex 1 habitat within the EU Habitat Directive. As one of the UK's Statutory Nature Conservation Bodies (SNCBs), Natural England are responsible for assessing and reporting on the condition of protected habitats and wildlife populations in Marine Protected Areas (MPAs) every six-years. Natural England's Marine Condition Assessments deliver this necessary statutory information on the condition of protected features in the SACs. The Condition Assessment process is applied to all fully marine features protected with marine components generating the required evidence to apply a condition category. As a sub-feature of 'Mudflats and sandflats not covered by seawater at low tide' and 'Sandbanks which are slightly covered by seawater all of the time', seagrass is reported on by both Condition Assessments and interim reports, such as this.

As part of the ReMEDIES Project, a series of Actions were identified as both the milestones and framework to the project. In regard to one of these actions, 'Action D1', which formulates the monitoring and evaluation aspect of the project, the perimeters were set for this report, constructing by the consolidation of the monitoring plan and collation of baseline data, a report-based investigation to help determine improvements in SAC condition and evaluate the success of the various restoration and management techniques. Whilst monitoring was impacted by pandemic restrictions from Covid-19, from the use of best available evidence and analysis of the monitoring within the scope of the ReMEDIES project timeline, this report will sit within the portfolio of ReMEDIES reports and identify the outcomes of the ReMEDIES operations and demonstrate the challenges for seagrass habitat management in the future. Owing to limitations in the confidence in the available data at some locations, additional best available evidence, evidence standards, and technical information should be used alongside other reports and data assemblages, for the SAC sites assessed to inform future management and support decision making.

Overview of the ReMEDIES Sites

Monitoring Site: Isles of Scilly Complex SAC

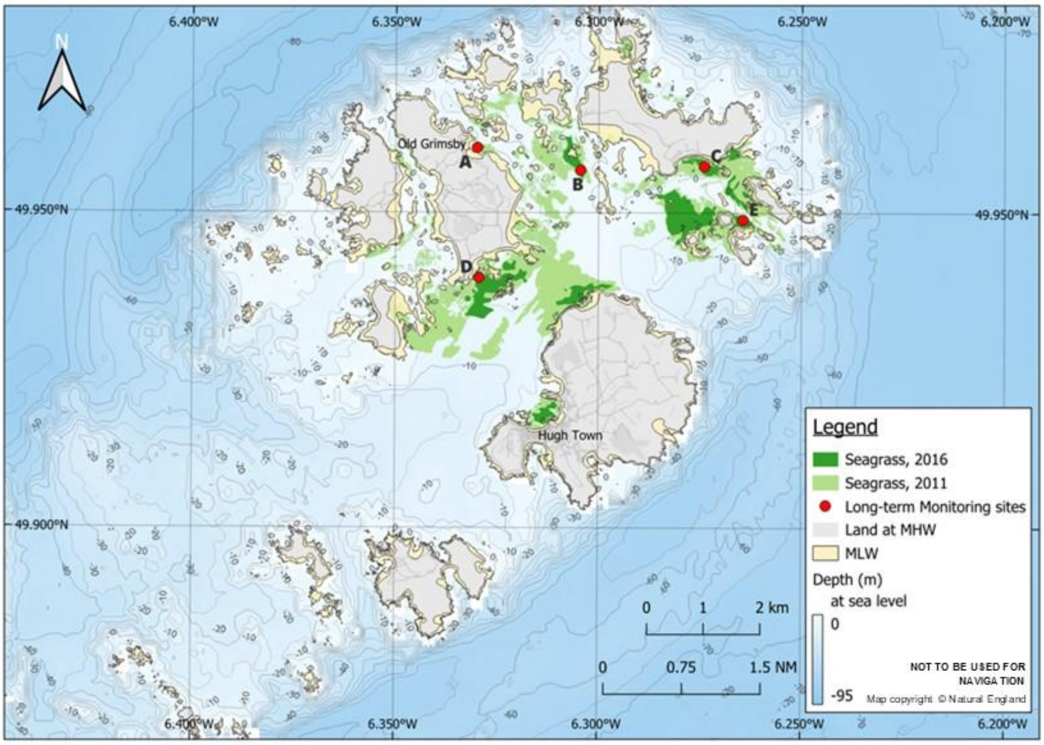

Figure 3: The sites where long-term monitoring of subtidal seagrass has taken place in the Isles of Scilly Complex Special Area of Conservation.
Map shows distribution of seagrass present, and points show long-term monitoring sites in partnership with Project Seagrass of: (A) Old Grimsby Harbour, (B) West Broad Ledges, (C) Higher Town Bay, (D) Broad Ledges Tresco, and (E) Little Arthur.

The Isles of Scilly are a granite archipelago south-west of the British mainland, with around 140 to 200 islands encompassing a rich diversity of marine habitats and species. Seagrass is listed as a sub-feature of the Isles of Scilly Complex SAC, and as it grows extensively and almost exclusively sub-tidally within the island grouping as a natural monoculture. Within the SAC, long-term monitoring has been achieved at five monitoring sites since 1996, and the SAC has become one of the longest and most consistently monitored seagrass areas in the UK. Seagrass in the most recent Condition Assessment in 2020 was recorded as 'unfavourable'/'declining', with recreational activities/boating pressure and Climate Change considered major drivers for this change, but more recent studies have suggested the monitoring areas have mostly stabilised – though at the Higher Town Bay site seagrass shoot density has declined and Old Grimsby Harbour has seen seagrass extent reduction since monitoring began in 1996 (Bull & Kenyon, 2023).

Engagement and Recovery Site: Essex Estuaries SAC

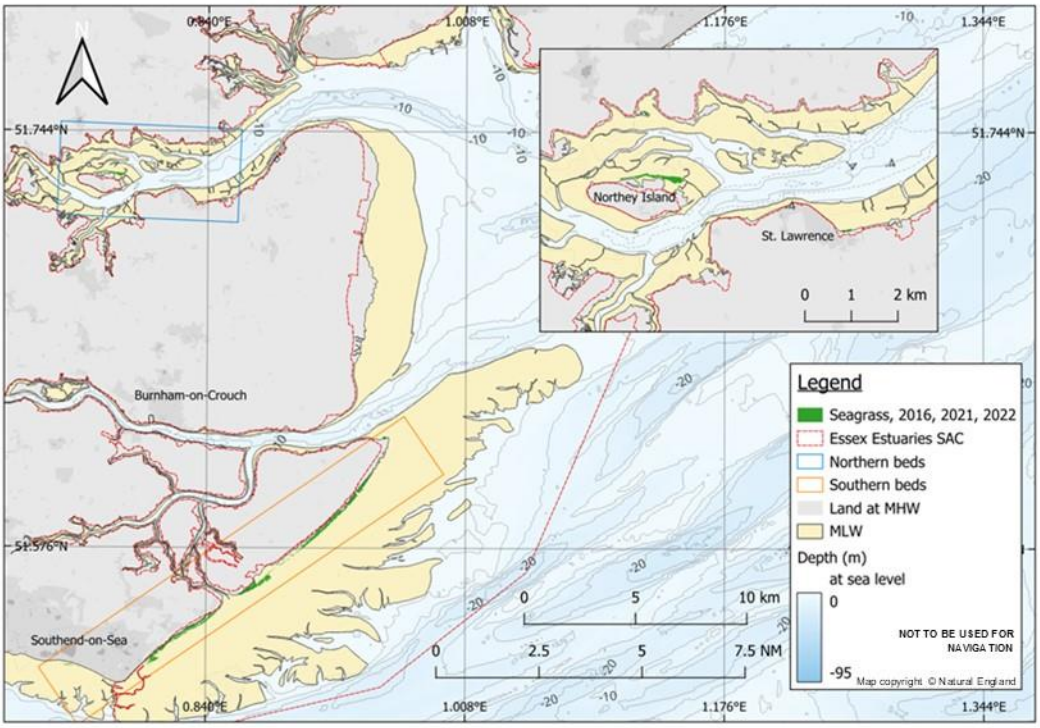

Figure 4: Intertidal and subtidal seagrass distribution in Essex Estuaries SAC.
For reference, the location of seagrass around Northey Island in the northern beds (blue box) on the Blackwater River is show in inset.

As the second largest estuarine site on the east coast of England, the Essex Estuaries SAC is an example of a coastal plain estuary system. Supporting a range of estuarine and marine communities, it provides habitats on sediments ranging from the finer estuarine muds and muddy sands to the coarser sands and gravels. Recorded as 'unfavourable'/'declining' in 2022, with trampling (Howard-Williams, 2022), and predation from birds (Unsworth and others, 2021), as pressures for seagrass in the SAC.

The Essex Estuaries SAC is part of the ReMEDIES project as it contains seven Sites of Special Scientific Interest (SSSIs), and 15 other protected areas, including five Special Protected Areas focusing on the protection of internationally important bird species.

Active Restoration Site: The Solent Maritime SAC

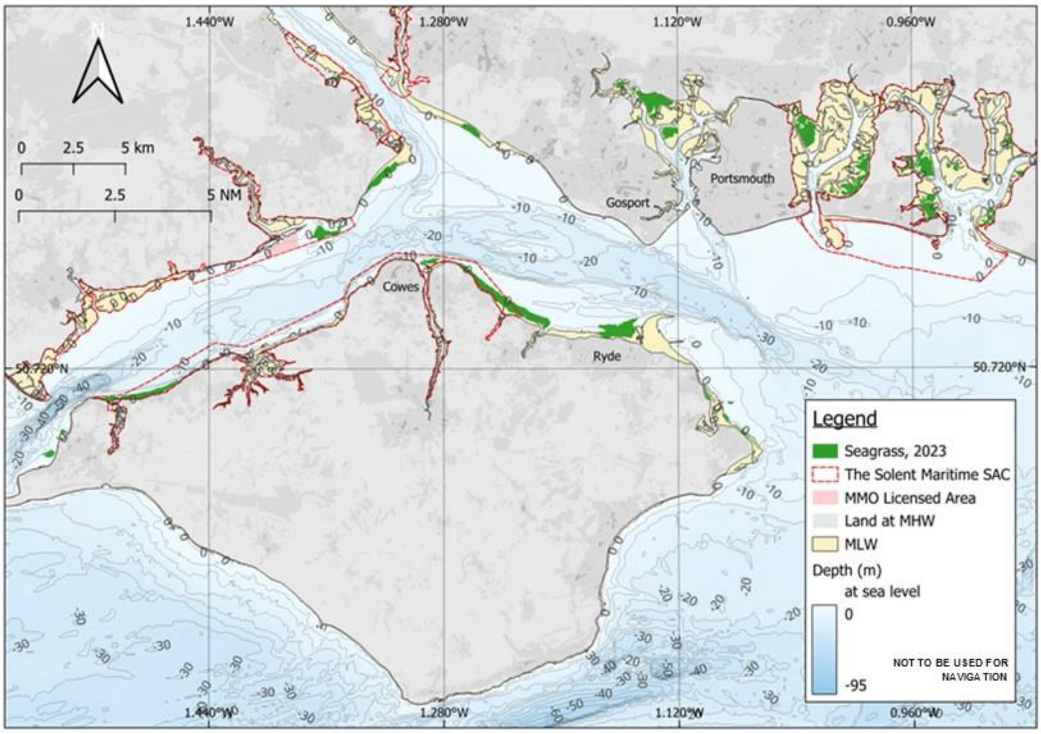

Figure 5:Seagrass distribution in the Solent Maritime SAC, with the MMO licensed area for restoration.

The Solent Maritime SAC incorporates inlets which have uncommon tidal regimes of double tides and prolonged periods of tidal stand when at high and low. The Solent therefore provides equally distinctive maritime habitats. In 2019, Natural England's Condition Assessment for seagrass recorded it as 'unfavourable'/'unknown'. Common seagrass, though extensive in certain areas, is under pressure from a variety of anthropogenic activities including commercial fishing, recreational boating and boating-based activities, navel shipping, and water sport activities.

Outside of the AMS at Cowes on the Medina River, Yarmouth to the west has four AMS, and VNAZ also has been installed in Osborne Bay, all to help manage anthropogenic pressure on the seagrass beds. Ocean Conservation Trust (OCT) have two areas using two methods for restoration. By 2024, from a combination of seed bags and seed injection as a total between Plymouth and Solent; The OCT have planted an area covering 8 Ha with common seagrass seed and mats – and though future monitoring is required - if successful this has the potential to develop into full seagrass bed.

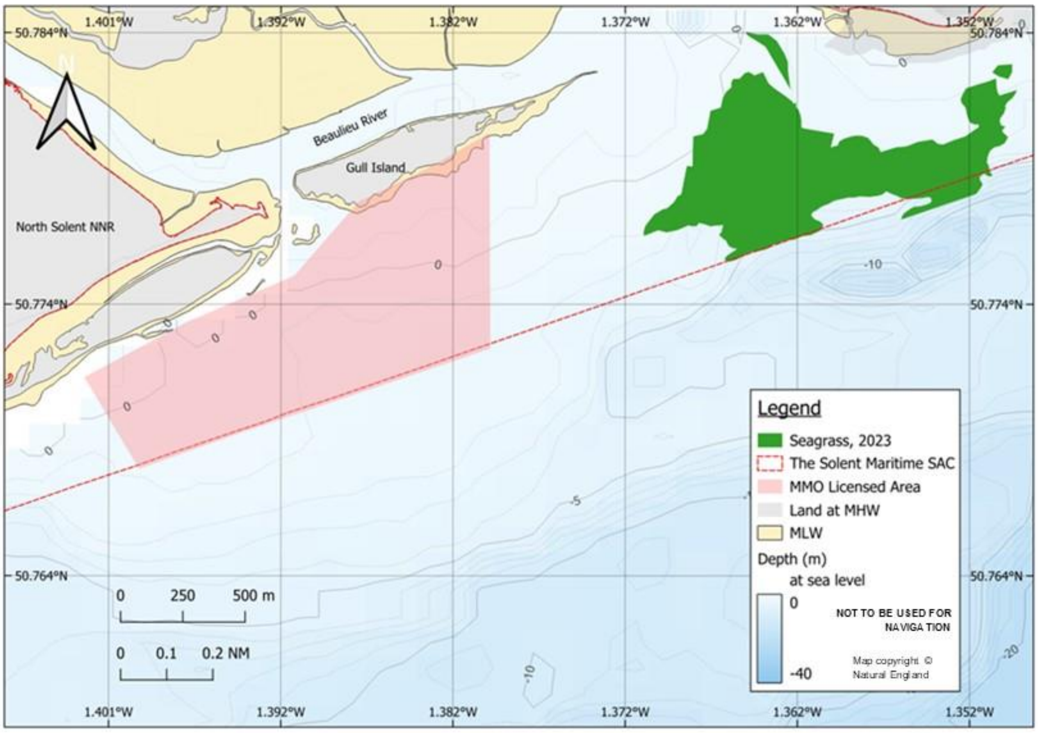

Figure 6: Location of ReMEDIES Restoration Site used by OCT, west of the mouth of the Beaulieu River.

Figure 7: Practical restoration process in the Solent.
Images show (a) seed bags being brought onto the vessel, (b) being deployed down the tubes to the seabed, and (c) in transit showing the size of the bags and the volume of bags released. Image © OCT

The Two Main Sites for Review

Recovery and Pressure Reduction Site: Fal and Helford SAC

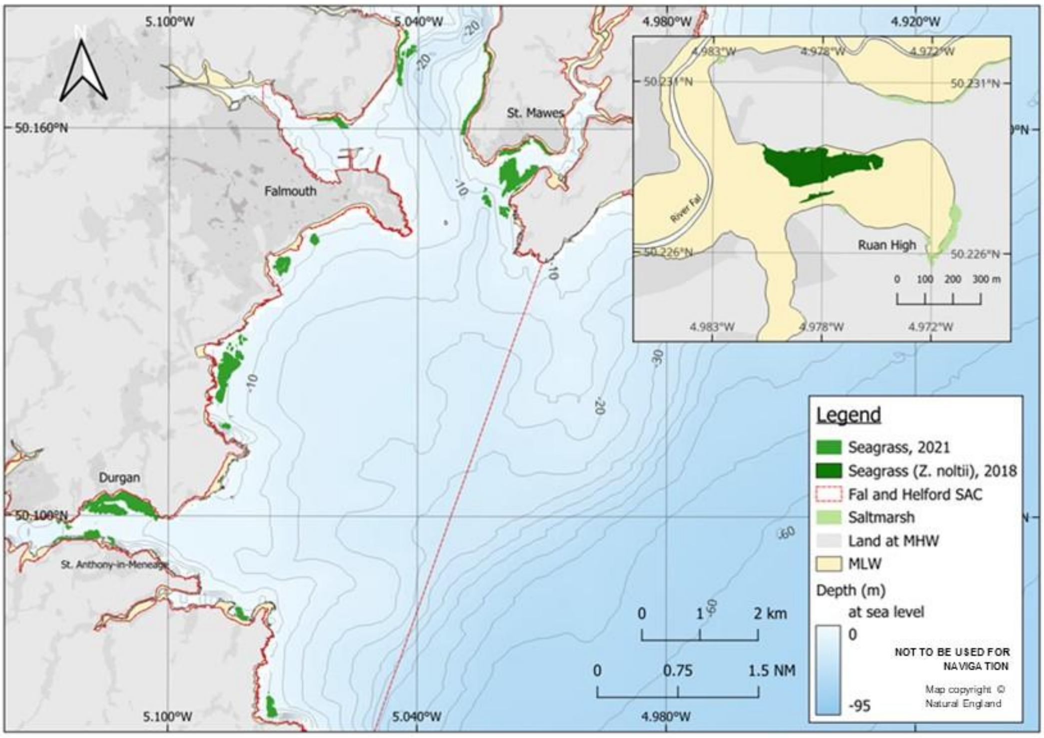

Figure 8: Seagrass distribution in the Fal and Helford SAC.
Area where intertidal dwarf seagrass (*Zostera noltii*) is located higher within the Carrick Roads is provided for reference.

The Fal and Helford SAC is complex system of rias, including some of the deepest natural channels in Europe. As a result, it is rich in marine biodiversity, supporting many marine biotopes and habitats, with associated marine flora and fauna. The Fal and Helford SAC is a 64 km² site, outlined by a line that runs between Zone Point in the northeast of the site, and Manacle Point in the southwest. The qualifying features are the associated habitats:

- Sandbanks slightly covered by sea water all the time.
- Mudflats and sandflats not covered by seawater at low tide.
- Large shallow inlets and bays
- Atlantic salt meadows (*Glauco-Puccinellietalia maritimae*) which colonises soft intertidal sediments of mud and sand.
- Estuaries
- Reefs

The site also supports one of Europe's most threatened endemic vascular plants, shore dock (*Rumex rupestris*), growing on the rocky and sandy raised beaches within the SAC. Seagrass grows in both the intertidal and subtidal areas within the Fal and Helford SAC (see figure 9).

In 2018, the condition of common seagrass in the Fal and Helford SAC was assessed as ‘unfavourable’/’no change’, with large beds found at Durgan and St. Mawes under pressure from boating.

Initially, mitigating impacts to common seagrass at Durgan, in the Helford Estuary Marine Conservation Zone (MCZ), was achieved by the deployment of a VNAZ. In 2020, Natural England conducted baseline surveys for recreational boating to observe the ongoing pressures the seagrass faces in Falmouth and Helford, suggesting how recovery could be supported. This gave a baseline for activities, and pressures that would impact common seagrass and adjoining sessile benthic species. More recently, Falmouth Harbour Commissioners have added a small number of AMS to their moorings and are also monitoring their performance towards conservation of common seagrass within the SAC (see, Spooner, 2023).

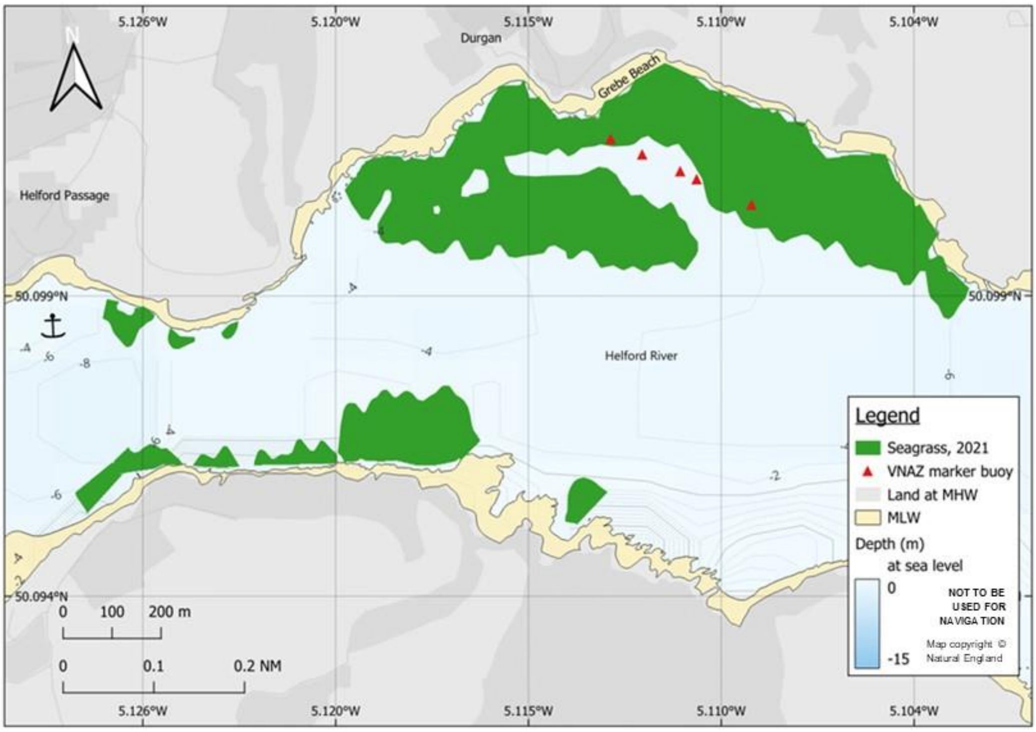

Figure 9: Seagrass distribution in the Helford River, Fal & Helford SAC.
Anchorage at Helford Passage with locations of VNAS locations at Durgan shown for reference.

Surveys carried out under the ReMEDIES project in 2020 and 2021 demonstrated that some residual pressures from anchoring in the seagrass were present despite areas where ‘no anchor zones’ were in place. However, most of these pressures were noticeably reduced (Day & Hayward-Smith, 2021: Hayward-Smith & Dallman 2022).

Alongside the seagrass, another sub-feature of the SAC is the slow growing, free-living coralline red algae, collectively known as Maerl (Phylum Rhodophyta) which may form expansive areas over the seafloor known as maerl beds. The two dominant maerl species in the Fal and Helford SAC are *Phymatolithon calcareum* and *Lithothamnion coralloides* with records of the third species *Lithothamnion glaciale*. Within the SAC, areas of dense maerl 'live and dead' have been identified throughout the lower Fal Estuary, Falmouth Bay and within the tidal Helford River. These maerl beds support a complex community of association species, including burrowing infauna and interstitial invertebrates including suspension feeding polychaetes and echinoderms. This is the only ReMEDIES site which includes maerl beds, protected as part of Annex I Sandbanks.

Figure 10: Common cuttlefish (*Sepia officinalis*) on maerl.
Image © Fiona Crouch.

Although maerl propagates mainly by fragmentation, its recovery after removal of a bed, fragmentation, or high-levels of mortality is low (Hall-Spencer & Moore, 2000: Wilson and others, 2003). Maerl can have ecological overlaps with seagrass, both in its habitat requirements in the marine environment, but also shares some of the pressures facing it.

Active Restoration Site: Plymouth Sound and Estuaries SAC

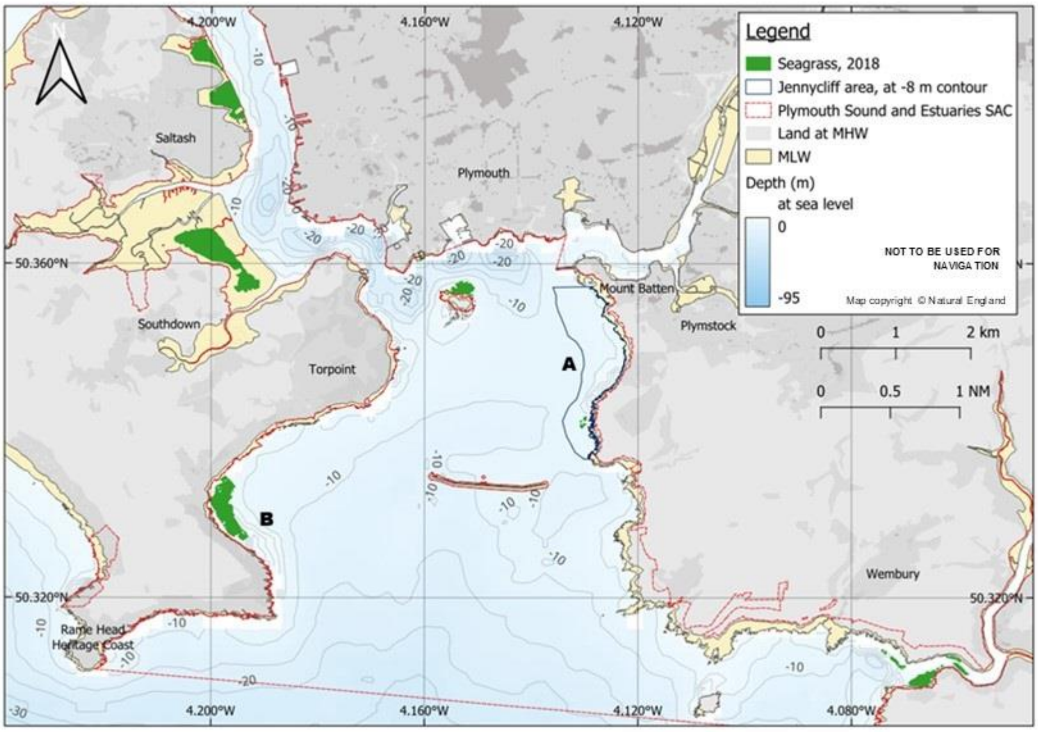

Figure 11: Seagrass distribution in Plymouth Sound and Estuaries Special Area of Conservation.
Areas of ReMEDIES restoration, VNAZ and AMS. Restoration Site of Jennycliff Bay (A) and Cawsand (B) on the southern border of the SAC are shown for reference.

Located on the south coast of England, Plymouth Sound and its associated tributaries comprise a complex mix of marine inlets with rich communities, some with unusual features representative of ria systems. The SAC designation habitat features are:

- Sandbanks slightly covered by sea water all the time
- Estuaries
- Large shallow inlets and bays
- Reefs
- Atlantic salt meadows
- The site also supports shore dock.

A further present Annex II species feature, Allis shad (*Alosa alosa*) has spawning populations in the River Tamar, the only known Allis shad spawning site in the UK. Common Seagrass beds are widely distributed within the SAC, with the largest bed at Cawsand Bay, whilst there are smaller beds to the north of Drake's Island, Jennycliff Bay, and the mouth of the Yealm Estuary including dwarf seagrass in the Tamar and St. John's Lake area (Figure 11). The most recent Condition Assessment in 2021 recorded subtidal seagrass beds as 'unfavourable'/'declining', with Jennycliff Bay having 0.7 Ha of common seagrass in 2018.

After preliminary Habitat Suitability Modelling to identify the suitability of areas for restoration (see, Early and others, 2022), after obtaining permissions and licensing from the MMO, the OCT used a combination of three main active methods of restoration at

Jennycliff, fully articulated in '*Seagrass Cultivation and Restoration'* by Newman and others (2024). OCT deployed up to 2023 seed bags (at 20,000 seeds per Ha), 300 plant pillows/mats (600 per Ha), and seed injection (at 10,000 activations per Ha) within the restoration zone (see figure 13), totalling 5.5 Ha of seagrass area at Jennycliff Bay.

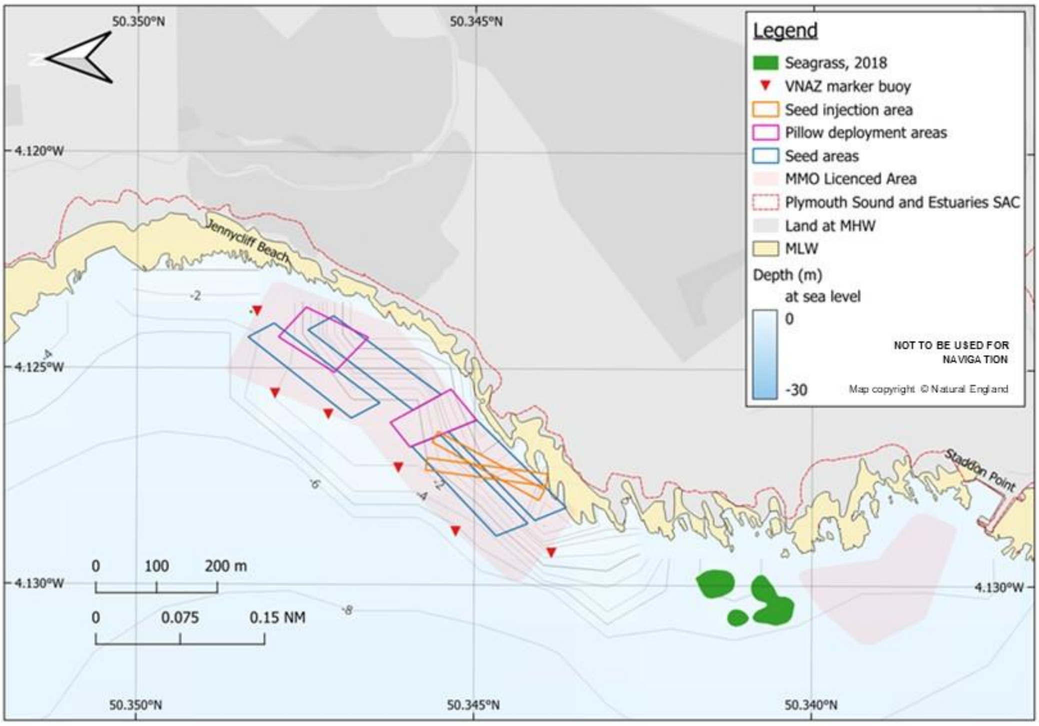

Figure 12: Location of the Ocean Conservation Trust restoration area within Jennycliff, Plymouth Sound and Estuaries Special Area of Conservation.
Image shows the intersected restoration modes, and the VNAZ marker buoys.

Plymouth Sound and Estuaries is a universally used site, with the volume of recreational activity pressures mostly concentrated to specific regions (Caals and others, 2024). For example, kayaking and [stand-up] paddleboarding are possible indirect pressures, as they allow access to low-tide regions where human-seagrass conflict could rise from trampling and snorkelling (Eckrich & Holmquist, 2000: Herrera-Silveira and others, 2010: NE, 2017).

As a shipping lane, there is an observed 'no anchoring' area from Penlee Point to just north-west of Pier Cove, with the western approaches Draystone red buoy just off Penlee Point. Within the bay, at Cawsand, AMS were applied on a dozen moorings using two designs of AMS; Seaflex, and Stirling by OCT (see figure 14). Cawsand Bay is a busy area, popular with recreational boaters and beachgoers alike. At the lowest tides below MLW, the moorings in Cawsand Bay are in 2 to 3 metres of water, making disturbance to the seabed from propellers a greater risk at low tide. Research by the MCS has shown that recovery from damage by exchanging traditional mooring systems with AMS is possible (Solandt, 2022), although multi-dynamic pressure factors are still considerable.

A VNAZ was established, specifically at Jennycliff Bay, to reduce impacts from recreational boaters and help protect areas of seagrass restoration. Outcomes of the restoration work are still under review, but current results seem promising (see, figure 14).

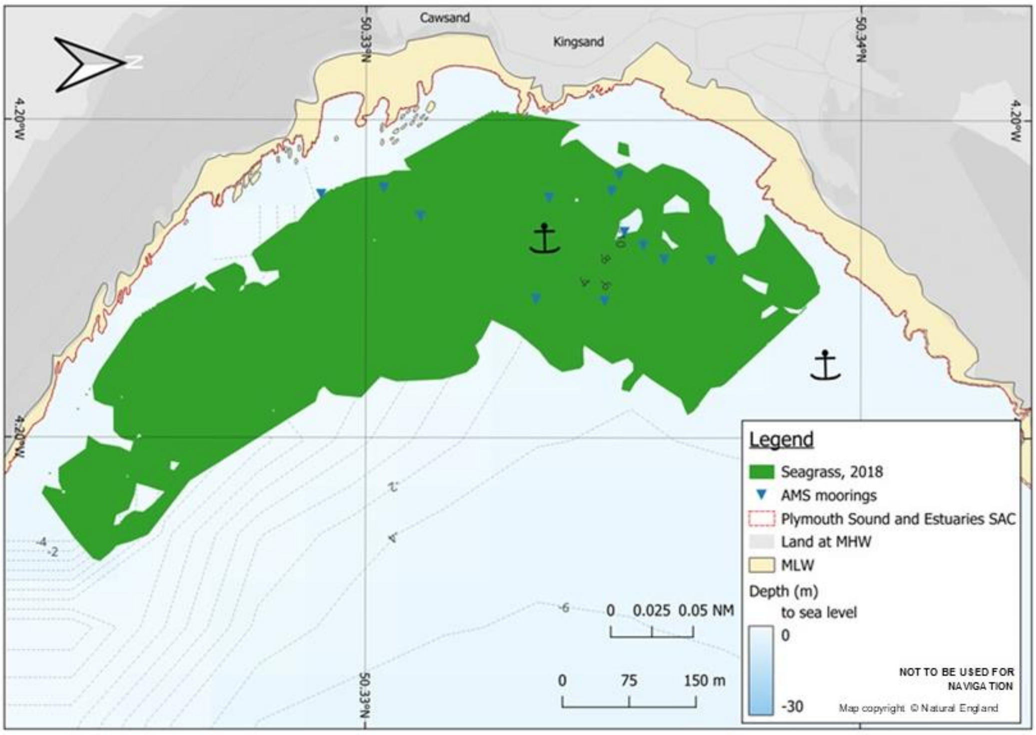

Figure 13: Cawsand Bay seagrass distribution in 2018, and locations of AMS moorings from ReMEDIES.
The map also shows the anchorage area adjacent to Cawsand and Kingsand.

A full summary of the different modes of modification and conservation measures applied in the ReMEDIES sites is provided in table 1.

Table 1: List of restoration and conservation measures, and their locations
Sites provided with notation. Note: some cells have been left blank.

Special Area of Conservation	AMS	Active Restoration	VNAZ	Long-term monitoring	Community engagement
Isles of Scilly Complex				X	
Essex Estuaries					X
Solent Maritime		X			X
Fal and Helford			X		X
Plymouth Sound and Estuaries	X	X	X		X

Method

This report focuses on two primary sources of monitoring evidence obtained in the field to inform the results and provide the assessment of the evidence of the seagrass condition, 1; Natural England Dive Survey Data (NE-DS), and 2; Environment Agency Drop-Down Video (EA-DDV) boat survey data. The two sampling regimes that provide this evidence for the report, are standardised surveying carried out by NE and EA on seagrass beds nationally. This therefore provides data that can be compared to other surveys, is repeatable for future assessments, and provides the level of accuracy to give good measures of reliability to the results in the report and therefore any conclusions determined from them. The full methodology for the standardised dive-based surveys can be reviewed in Curtis (2015), and Bunker and others (2020), and for the standardised DDV surveys in Kenworthy (2020), and Green (2022). Echosounder data for the sites was not used as it was not available consistently across all the sites, so therefore would not give comparable results. The full list of field data collection dates is given below in table 2. These different surveys and monitoring studies conducted by the EA and NE were used to provide the data used in the analysis. This study uses the baselines outlined in the table below as the comparison in the modelling exercise but will draw on prior information and data in the summery to interpret the outcomes of the results.

Table 2: Report specific monitoring and surveys of the SACs, 2015 to 2024

Start Date	SAC	Data type	Finish date
22/06/2015	Fal & Helford	Natural England Dive Survey (NE-DS)	25/06/2015
17/08/2015	Fal & Helford	Environment Agency Drop Down Video (EA-DDV)	27/08/2015
28/06/2018	Plymouth Sound & Estuaries	Environment Agency Drop Down Video (EA-DDV)	03/07/2018
23/07/2018	Plymouth Sound & Estuaries	Natural England Dive Survey (NE-DS)	26/07/2018
01/07/2021	Fal & Helford	Environment Agency Drop Down Video (EA-DDV)	08/09/2021
13/09/2021	Fal & Helford	Natural England Dive Survey (NE-DS)	17/09/2021
10/07/2023	Plymouth Sound & Estuaries	Natural England Dive Survey (NE-DS)	13/07/2023
08/08/2023	Plymouth Sound & Estuaries	Environment Agency Drop Down Video (EA-DDV)	03/09/2023
29/07/2024	Fal & Helford	Natural England Dive Survey (NE-DS)	01/08/2024

Attribute Assessment

To focus and standardise the assessment and provide the basis to make judgements on the status of the seagrass beds, elements of the attributes for the seagrass sub-feature of the SACs chosen to assess, were:

i. Hab_Att_1.01, Extent and distribution, by EA-DDV

ii. Hab_Att_3.09, Structure: non-native species and pathogens (habitat), by NE-DS
iii. Hab_Att_3.14, Structure: biomass, by NE-DS

Analysis

Extent and distribution

Extent and coverage distribution estimates were spatially modelled as estimated seagrass area (in m^2) using EA-DDV point data to generate each contour layer. This was estimated based on the attribute of percentage coverage of seagrass as point vector layer in the opensource QGIS spatial software within a coordinate reference system (CRS) of EPSG:27700 or OSGB36/British National Grid. The spatial tool calculated contours at multiples of a fixed intervals of 5% seagrass coverage by nearest neighbour triangulation. From this layer, the seagrass coverage was pooled at 10% seagrass coverage increments (excluding the 5-10% coverage range) from 5-100% seagrass coverage. The total extent was then extracted from the combined coverage layers, providing the total extent (maximum extent of geo-spatial created object layer) for each bed in the individual SACs for each EA survey. Coverage < 5% was excluded from the model as to qualify as a *Zostera* 'bed', seagrass densities should provide at least 5% coverage (OSPAR, 2008). This was replicated for both sites, Plymouth Sound and Estuaries, and Fal and Helford, for each of the beds within these SACs at the different yearly increments, outlined in table 2. For Fal and Helford, where the beds within the Helford Passge and St. Mawes areas are in proximity to each other and individually named beds are interconnected, the seagrass beds were pooled for total extent estimation as well as the individual bed extent estimation. This pooling process was also followed for Plymouth's Red Cove, as Red Cove North and Red Cove South had overarching point sampling layers, joining area of these two beds. All the beds were tested for differences in extent by years, per individual bed as well as total difference in extent per years across different surveys. The estimated percentage change ($\%C_E$) in the beds per year (y) was calculated by area of seagrass (A) in m^2 by equation 1, below, and used in non-parametric tests to assess differences between years of surveys, by percentage coverage, percentage change by year, and total extent. For clarity, extent area in m^2 was converted to hectare (Ha) after modelling and calculations.

$$\%C_e = \frac{(Ay_{ii} - Ay_i)}{Ay_i} * 100$$

Equation 1: calculation of estimated percentage coverage

Non-native species and pathogens

Possible non-native species and pathogens were modelled by the NE-DS records of infection burden measured by leaf darkening score. The leaf-darkening score, whilst not a formally defined metric, can be used to assess the health of seagrass beds as it provides a ranked estimate of darkening on seagrass leaves which can come directly and indirectly (*e.g.* as stress mechanism or 'wasting disease') from the protist *Labyrinhula* spp. (Burdick and other, 1993: Sullivan and others, 2018). As the disease manifests as leaf discolouration and necrosis, causing rapid loss of photosynthetic ability and buoyancy in leaves (Duffin and others, 2020: Ralph & Short, 2002), it is a suitable measurement of non-native species and pathogens. The scale has 'absence of infection' (score = 0), to 'high infection' (score = 5): by combining the scores which showed presence of darkening (score 1 – 5), independently of scale of infection, the 0 to 5 ranked scoring systems were transformed into presence and absence data and separate binary categories. These were then summed in contingency tables where they can be analysed for association between infection presence at sites using Pearson's Chi-squared test with Yates' continuity correction for 2 x 2 tables. A contingency table (cross-tabulation) summarises the frequency distribution of categorical variables, showing how they interact. The test determines if observed frequencies in the table differ significantly from expected frequencies under independence, allowing for the assessment of relationships between presence or absence of infection and different seagrass beds.

Biomass

Because seagrass biomass is the amount of organic material produced by seagrasses, encompassing both above-ground (shoots) and below-ground (rhizomes and roots) parts, longest leaf-length was analysed from the NE-DS data for each site to give a practical estimation of biomass present in the bed. As there were inconsistencies in the number of transects per bed, the number of quadrates per transect, and the number of leaves measured varied in each sampling quadrate across all the NS-DS datasets; data for each transect was pooled by each quadrate, and then transect, to give a single mean sample estimate of biomass per bed. These measurements from each bed's leaf lengths were statistically tested against each year by individual beds using non-parametric tests. Site data for Plymouth Sound and Estuaries was compared by yearly comparisons per bed, and replicated for both sites, Plymouth Sound and Estuaries, and Fal and Helford, for each of the beds within the SACs at the different yearly time periods.

The statistical structure applied

All the non-parametric and parametric tests used a one-way analysis of the seagrass extent data for each comparison across the years and beds. As the beds represented the same group with annual conditions, the tests assumed the seagrass bed by site analysis by year was paired, unless the groups had uneven sample sizes because of the natural variation within the yearly samples, in which case an independent one-way un-paired test was used. Parametric multiple comparisons for biomass were conducted by a pairwise t-test as *post-hoc* test where the initial p-value allowed it. This compared all possible pairs of means within the sample with a pooled standard deviation. For non-parametric multiple comparisons tests, a *post-hoc* Dunn test was applied to the results that were significant. With all *post-hoc* tests, a Bonferroni correction for comparison of multiple p-values was applied to limit family-wise error rate or increased risk of a type I error. A reduced sample procedure of removing 0 in observations with zero-inflated data applied to ranked-based tests used in the extent analysis, but outliers remained in the models as they can't be excluded as not being from natural variation.

Error measurements for the spatial models of seagrass extents used a combination of, 1; root mean square error (RMSE), which measures the accuracy of expected (true) values compared to observed values by quantifying the average magnitude of errors between predicted and actual measurements, giving higher weight to larger errors due to the squaring operation, and 2; element-wise absolute difference and the element-wise absolute percent difference, both measuring the absolute deviation between corresponding elements in two (observed vs. expected value) datasets, and treats all errors linearly. These two error estimations are commonly used in spatial analysis for error quantification (Congalton & Green, 2019: Li and others, 2005). The assessment of the error within the analysis was between the observed (report model) and expected (validation model). The difference between the observed and expected was tested by a likelihood ratio test (G-test) for proportions and cross validated with a (paired) nonparametric one-way analysis.

The analysis of data was achieved in opensource R-Programming language (R Core Team, 2022) using either base or dplyr, psych, and Metrics package. Statistical significance was determined at the *p-value* = < 0.05 level for all analysis (α = 0.05).

Table 3: Full list of seagrass beds under review in report, with beds that have been amalgamated for analysis.

The beds that have been amalgamated in the assessment are Red Cove 1, Helford Passage 2, Carrick Roads 3, Falmouth Bay 4, and Percuil River 5.

Plymouth Sound & Estuaries	Fal & Helford
Tombs Rock	Polgwidden Cove (Durgan)[2]
Cawsand Bay	Amsterdam Pont to Carricknath Point[5]
Firestone Bay	St. Mawes Harbour[5]
Cellar Cove	Flushing
Drakes Island	Penarrow Point[3]
Red Cove[1]	Gyllyngvase[4]
Red Cove[1]	St. Mawes Bank[3]
Jennycliff (north)	Passage Cove[2]
Jennycliff (south)	Parbean Cove
	Swanpool[4]
	Porthhallow
	Bosahan[2]

Results

Plymouth Sound & Estuaries: Comparisons of Estimated Extent, 2018 verses 2023

Table 4: Comparisons of the total extent of the seagrass beds within Plymouth Sound and Estuaries SAC as estimated by the spatial model
Table shows calculated coverage, per bed in m^2, in 2018 and 2023, difference between years, and the percentage change. Total difference as loss (minus) or gain (plus) in SAC is provided at base of table. Beds where some kind of active modification system from ReMEDIES was applied is denoted by +

Bed name	area 2018 (m^2)	area 2023 (m^2)	Difference (y_2 - y_1)	$\%C_e$
Cawsand Bay+	179173	215667	**+36493**	20
Firestone Bay	3165	5498	**+2333**	74
Cellar Cove	47532	49098	**+1566**	3
Drakes Island	39174	33898	**+5276**	13
Red Cove	19123	19684	**+561**	3
Jennycliff (north)	29	0	**-29**	-100
Jennycliff (south)	6432	4760	**-1672**	-26
Tombs Rock	13654	8197	**-5457**	-40
		Total (m^2) =	**+28519**	
		Total (Ha) =	**+3**	

Indicative results of the estimated total extent, per year, for Plymouth Sound and Estuaries where active modification techniques were applied at Cawsand Bay and Jennycliff Bay (excluding Jennycliff north and south), are shown in Table 2. Results show a projected overall gain in the SAC of 3 Ha of seagrass beds over the five-year period between surveys, but this was not found to be statistically significant (Wilcoxon test: V = 18, df = 15, p-$value$ = 1). There was a 100% reduction of seagrass at Jennycliff (north) within the time-period. Gains in bed size at Cawsand Bay, where 22 Ha (215667 m^2) was estimated from an 18 Ha (179173 m^2) in 2018 were also projected. However, differences in monitoring and sampling size have impacted the estimation, with a lot of known uncertainty over the accuracy presented here (see limitations and further results). Figure 17 shows seagrass sampling points, where the spatial model provided an average estimated bed size of 3 Ha (31870 ± 43532 m^2) in 2018, and 5 Ha (48489 ± 92690 m^2) in 2023.

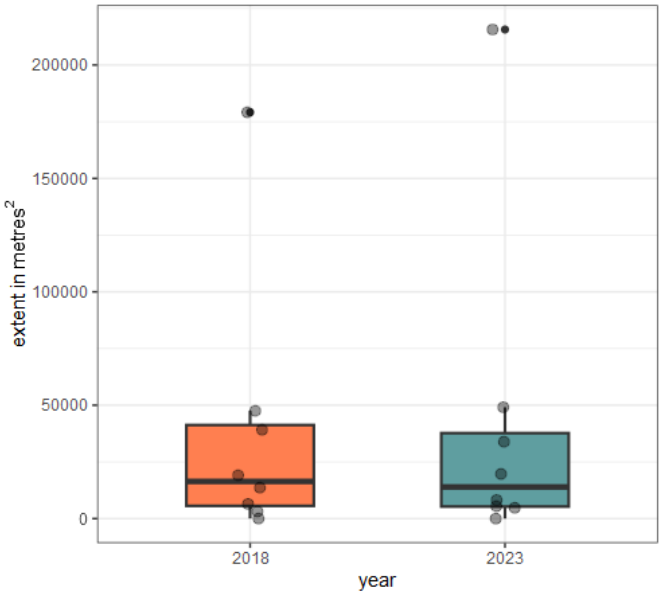

Figure 14: Boxplot of extent data for Plymouth Sound and Estuaries.
Outliers of Cawsand Bay are shown.

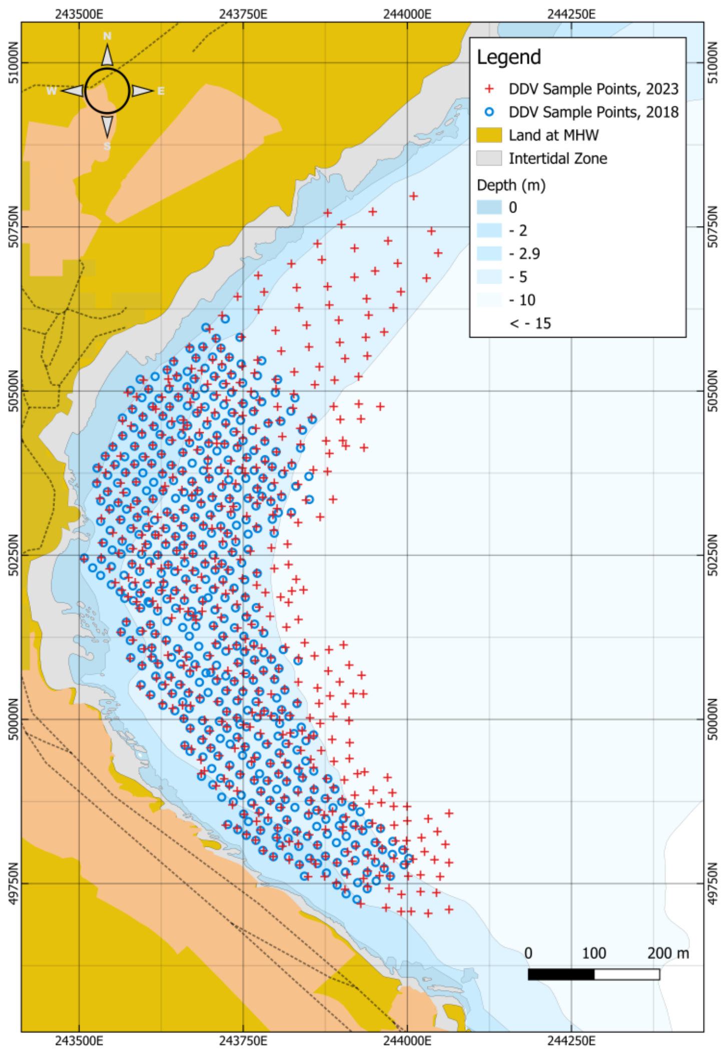

Figure 15: Map of seagrass data points at Cawsand Bay, 2018 and 2023.
The different survey coverage is shown, with 2023 being a wider area surveyed

Whilst modelled Cawsand extent is estimated to have gained size, see figure 18, the extent change between the years was not significantly different (Wilcoxon test: $V = 35$, $df = 19$, $p\text{-}value = 0.492$). It is not clear how the different amount of sampling points between the two years has affected these results, and this is discussed in limitations section (page 56); as such, low confidence can be applied to the temporal data presented due to different survey areas between years. The mean distance between sampling points was 17.17 m for 2018 ($n = 484$) with a RMSE = 2.529% (33.92 ± 30.62%) for the coverage estimates, and 16.81 m for 2023 ($n = 607$) with a RMSE = 2.095% (11.88 ± 13.77%) respectively. The model suggests the largest estimated coverage for both 2018 and 2023 was the 10-20% range, with 2018 having 3 Ha (32997 m^2) of estimated coverage within this range, and 10 Ha (99142 m^2) in 2023. Extent of coverage within the 60-100% range was 1 Ha (28340 m^2) in 2018, but 0.1 Ha (1461 m^2) in 2023; a $\%C_e = $ -85 of seagrass in this 60-100% range, suggested bed size has increased but health of bed has declined due to a reduced area of core dense seagrass required for resilience.

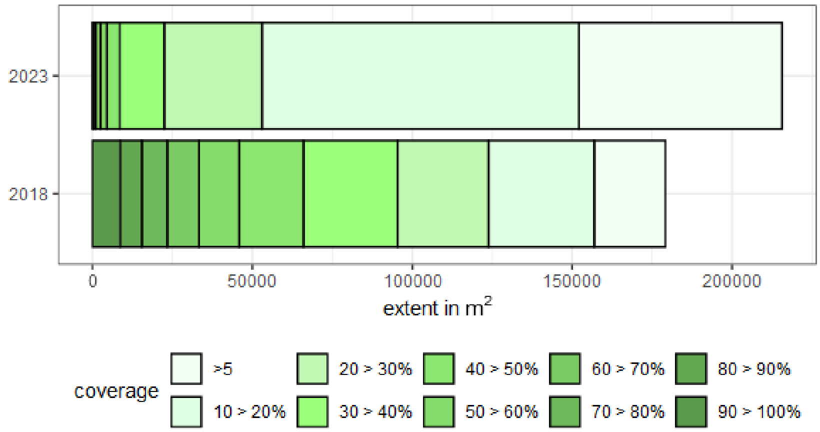

Figure 16: Extent per percentage coverage for Cawsand Bay, years 2018 and 2023.

The seagrass at Cellar Cove is a large bed within the SAC, with a relatively constant estimated coverage extent between years. This is further shown with no significant difference found (Wilcoxon test: $V = 15$, $df = 19$, $p\text{-}value = 0.232$). The difference in the sampling points was also noted in the volume per year, with fewer in 2018 ($n = 169$) to 2023 ($n = 204$). 2018's RMSE = 2.288% (42.79 ± 42.62%) from the coverage model was slightly higher than 2023's, which was RMSE = 2.158% (38.20 ± 42.33%). Both, however, demonstrate extremely low levels of difference between the observed coverage and estimated coverage within the model, and that beds density is constant between years. Based on modelled coverage, a slight change in the health was detected, as the 60-100% coverage range was estimated to have dropped from 3 Ha (26404 m^2) in 2018 to 3 Ha (26101 m^2) in 2023. This represents a $\%C_e$ = -1 in healthy bed at Cellar Cove. However, coverage within the 60-80% coverage range was estimated to have increased within the beds within the surveys from 0.7 Ha (6604 m^2) to 1 Ha (12025 m^2), indicating the loss in health is minimal.

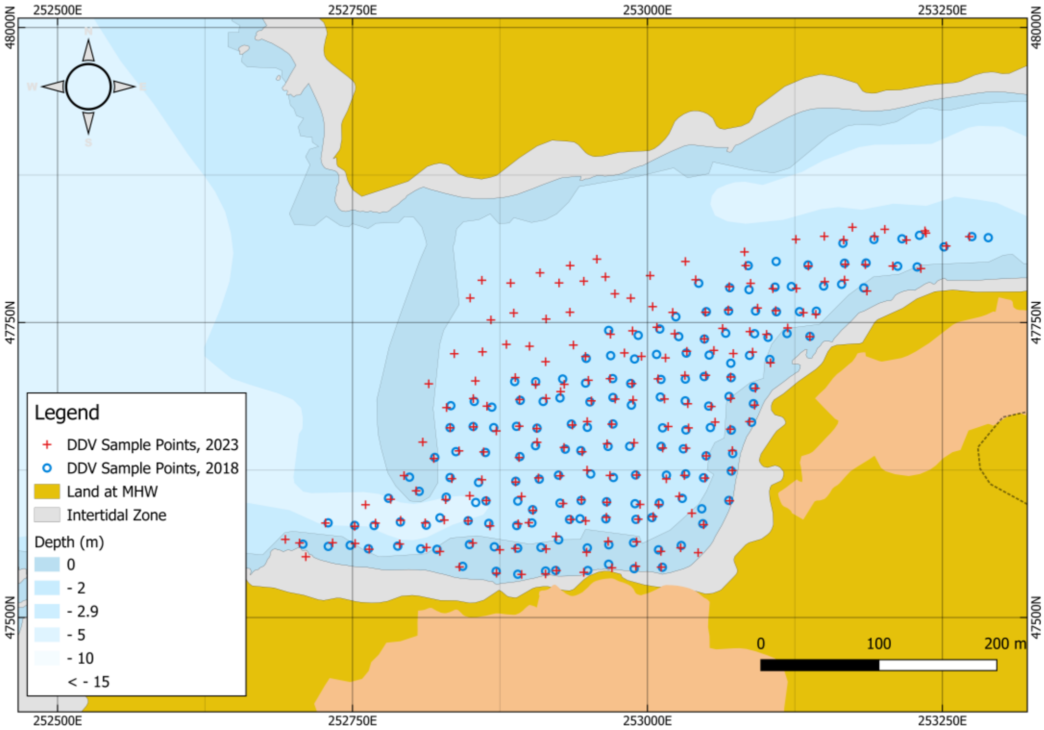

Figure 17: Map of seagrass data points at Cellar Cove, 2018 and 2023.
The different coverage is shown.

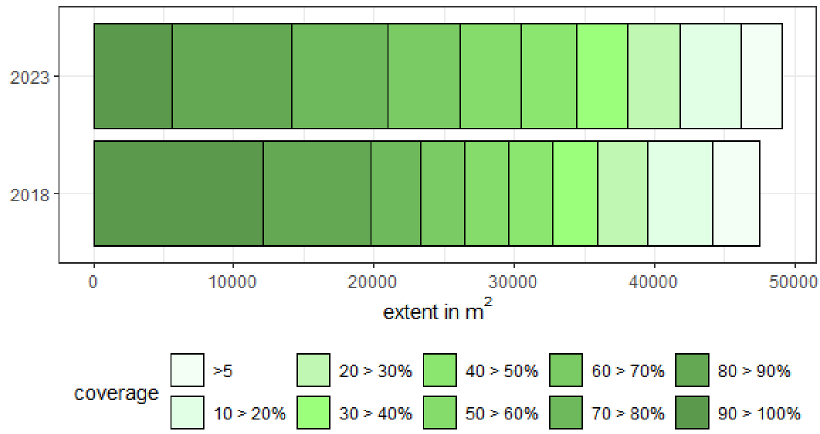

Figure 18: Extent per percentage coverage for Cellar Cove, years 2018 and 2023.
The bed shows stability in estimated coverage and extent.

Results on seagrass around Drakes Island suggest negative ecological changes within the area. This was based on 2018 sampling points (n = 138) that gave a model RMSE = 2.059% (36.29 ± 42.95%) from a mean distance of points at 16.95 m. For 2023, sampling points (n = 151) gave a RMSE = 1.662% (15.66 ± 21.50%) from a mean distance of sampling points of 16.61 m. Whilst there is no significant difference between survey years (Wilcoxon test: V = 27, df = 19, *p-value* = 1), the model suggests there has been large losses in the estimated extent between the 60-100% coverage range. A $\%C_e$= -85 was estimated within this range, with 2018's 1 Ha (12892 m^2) dropping to 0.2 Ha (1895 m^2) by the next survey in 2023. Estimated extent as a total, however, has increased. With coverage within the 30-40% coverage range showing the marginal gains in estimated extent; a change from 0.3 Ha (3402 m^2) in 2018 to 0.8 (8484 m^2) in 2023. The largest extent in each year was 10-20% range in 2018, of 0.4 Ha (4318 m^2), to 0.9 Ha (8484 m^2) of 20-30% in 2023. Together, the results indicate a decrease in density of the bed at Drakes Island could be occurring.

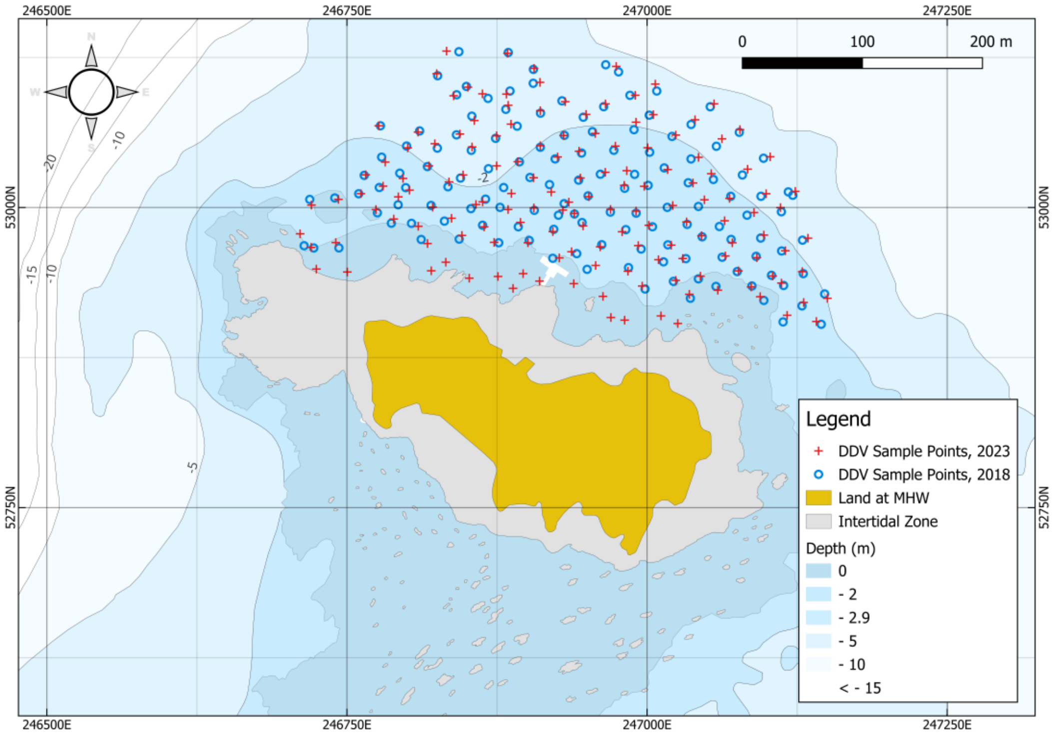

Figure 19: Map of estimated seagrass sampling points at Drakes Island, 2018 and 2023.

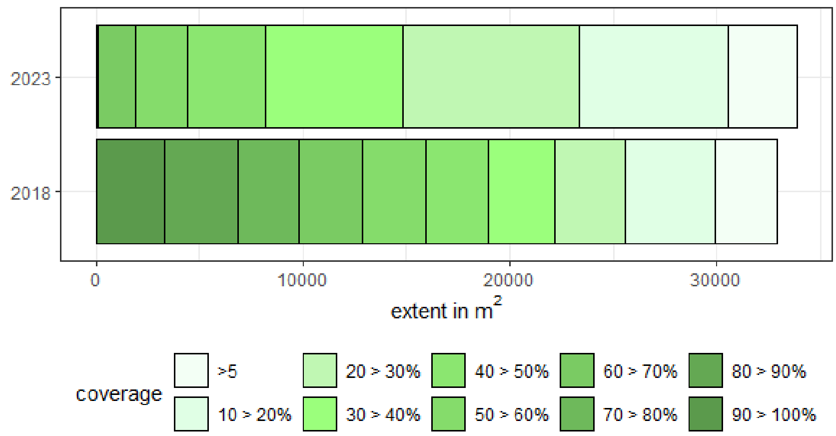

Figure 20: Extent per percentage coverage for Drakes Island, years 2018 and 2023.

At Firestone Bay sampling points (as shown in figure 23) were relatively proportionate by each year, as demonstrated by 2018's (n = 46) mean distance of 14.87 m to 2023's (n = 45) 16.14 m. The accuracy in the model from the points gave a RMSE = 0.722% for 2018 (3.04 ± 6.99%), and for 2023 a RMSE = 1.406% (6.27 ± 16.10%). The changes to Firestone Bay were not statistically significant (Mann-Whitney U test: W = 20, df = 11, *p-value* = 0.282). From a bed that was estimated to be entirely in the 5-30% coverage range in 2018, a 5-year difference has increased the bed size and the coverage to a 5-90% coverage range. 25% of the 2023 bed, or 0.02 Ha (207 m^2), is now in the 60-90% range category, suggesting a positive change in the health of the bed. A small seagrass bed within the SAC, Firestone Bay has been estimated to have changed considerably during the timeframe in the study.

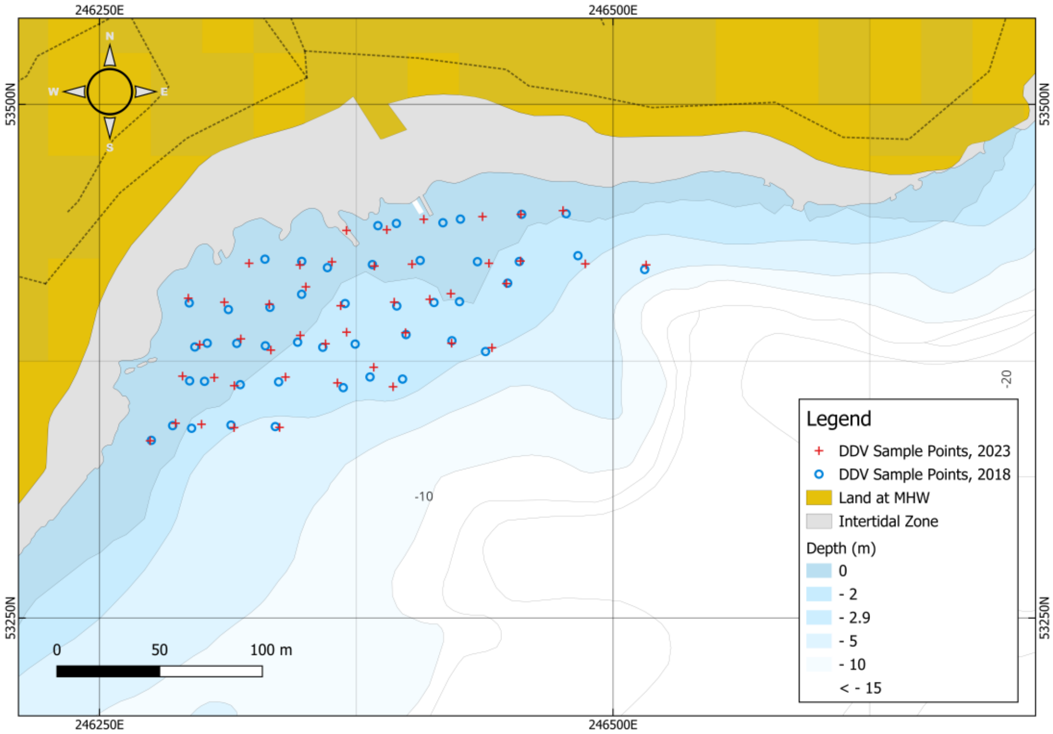

Figure 21: Map of sampling points at Firestone Bay, 2018 and 2023.
The similarity in coverage is shown.

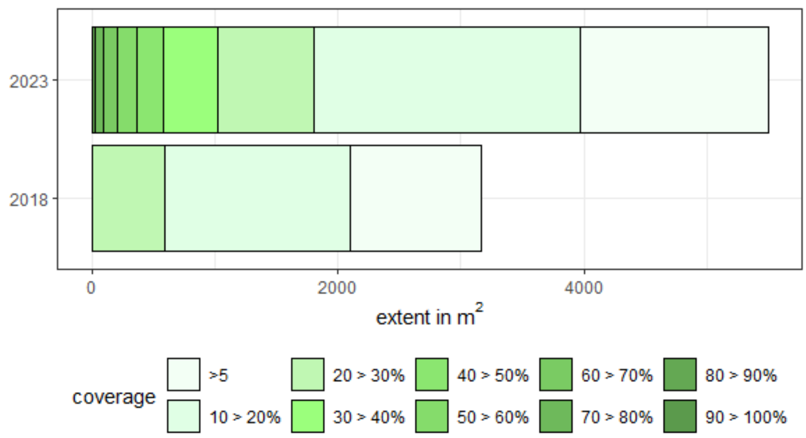

Figure 22: Extent per percentage coverage for Firestone Bay, years 2018 and 2023.

The EA-DDV surveyed historically known beds at Jennycliff, which lie to the north and south of the ReMEDIES restoration area (see figure 23). These previously surveyed beds showed considerable differences between study periods. Jennycliff North had a single small patch of seagrass in 2018, with a model estimated extent of 0.0029 Ha (29 m^2) within the 5-10% coverage range with a RMSE = 0.267% in the spatial model. Counts of sample points during the first survey in 2018 (n = 14) (0.42 ± 1.55%) were like 2023's (n = 13), with 2018 having a mean distance of 19.12 m for the sample points, 2023 having a mean distance of 18.82 m. The Jennycliff north bed was not found by the 2023 survey, with samples giving 0% cover at the sampling points for this year. Jennycliff south moved from a bed with a modelled extent coverage range of 5-40% in 2018, to a 5-20% estimated coverage range bed. Sampling points for Jennycliff south were proportionally similar, but sampling distribution was notably different. 2018 (n = 77) had a mean distance of 17.39 m, whilst 2023 (n = 79) had a mean distance of 6.67 m. The accuracy of the model for 2018 was therefore given as a RMSE = 1.473% for 2018 (2.97 ± 6.34%) and a RMSE = 2.239% for 2023 (2.29 ± 3.691%). However, analysis of the coverage showed that no significant difference was found (Mann-Whitney U test: W = 3, df = 5, $p\text{-}value$ = 0.817). The changes in the 5-20% coverage for both beds 2018 and 2024 were a $\%C_e$ = -19, with 0.6 Ha (5880 m^2) in 2018 and 0.5 Ha (4760 m^2). The condition of both the historic beds within Jennycliff shows serious declines, and, the breakdown of bed structure within these survey periods for the seagrass at the north and south of Jennycliff Bay.

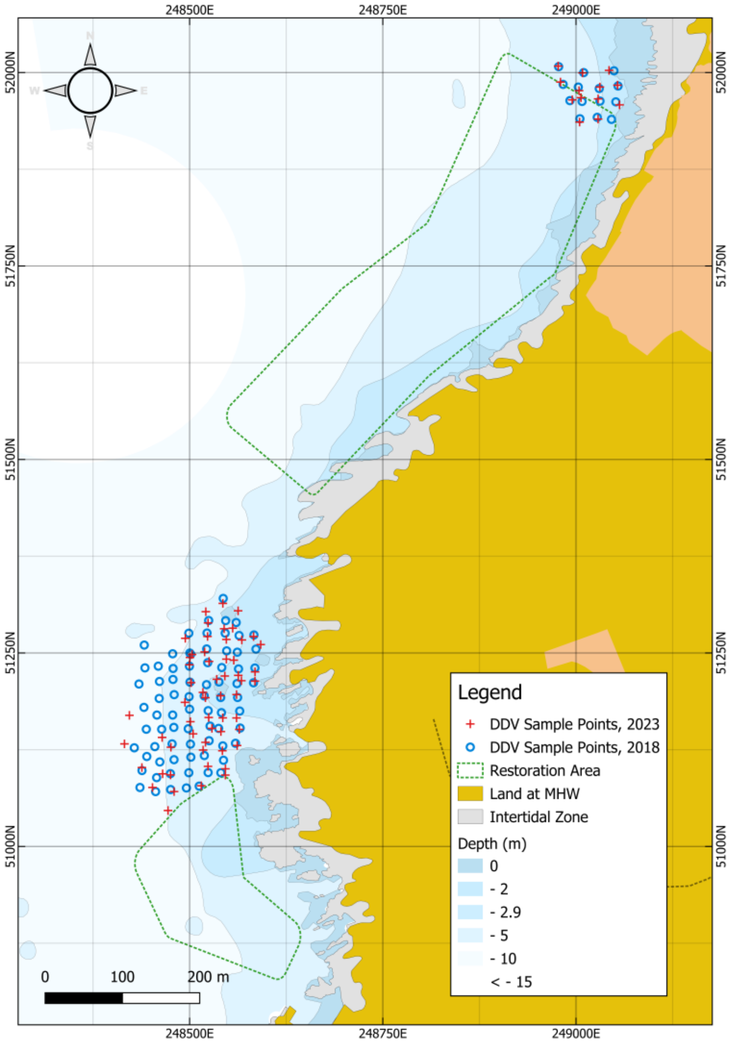

Figure 23: Map of sampling points at Jennycliff (north and south), 2018 and 2023.

The ReMEDIES restoration area is shown in the green polygon, mostly outside the sampled areas by EA-DDV.

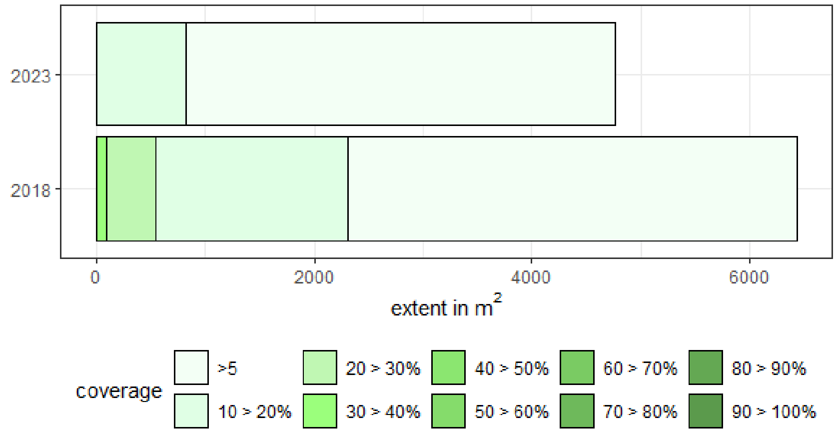

Figure 24: Extent per percentage coverage for Jennycliff (south), years 2018 and 2023.

Red Cove north and south beds (Figures 27 and 28) are shown to be another combination of seagrass bed that has remained relatively stable in extent but has had interesting changes to the structure. One of the relatively largest changes being the estimated extent increase in 90-100% coverage range, where 0.007 Ha (79 m^2) in 2018 was estimated at Red Cove to 0.04 Ha (374 m2) in 2023 occurred between survey times. Additionally, decreases in the 40-90% coverage range, where a $\%C_e$ = -14 contraction in coverage occurred. But, with small gains and losses in density, the bed has been stabling within the survey periods and no significant change between the estimated extent by the coverage was found (Wilcoxon test: *V* = 23, *df* = 19, *p-value* = 0.695). The model RMSE = 1.952% for 2018 (28.48 ± 35.36%) for sampling points (*n* = 84) at 16.52 m mean distances, and 2023 RMSE = 2.340% (39.84 ± 39.86%) for sampling points (*n* = 82) with a mean distance of 10.87 m follows the model prediction showing marginal differences between study periods.

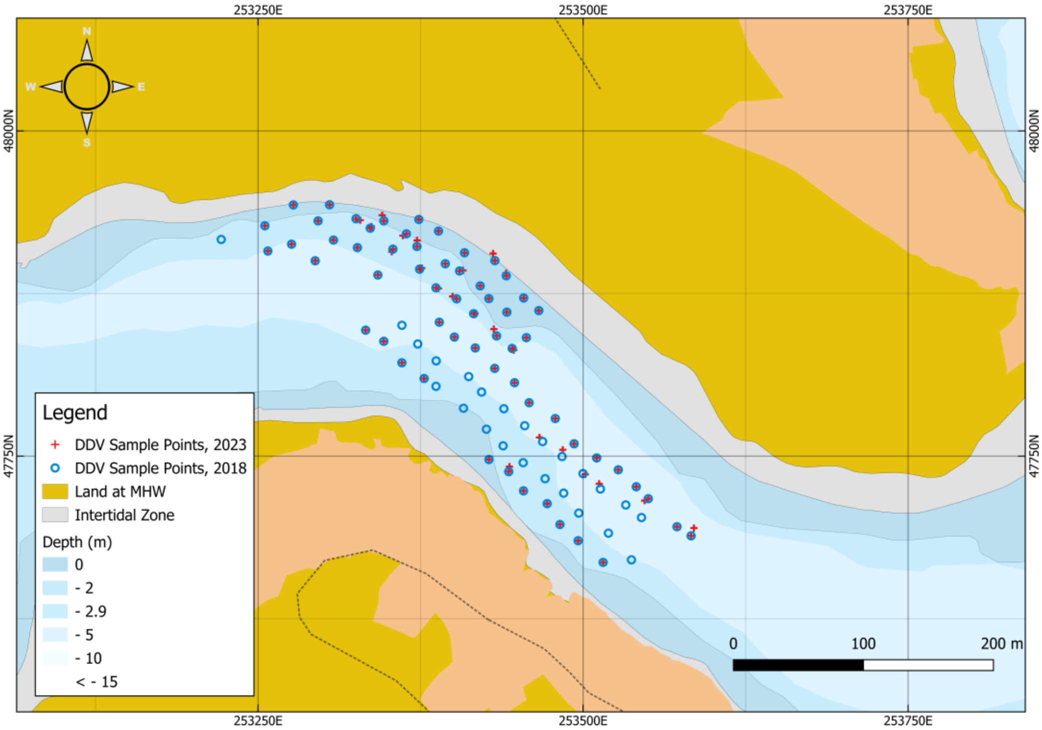

Figure 25: Map of sampling points at Red Cove, 2018 and 2023.

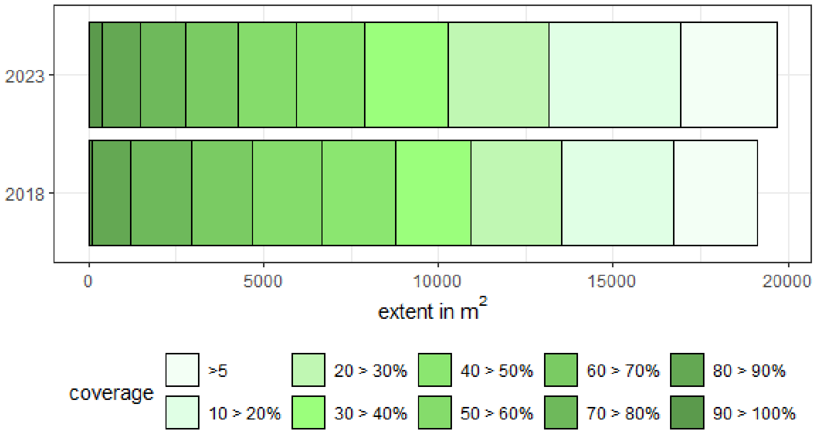

Figure 26: Extent per percentage coverage for Red Cove, years 2018 and 2023.

Tombs Rock showed did not have sufficient data to test statistically with any confidence. The spatial model's sampling points did show differences in mean values for percentage coverage, with 2018 (n = 53) being lower (6.23 ± 6.64%) than 2023's (n = 51) (3.84 ± 4.44%), providing a RMSE = 2.390% for 2018, and RMSE = 2.214% for 2023. This was based on a mean distance of 17.62 m in 2018 and 17.18 m in 2023. From a bed in the 5-30% coverage range in 2018, it is estimated by 2023 to be only within the 10-20% coverage range. 0.1 Ha (1208 m^2) of bed estimated to be remaining in 2023. These results should be interpreted by the availability of only a small dataset, however. Figure 30 gives a better indication of the changes at the bed, which although is relatively small within the SAC, has changed structurally.

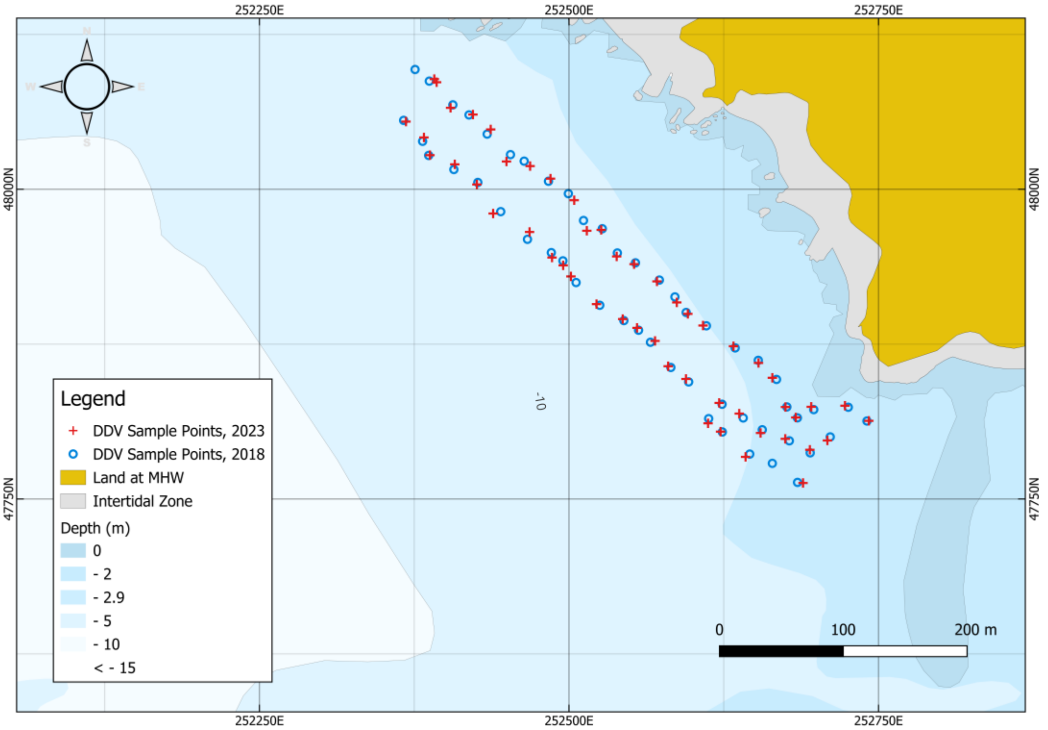

Figure 27: Map of sampling points at Tombs Rock, 2018 and 2023.

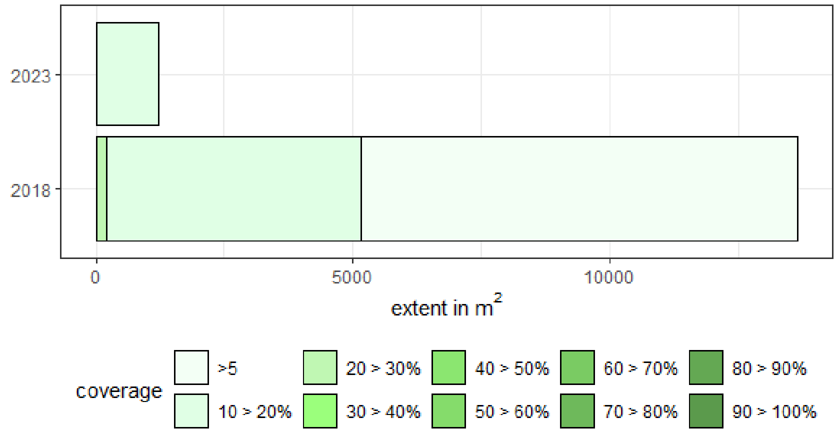

Figure 28: Extent per percentage coverage for Tombs Rock, years 2018 and 2023.

Plymouth Sound & Estuaries: Assessment of Infection Burden (non-native species and pathogens) across years

Comparison of in-year (2018) potential infection burden, estimated by leaf darkening, at three seagrass beds in Plymouth SAC is summarised in Table 4. Analysis showed a significant difference between seagrass beds and present or absence of possible infection by indicated by leaf darkening (Pearson's Chi-squared test: $\chi^2 = 136.3$, $df = 2$, $p\text{-}value = < 0.05$), with Drake's Island having the highest proportion of infection present from the sampled bed.

Table 5: Contingency table for 2018 seagrass infection burden estimation by presence/absence at Plymouth Sound and Estuaries SAC.
Proportions of counts of infection per leaf in parenthesis.

Seagrass Bed	Presence	Absence	Totals
Cawsand Bay	763 (0.423)	1040 (0.577)	**1803**
Drakes Island	424 (0.525)	384 (0.475)	**808**
Yealm	788 (0.311)	1745 (0.689)	**2533**
Totals	**1975**	**3169**	

Figure 31 shows the relationship between proportions and level of infection between beds. Tests showed no significant difference between seagrass beds (Kruskal-Wallis test: χ^2 = 0.187, *df* = 2, *p-value* = 0.911), with marginally an absence of infection burden constant between the different beds. The results suggest that whilst there is an identified difference in infection within the SAC's seagrass beds, levels of infection do not statistically differ by proportions of infection in beds and are low with > 50% of the seagrass beds in the Plymouth area being absent of infection in the 2018 sample area. This is further shown by only 5.07% of the bed having an infection burden in the range of a half- to high-infection ranks for 2018 (Figure 31).

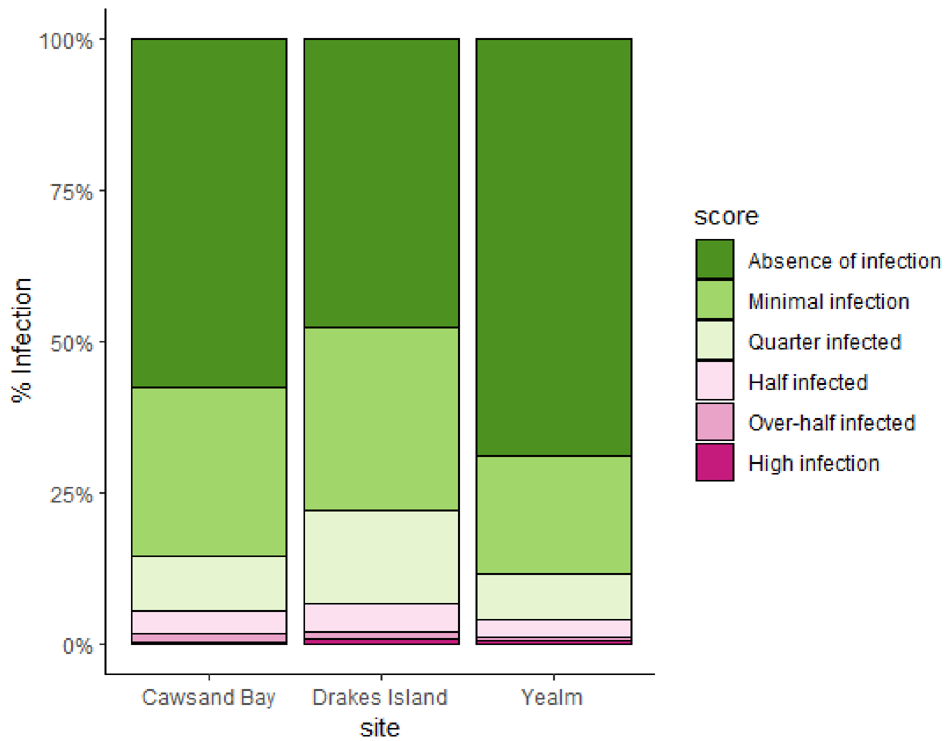

Figure 29: Stacked bar chart of levels of infection (0 low to 5 high) at Plymouth seagrass beds for 2018.

Comparing the results of 2023 in-year, infection burden (presence/absence) significantly differed between sites (Pearson's Chi-squared test: $\chi^2 = 256.71$, $df = 1$, $p\text{-}value = < 0.05$). Low-infection burden was found for Cawsand Bay, with 73.29% of the bed in the sample area being absent of any infection. Drakes Island was proportionally more affected by the data collected, however, 3.51% of this is in the half-infected and above ranking proportion, suggesting infection is localised, but not in the highly-infected category, and indicating a possible bed-specific influence on infection.

Table 6: Contingency table for 2023 seagrass infection burden estimation by presence/absence at Plymouth Sound and Estuaries SAC.

Proportions of counts of infection per leaf in parenthesis.

Seagrass bed	Presence	Absence	Totals
Cawsand Bay	394 (0.267)	1081 (0.733)	**1475**
Drakes Island	672 (0.576)	495 (0.424)	**1167**
Totals	**1066**	**1576**	

Comparison between seagrass beds for 2021 showed no significant difference between the scored infection burden results (Wilcoxon test: $V = 6$, $df = 11$, $p\text{-}value = 0.438$), which follows the finding of the contingency table analysis. Cawsand Bay, had a very low-level of infection burden by ranking, with only 0.61% of ranked data in the half- to highly-infected category. This can be shown in the bar chart, figure 32, page 36. The results suggest for 2023, that a high-level of infection burden were found to be low overall, with Drakes Island being a more infected seagrass bed, but still mostly in the lower ranked infection categories.

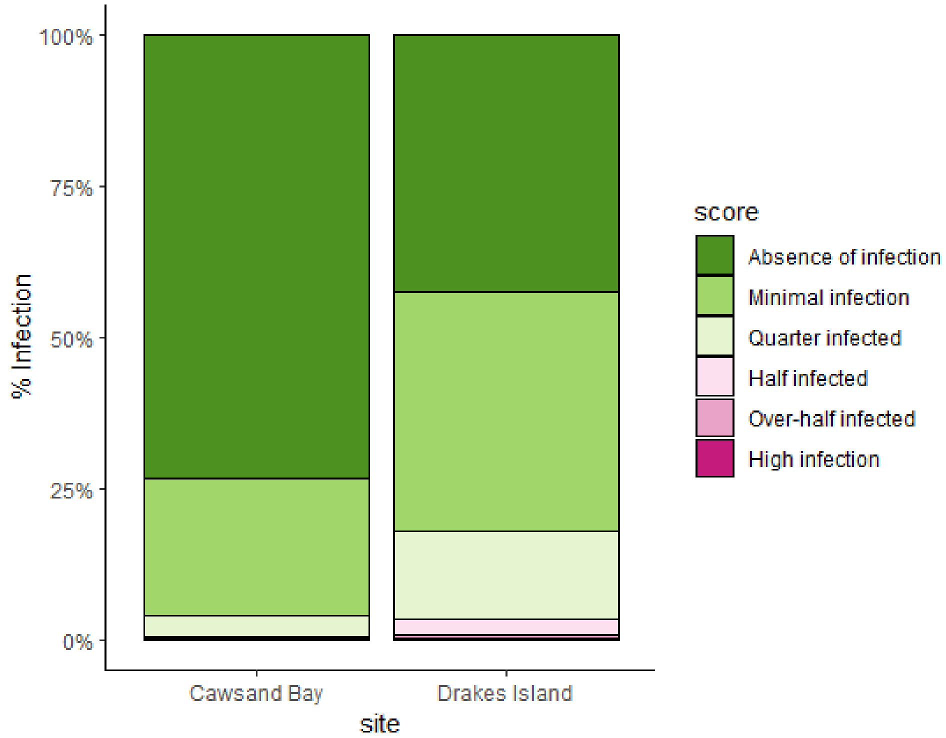

Figure 30: stacked bar chart of levels of infection (1 low to 5 high) at Plymouth seagrass beds for 2023.

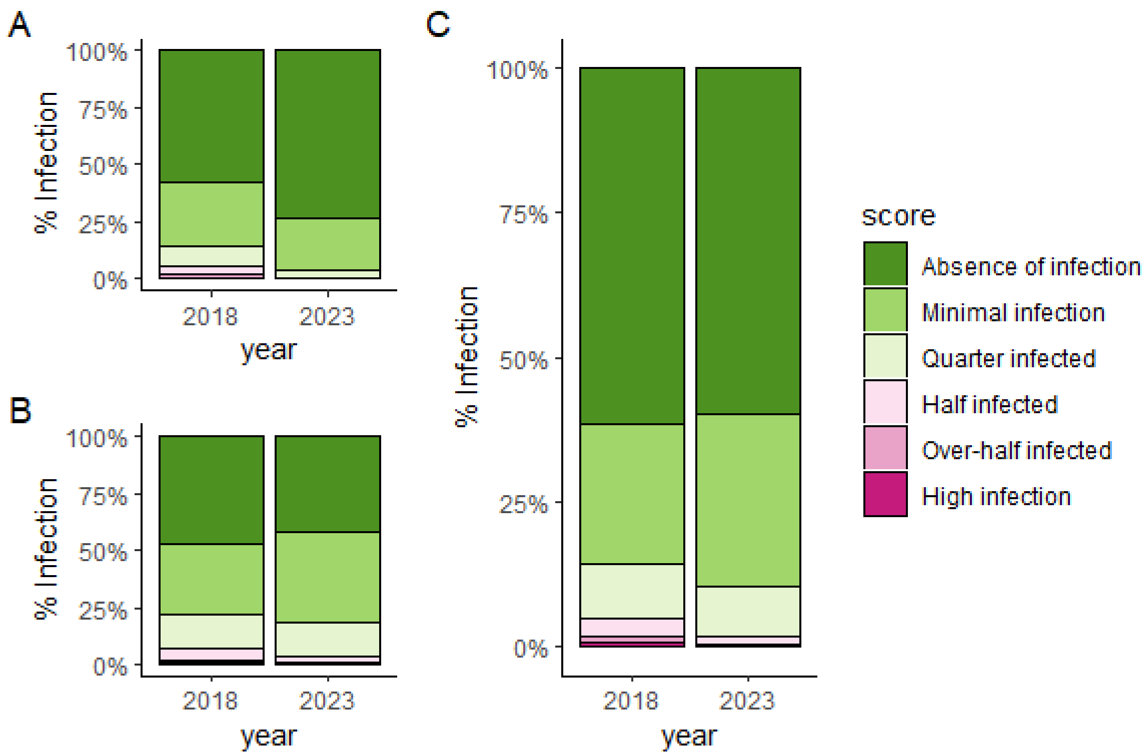

Figure 31: Stacked bar chart comparing infection burden in Plymouth Sound & Estuaries SAC.
Plot shows Cawsand Bay (A), Drakes Island (B), and total infection all beds (C) for 2018 and 2023.

When comparing beds between years, (Figure 33), there is lower level of infection, indicated by leaf darkening, across the years within the seagrass beds at Plymouth Sound and Estuaries. Cawsand Bay shows a stable but low presence of infection burden, demonstrated by the lack of significance (Wilcoxon test: V = 15, df = 11, *p-value* = 0.438) for the yearly difference. The proportion of leaves free from infection increased from 57.68% in 2018, to 73.29% of the Plymouth Sound and Estuaries seagrass beds in 2021, with high-infection declining from 0.28% of sampled leaves to 0.07% by 2021. No significant difference was found in the infection rate at Drakes Island (Wilcoxon test: V = 15, df = 11, *p-value* = 0.436) between the two survey times, with 47.52% of the sampled leaves infection free in 2018, and 42.42% in 2021. All beds combined, per year, shown in figure 33 (C), page 36, shows this stable burden of infection once again. Testing the differences between years for all beds was not statistically significant (Wilcoxon test: V = 15, df = 11, *p-value* = 0.438).   The proportion of leaves free of infection was 61.61% in 2015 and 59.65% in 2021. The proportion of leaves sampled in Plymouth in the high-infection burden categories dropped from 5.07% of the beds in 2015 to 1.89% of the beds in 2021.

These results suggest that Plymouth Sound and Estuaries has remained relatively stable in infection rates between the years. The results also suggest low overall infection burdens are present, and even small declines in the higher-infection burdens on biomass present at the SAC.

Plymouth Sound & Estuaries: Assessment of Longest Leaf-Length (biomass), 2018 verses 2023

When considering longest leaf length – as a proxy for biomass – Cawsand Bay (figure 34: A) showed the lowest leaf-length in 2018 (52.91±16.83 cm), with a similar average in 2023 (55.84 ±10.97 cm), with no significant different (t-test: t = -0.919, df = 97, *p-value* = 0.361), and between the two years sampled the variation is limited.

Drakes Island longest leaf-length data showed slightly more differences; however, these were also not significant (t-test: t = -1.773, df = 63.963, *p-value* = 0.08), in 2018 the averages length (76.12 ± 23.94 cm) was slightly shorter than 2023's leaf-lengths (85.79 ± 20.37 cm). The differences can be inspected in figure 34 (B).

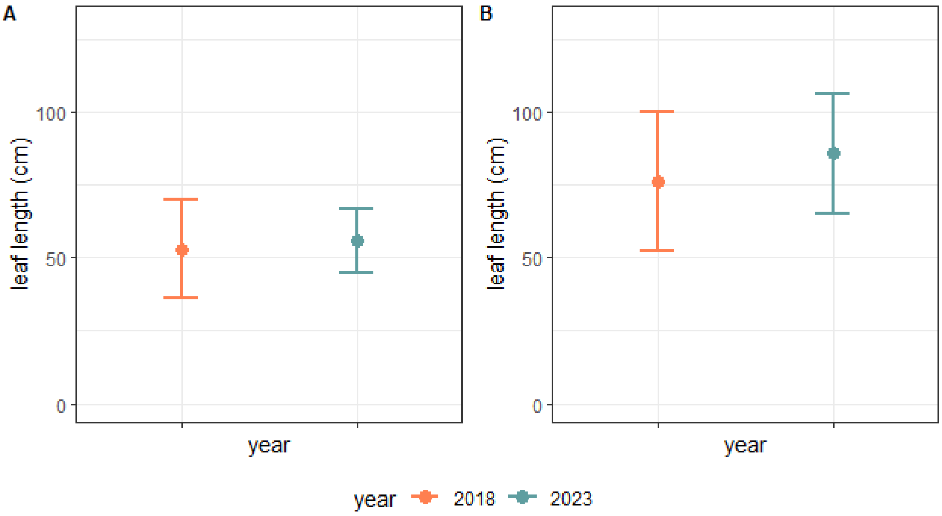

Figure 32: Biomass by leaf-length for Plymouth Sound and Estuaries SAC for 2018 and 2023.
Plots shows Cawsand Bay (A) and Drakes Island (B) mean biomass, with whisker bars of standard deviation, from the samples.

When comparing the differences between the seagrass beds in Plymouth Sound and Estuaries across the years, analysis showed a significance difference between seagrass sample sites (ANOVA: $F_{(3,\ 161)}$ = 29.454, *p-value* = < 0.05). Both Cawsand Bay and Drakes Island leaf-length differed in 2018 (t-test: $t_{(65,\ 36)}$ = *NA*, p-value = < 0.05), and 2023 (t-test: $t_{(3,\ 30)}$ = *NA*, p-value = < 0.05). The totals for the longest leaf-length for both seagrass beds in 2018 and 2023, showed a significant difference between years (t-test: t = -9.038, *df* = 163, *p-value* = < 0.05). Consolidation of these results suggest that, using longest leaf-length as a proxy for biomass; by these estimates in-year biomass variation within the beds is minimal, with average leaf-length changing a limited degree by year per bed.

Fal & Helford: Comparisons of Estimated Extent, 2015 verses 2021

Table 7: Comparisons of the total extent of the seagrass beds within Fal & Helford SAC.
Table shows calculated coverage, per bed in m^2, in 2015 and 2021, difference between years, and the percentage change. Total difference as loss (minus) or gain (plus) in SAC is provided at base of table. Lack of baseline data denoted *. Beds where some kind of active modification system from ReMEDIES was applied is denoted by +

Bed name	area 2015 (m^2)	area 2021 (m^2)	Difference (y_2 - y_1)	$\%C_e$
Polgwidden Cove (Durgan)+	153852	279828	**+125976**	82
Amsterdam Pont to Carricknath Point	182292	255930	**+73638**	40
Bosahan	36631	64429	**+27798**	76

Bed name	area 2015 (m^2)	area 2021 (m^2)	Difference (y_2 - y_1)	$\%C_e$
St. Mawes Harbour	41932	68979	**+27048**	65
Porthhallow	-	17820	**17820**	*
Flushing	26309	37672	**+11363**	43
St. Mawes Bank	68386	78055	**+9669**	14
Parbean Cove	13851	16598	**+2747**	20
Gyllyngvase	21525	4214	**-17311**	-80
Penarrow Point	96036	82197	**-13840**	-14
Swanpool	66310	13816	**-52495**	-79
Passage Cove	90463	11208	**-79255**	-88
		Total (m^2) =	**+133159**	
		Total (Ha) =	**+13**	
	Without Porthhallow	**Total (m^2) =**	**+115339**	
		Total (Ha) =	**+12**	

The Fal and Helford, where VNAZ and community engagement for the restoration of seagrass beds was applied, showed no significant difference across all beds between years (Wilcoxon test: V = 26, df = 23, *p-value* = 0.339). New beds were monitored by 2021, bringing the total estimated gain in seagrass bed extent in the SAC above 12 Ha. Figure 35 shows that estimated extent changes in certain beds have been a factor in the total extent increase, as seen in by the quartile distributions between years. Average bed size increased from 7 Ha (72508 ± 54770 m^2) to 8 Ha (78023 ± 92974 m^2), between the 2015 to 2021 periods. Figure 35 shows these differences more descriptively; the pattern of the boxplots suggests the size of most of the beds within the SAC are within a range of each other with the differences between the median size of seagrass beds in the SAC being negligible, and the two outliers to this of Polgwidden Cove (Durgan) and Amsterdam Pont to Carricknath Point by 2021.

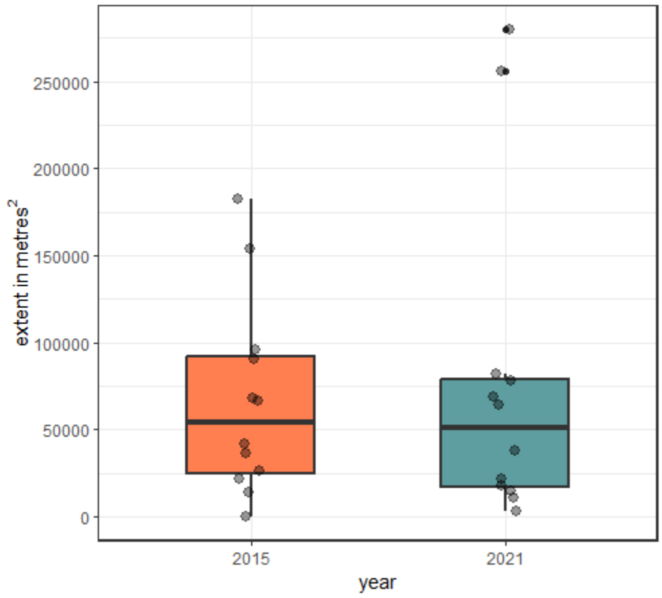

Figure 33: Boxplot of estimated seagrass extent at Fal and Helford SAC.

Carrick Roads Beds showed no significant difference between years (Wilcoxon test: *V* = 30, *df* = 19, *p-value* = 0.846). Figure 38 (C), page 41, shows coverage is comparable between years, with estimated total extent for Carrick Roads Beds equalling 16 Ha (164423 m^2) in 2015 and 16 Ha (160252 m^2) in 2021, indicating a slight drop in extent overall in the area.

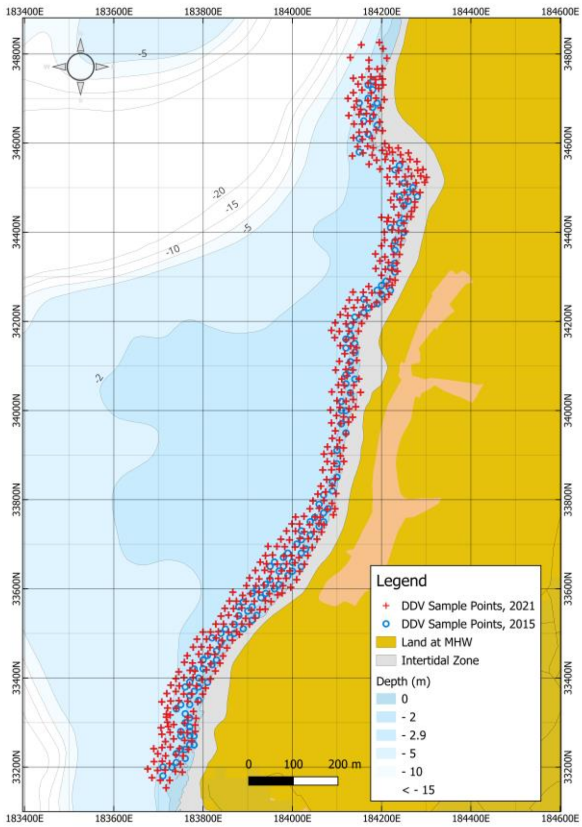

Figure 34: Map of sampling points at St. Mawes Bank, part of Carrick Roads Beds, 2015 and 2021.

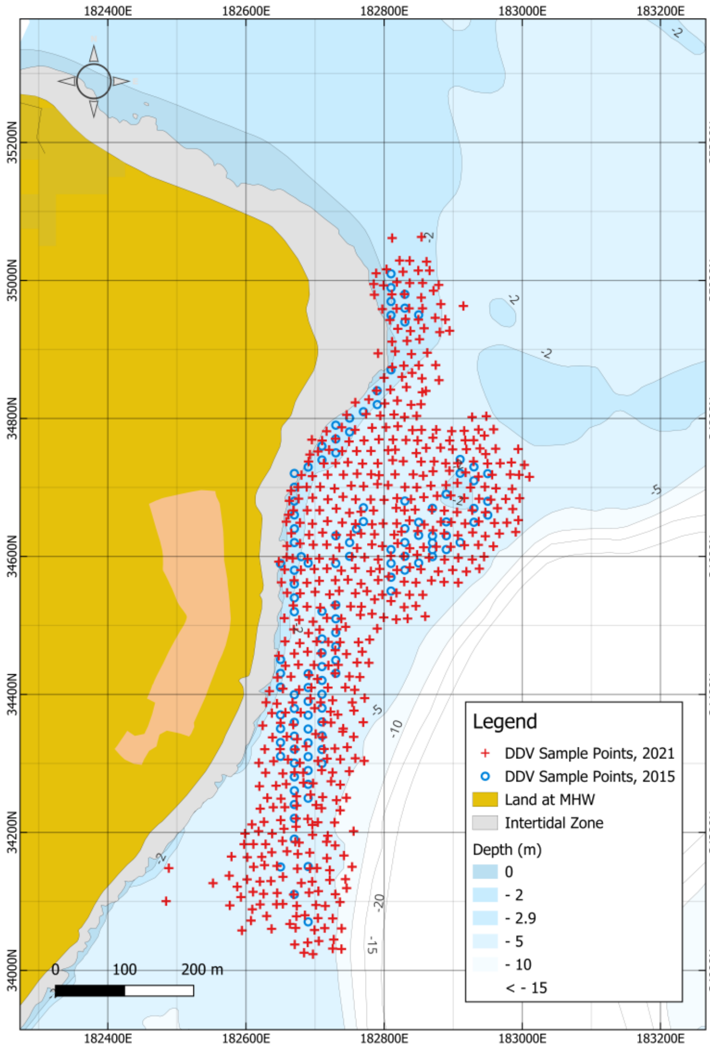

Figure 35: Map of sampling points at Penarrow Point, part of Carrick Roads Beds, 2015 and 2021.

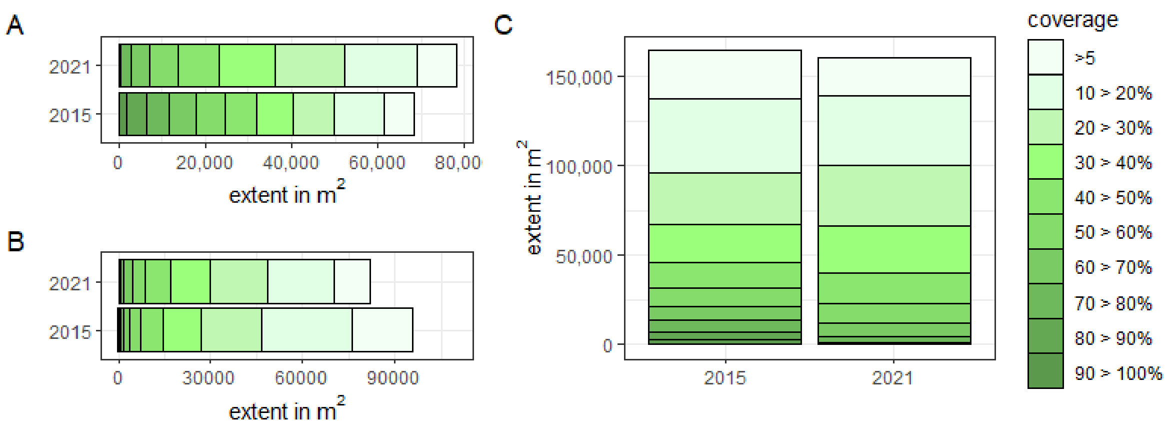

Figure 36: Extent per percentage coverage for St. Mawes Bank (A) and Penarrow Point (B), with combined Carrick Roads (C).

Analysis of the estimated extent of density categories between St. Mawes Bank 2015 and 2021, showed no significant difference between years (Wilcoxon test: V = 20, df = 19, *p-value* = 0.492), and similarly with Penarrow Point, no significant difference was found

(Wilcoxon test: V = 30, df = 19, p-$value$ = 0.846) within the same time periods. Sampled points were almost identical in mean distance for St. Mawes Bank, with 2015 at 16.25 m mean distance between points and 2021 16.21 m, but there were major differences in the counts, with 2015 (n = 150) being larger than 2021 (n = 406). The St. Mawes Bank model's RMSE = 3.428% for 2015 was slightly larger than 2021's RMSE = 1.895%, and 2015 for St. Mawes Bank having a lower coverage (61.81 ± 31.85%) than 2021 (19.53 ± 27.69%). Penarrow Point's sampling points were distributed differently, with 2015's having a mean distance of 20.87 m and 2021 15.84 m. The difference in the coverage for Penarrow Points was also shown in the sample point data, with a notable difference in the mean between the years; 2015's (n = 120) being a lower coverage (32.10 ± 22.07%) than 2021's (n = 542) (12.44 ± 21.68%) with a RMSE = 2.204% for 2015, and RMSE = 1.518%.

Gyllyngvase and Swanpool beds were the two beds with the highest loss in the Fal & Helford SAC. Whilst no significant difference was found between the 2015 and 2021 surveys (Mann-Whitney U test: W = 20, df = 9, p-$value$ = 0.315), tabled data has shown negative trends within the estimated coverage. This is shown in figure 41, page 44, with 2015's total estimated extent of 9 Ha (87836 m^2) dropping to 2 Ha (19141 m^2) by 2021; a $\%C_e$ = -78 for Falmouth Bay.

Individually, seagrass at Swanpool showed no significant difference between modelled total bed area of > 5% coverage, though a shift towards less sparse coverage is indicated. Whilst there were large reductions in seagrass coverages estimates between 2015 and 2021 (Mann-Whitney U test: W = 16, df = 9, p-$value$ = 0.267), and for Gyllngvase (Mann-Whitney U test: W = 14, df = 10, p-$value$ = 1), these were not significant. The spatial model estimated complete losses between the 30-70% estimated seagrass coverage range for Swanpool, and 40-70% estimated seagrass coverage range for Gyllyngvase. Seagrass coverage within the 5-10% coverage range was calculated as the highest extent for Gyllngvase in 2021, with 1 Ha (10239 m^2) of estimated seagrass bed present within the is range, and for Swanpool 0.3 Ha (3436 m^2) within the 10-20% estimated coverage range. These coverage extents are estimated to make up 80% of Gyllyngvase's and 54% of Swanpool's seagrass coverage, in 2021. Inspection of the sampling points also backs up these losses, with sampling points recording the average coverage being below the > 5% to qualify as a seagrass bed, with 2015 (17.48 ± 14.96%) changing to a lower coverage in 2021 (1.59 ± 3.98%) for Swanpool, and Gyllyngvase changing from a higher coverage in 2015 (15.11 ± 13.09%) to a lower coverage by 2021 (1.21 ± 3.98%). The sampling points had a mean distance of 19.60 m in 2015 (n = 166), 36.08 m in 2021 for Swanpool (n = 93), and 18.80 m for Gyllyngvase's 2015 (n = 63) sampling points with a 34.93 m for 2021's (n = 53). The model for these spatial estimations having a RMSE = 2.428% for 2015 and a RMSE = 1.270% in 2021 for Swanpool, and RMSE = 4.850% for 2015 and a RMSE = 1.244% for Gyllyngvase. Despite 2015 for Gyllyngvase being a slightly higher model error estimation, it is still within the 10% boundaries for the coverage; suggesting suboptimal conditions with the Falmouth Bay beds, and a possible decline of seagrass in the future due to low resilience of lower density seagrass areas based on the model estimates.

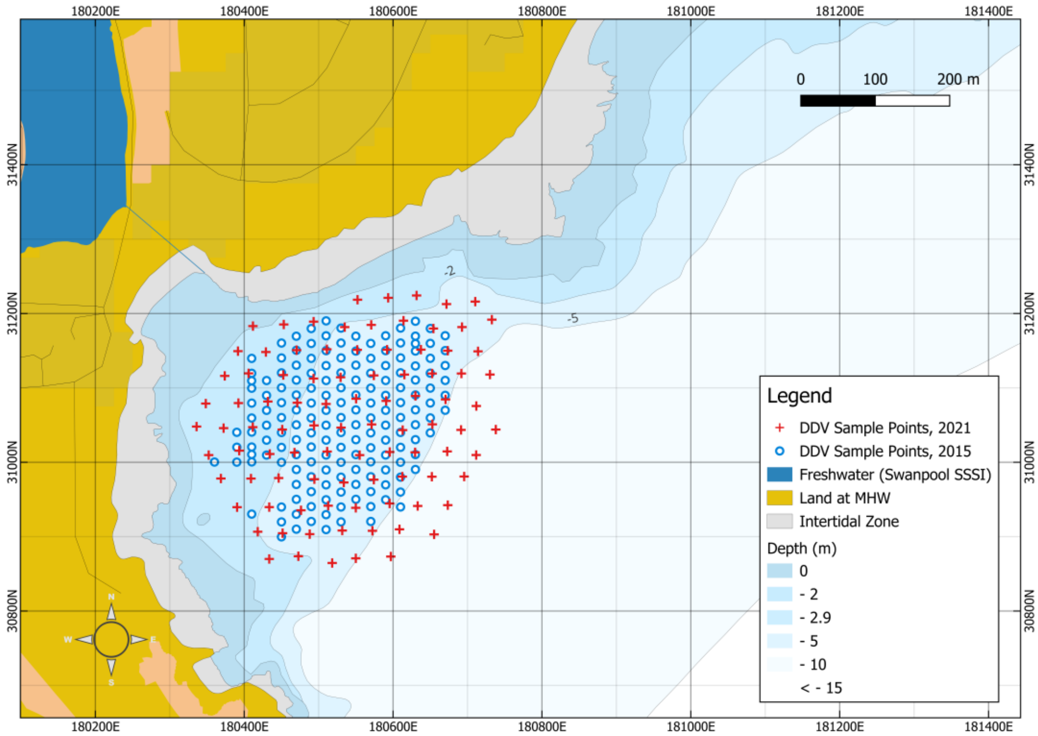

Figure 37: Map of estimated seagrass coverage at Falmouth Bay Beds, Swanpool 2015 and 2021.

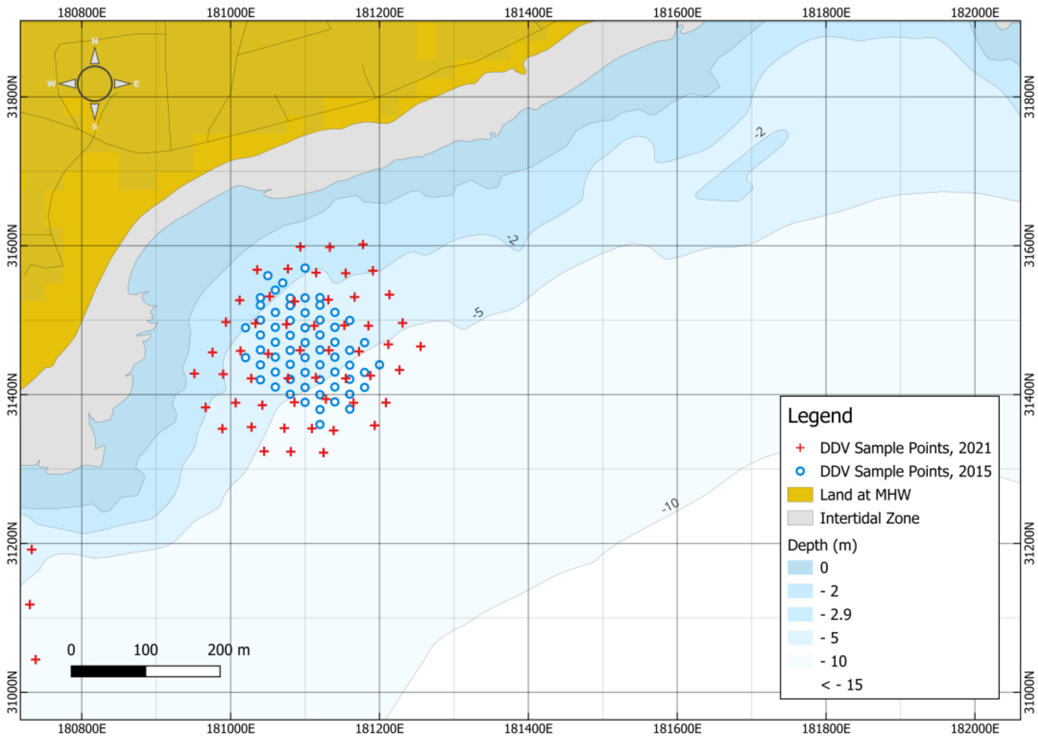

Figure 38: Map of estimated seagrass coverage at Falmouth Bay Beds, Gyllyngvase 2015 and 2021.

Figure 39: Extent per percentage coverage for Swanpool (A) and Gyllyngvase (B), with combined Falmouth Bay (C).

Whilst the spatial model did estimate the overall bed extent for the total area being 28 Ha (280947 m^2) in 2015 and 35 Ha (352316 m^2) in 2021 respectively, Helford Passage Seagrass Beds have shown no significant difference (Wilcoxon test: V = 16, df = 19, *p-value* = 0.275).

These changes are, by individual beds, more complex. Seagrass for Polgwidden Cove (Durgan) showed no significant difference (Wilcoxon test: V = 12, df = 19, *p-value* = 0.131). There were losses in the estimated 70-100% coverage range, a $\%C_E$ = -55. The 20-40% estimated coverage range in this seagrass expanded, with 3 Ha (26642 m^2) in 2015 changing to 10 Ha (104816 m^2) in 2021; a $\%C_e$= +293. A bigger contrast was noted at Passage Cove. Here, all estimated coverage in the 5-100% range declined, as shown in figure 43. Analysis showed this was significant (Wilcoxon test: V = 55, df =19, *p-value* = < 0.05). The greatest losses, being in the 60-100% coverage range, where 1 Ha (8990 m^2) in 2015, declined to 0.04 Ha (387 m^2); a $\%C_E$ = -96 in dense seagrass bed. Bosahan has seen increases in extent of denser seagrass based on modelled estimated extent. Seagrass extent estimated in the 60-100% coverage range increased from 0.6 Ha (6259 m^2) in 2015, to 2 Ha (18750 m^2) in 2021. This difference was found to be significant (Wilcoxon test: V = 0, df = 19, *p-value* = < 0.05). Helford Passage seagrass beds seem to have decreased in seagrass bed density, but not a uniform decrease across the entire area, and increases in extent more generally though mostly in lower coverage seagrass. The sampling points for the spatial model provide more clarity to this change. The sampling points had a mean distance of 19.96 m for 2015 (n = 53) and 16.73 m for 2021 (n = 211) for Polgwidden Cove (Durgan), with a spatial model RMSE = 2.465% for 2015 (69.25 ± 27.74%), and RMSE = 2.026% for 2021 (29.18 ± 41.66%). For Boshan, the mean distance was 20.74 m in 2015 (n = 75), and 30.23 m in 2021 (n = 89), with a RMSE = 2.678% in 2015 (39.52 ± 32.65%), and 2021 RMSE = 1.899% in 2021 (29.59 ± 41.91%). At Passage Cove, the spatial model had a RMSE = 3.257% (38.60 ± 25.46%) from a mean distance of 23.55 m for 2015 (n = 144), and a RMSE = 1.302% (6.16 ± 19.23%) for a mean distance of 29.87 m for 2021 (n = 56). While there is still consistent accuracy in the spatial model used for coverage and extent estimation, sampling points between the beds of Passage Cove and Polgwidden Cove (Durgan) had different survey coverage and bed labelling, because of the patchy bed network in 2015. This is further discussed in the summary.

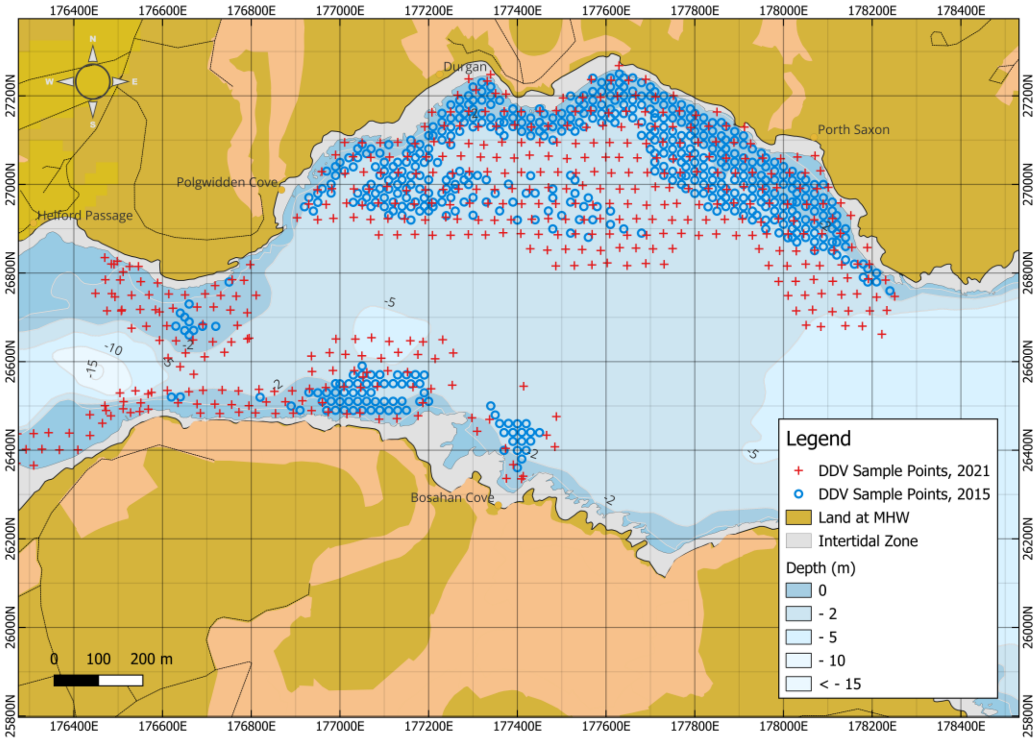

Figure 40: Map of sampling points at Helford River seagrass beds, combined from Polgwidden Cove (Durgan), Passage Cove, and Bosahan, 2015 and 2021.

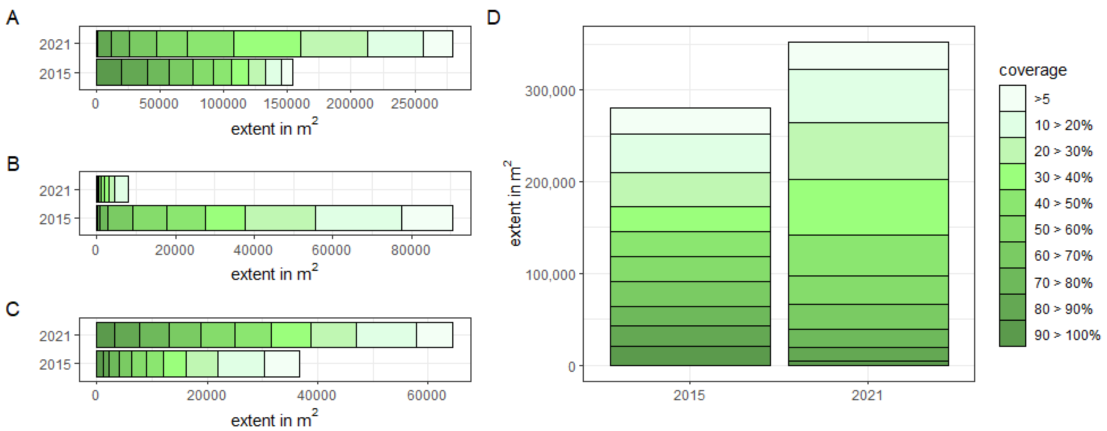

Figure 41: Extent per percentage coverage for Polgwidden Cove (Durgan) (A), Passage Cove (B), and Bosahan (C), with combined Helford Passage Seagrass Beds (D).

Penbean Cove has appeared to have become less fragmented. In 2015, the model estimated extent was largest in the 10-20% coverage range, changing to the 60-70% coverage range by 2021. In the same period, the 5-20% coverage for the spatial model coverage range of the bed dropped from 0.4 Ha (4439 m^2) in the 2015, to 0.3 Ha (2751 m^2) by 2021; a $\%C_E$ = -38. However, this change was not found to be significant (Wilcoxon test: V = 10, df = 19, p-value = 0.084). The model presented an increase from 0.04 Ha (370 m^2) to 0.2 Ha (2049 m^2) from 2015 to 2021, in estimated extent for the 80-

90% coverage range. Figure 45 shows this estimated increase in higher-density coverage, with overall extent also increasing, suggesting Penbean Cove has high shoot density and extent in 2021 compared to 2015. The sampling points for the two survey years showed certain differences. The mean distance between the sampling points was larger for 2015, being 21.00 m whilst 2015 was 17.86 m. The spatial model has a RMSE = 1.607% for 2015 (n = 36) (58.08 ± 26.40%) and RMSE = 1.252% (n = 60) (9.77 ± 26.10%). This difference in mean estimates for coverage in the sampling points is consistent with the main finds here, with 2021 presenting a lower coverage than 2021 at Penbean Cove.

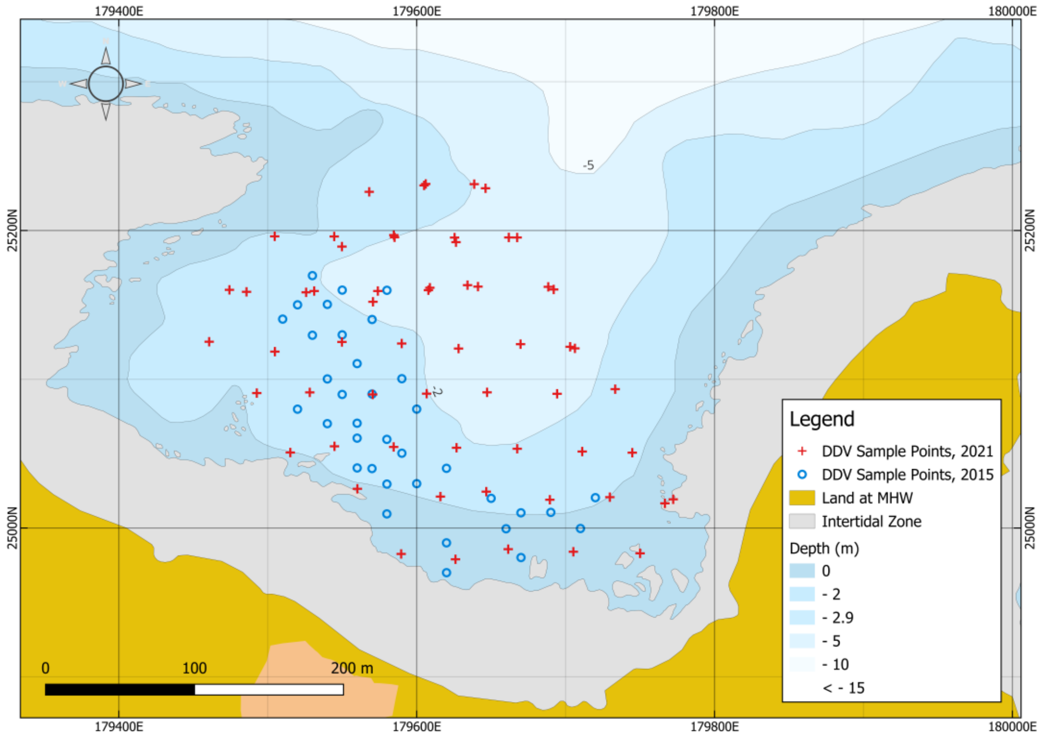

Figure 42: Map of sampling points at Penbean Cove, 2015 and 2021.

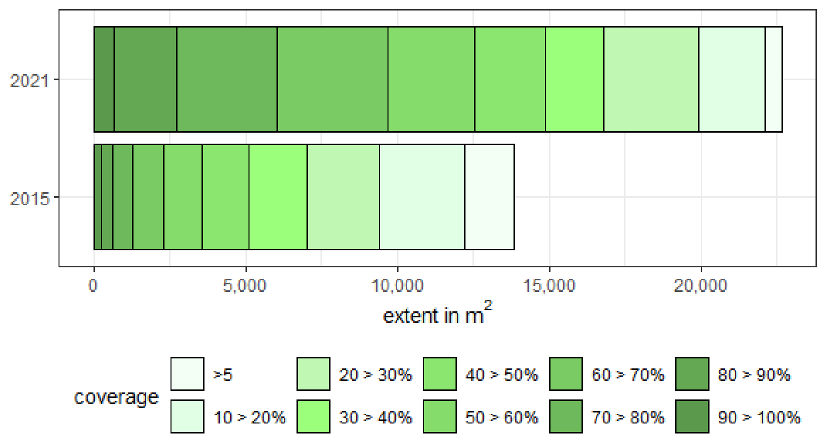

Figure 43: Extent per percentage coverage for Penbean Cove, years 2015 and 2021.

Penryn River Seagrass Beds showed a significant difference (Wilcoxon test: V = 2, df = 19, p-$value$ = < 0.05) between years, which includes an increase in estimated bed size and increase in estimated coverage. As seen in figure 47, this is also an estimated increase in higher coverage seagrass present. This bed at Flushing only saw a reduction in the estimated 20-30% coverage range: $\%C_e$ = -5. The largest extent in coverage was the 10-20% coverage range in 2015, by 2021 this had moved to 90-100% coverage range, an area of 0.7 Ha (7137 m^2). The bed range in the health 60-100% coverage range was higher in 2021 than in 2015 with a calculated increase in extent of $\%C_e$ = 90. The spatial model for the coverage estimations had various levels of accuracy but was still within the 10% range threshold. The 2015 (n = 53) RMSE = 4.607% and the 2021 (n = 211) RMSE = 1.786%, indicating the model was slightly less accurate for the 2015 estimation. This was based off a slightly larger mean distance of sampling points in 2015 of 19.96 m, and 16.73 m in 2021, and adding to the less accurate 2015 prediction from the fewer sampling points over a larger distribution. The sampling points also showed differences in the coverage, mirroring the results of the spatial model's projects of the increase in bed coverage, with 2015 (69.25 ± 27.74%) being denser than 2021's coverage (29.18 ± 41.66%), the results indicating that Flushing bed is both increasing in size as well as density of the bed.

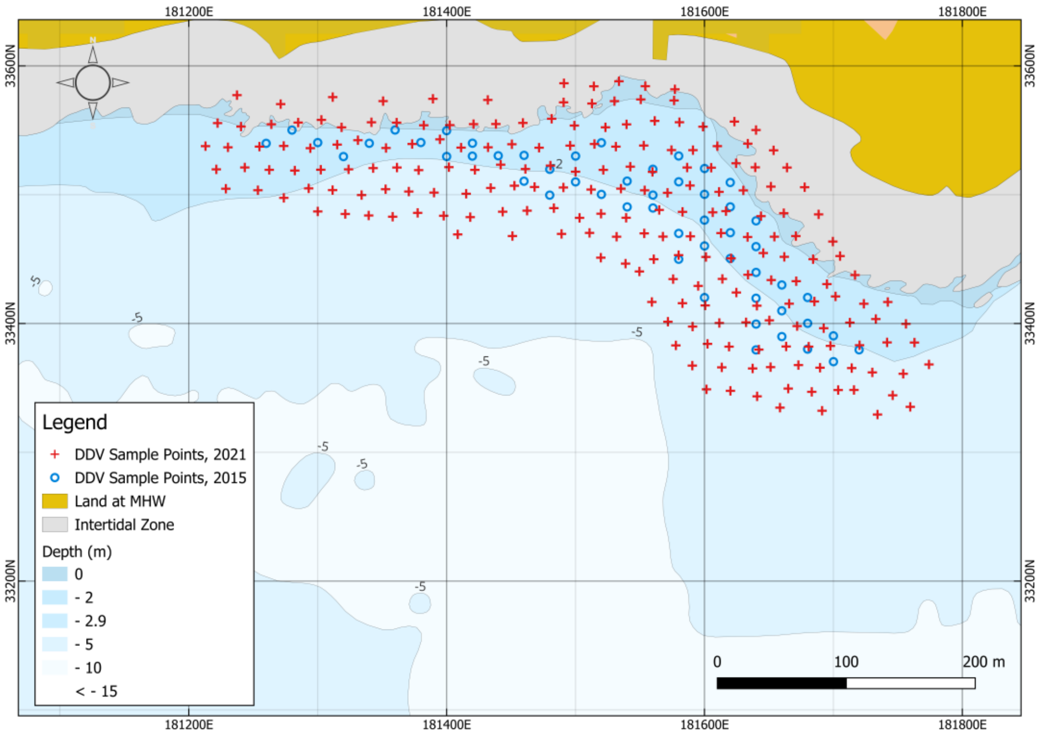

Figure 44: Map of sampling points at Penryn River Beds (Flushing), 2015 and 2021.

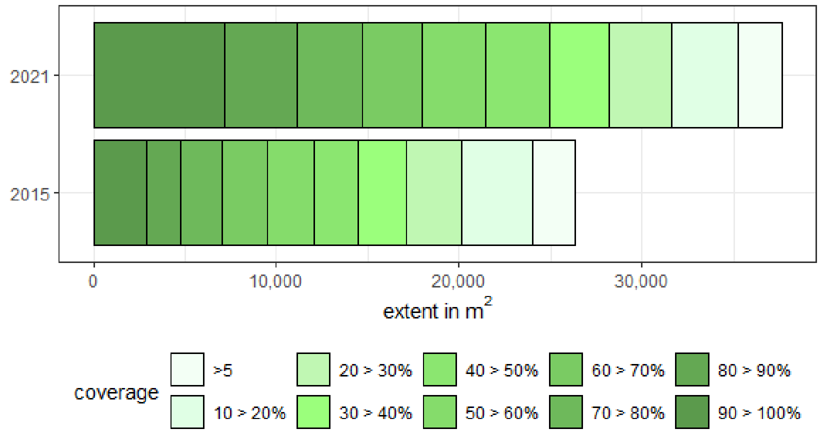

Figure 45: Extent per percentage coverage for Penryn, years 2015 and 2021.

The Percuil River Seagrass Beds are made up of St. Mawes Harbour and Amsterdam Point to Carricknath Point seagrass beds. The whole of Percuil River Seagrass Beds having an estimated extent $\%C_e = 71$, a significant difference in extent between survey years (Wilcoxon test: $V = 2$, $df = 19$, $p\text{-}value = < 0.05$). Estimated seagrass extent for the total area was highest in the 20-30% coverage range, in 2015. By the 2021 survey date, this had changed to 10-20% coverage range. The largest gains for the total beds for the Percuil River Beds has been in the 5-10% coverage range.

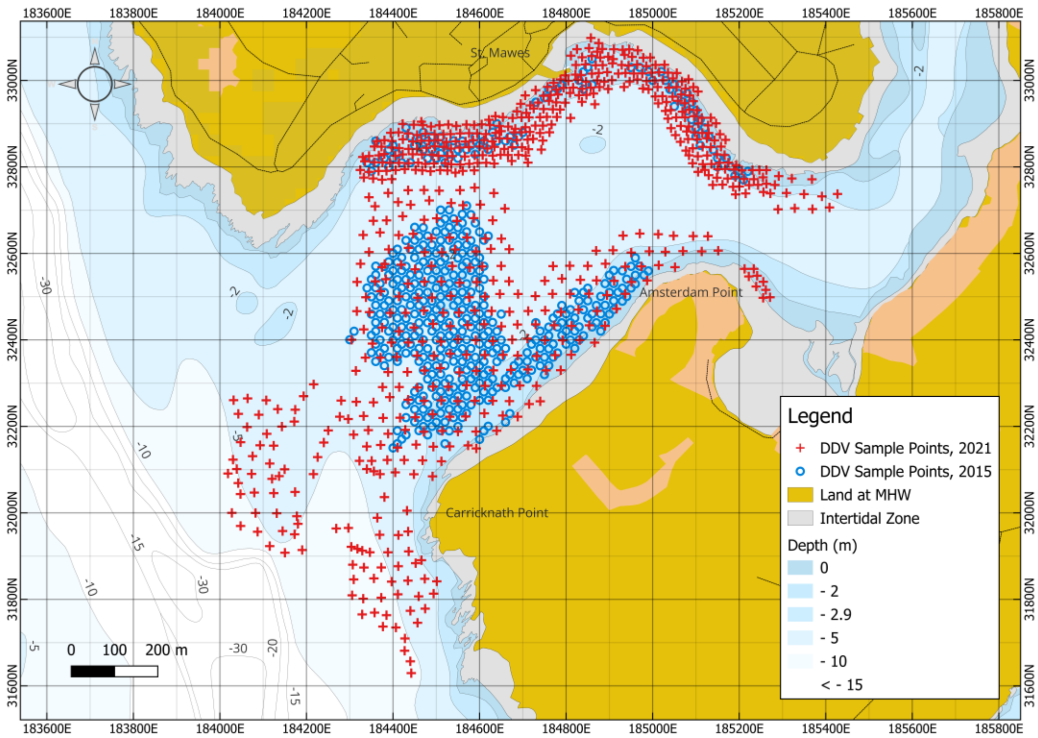

Figure 46: Map of sampling points at Percuil River Beds, 2015 and 2021.

Figure 47: Extent per percentage coverage for Amsterdam Pont to Carricknath Point (A), St. Mawes Harbour (B), with combined Percuil River Beds (C).

These changes are better considered by the individual bed characteristics. Amsterdam Point to Carricknath Point was not significant (Mann-Whitney U test: *W* = 22, *df* = 17, *p-value* = 0.122). The spatial model did estimate the 70-100% coverage range in 2015 was 3 Ha (34818 m^2), dropping to 3 Ha (27427 m^2) by 2021; a $\%C_e$ = -21. This suggests that there has been expansion in the bed area, but mostly lower coverage range, and not statistically significant amounts of coverage change have occurred. The sampling points for Amsterdam Point to Carricknath Point were also indicative of this drop in coverage, with 2015 being (50.99 ± 28.31%) than 2021's coverage (15.65 ± 26.56%). The sampling areas for these points also being different, with 2015 (*n* = 394) mean distance of 16.55 m being shorter than 2021's (*n* = 326) mean distance of 33.76 m. The spatial model for this bed giving a RMSE = 3.485% for 2015, and RMSE = 1.906% for 2021. For St. Mawes Harbour, a significant difference was found (Wilcoxon test: *V* = 0, *df* = 19, *p-value* = < 0.05) by estimated extent difference between survey years, with a model prediction of increases of 0.3 Ha (2687 m^2) within survey periods. Within this bed, estimated extent has increased across all the coverage ranges, with sizable increases in the 80-100% and 30-60% coverage range. The largest extent for both 2015 and 2021, was estimated in the 10-30% coverage range. Modelled extent of denser areas also changed, with the 60-100% coverage range showing an estimated increase in $\%C_e$ = 190. Whilst the sample sizes differed considerably, with 2015 (*n* = 91) having a much smaller number of points with a slightly larger mean distance of 18.63 m, to 2021's (*n* = 339) points and mean distribution of 17.08 m, the spatial model gave a RMSE = 7.712% (46.84 ± 26.81%) for 2015 which was the largest error for the SAC, whilst 2015 RMSE = 1.637% (24.07 ± 34.98%).

Together, as seen in figure 49, above, the model estimates suggest Percuil River Seagrass Beds had a greater extent in 2021, with St. Mawes Harbour having increased in density beds whist Amsterdam Pont to Carricknath Point coverage being less dense by comparison.

Fal & Helford, Infection Burden (non-native species and pathogens)

The Fal and Helford survey data for 2015 showing infection burden in the half- to highly-infected ranked categories of the sampled leaves was highest in St. Mawes Harbour and the Polgwidden Cove (Durgan) seagrass bed, which is part of the Percuil River Seagrass Bed system. There was minor evidence of infection for Falmouth in 2015, with the two largest St. Mawes Harbour with 3.14% of samples indicating seagrass within the half- to high-infection categories, and Polgwidden Cove (Durgan) 1.28%. At Penarrow Point no seagrass leaf samples were within the half- to highly-infected category, and having no leaves found with minimal infection. There was also a significance in association of presence absence to beds of leaf darkening (Pearson's Chi-squared test: $\chi^2 = 23.593$, df = 3, *p-value* = < 0.05), with crosstabulation indicating that presence and absence of infection is dependent on sites in the survey period.

Table 8: Contingency table for 2015 seagrass infection presence/absence at Fal and Helford SAC.

Proportions of counts of infection per leaf in parenthesis**.**

Seagrass bed	Presence	Absence	Totals
Polgwidden Cove (Durgan)	178 (0.163)	916 (0.837)	**1094**
Penarrow Point	9 (0.110)	73 (0.890)	**82**
St. Mawes Bank	315 (0.222)	1103 (0.778)	**1418**
St. Mawes Harbour	66 (0.259)	189 (0.741)	**255**
Totals	**568**	**2281**	

Focusing on the broader categories of infection burden, as seen in figure 50, the percentage of infection in the survey areas shows low levels of infection. No significant difference was found between infection burden between seagrass bed locations (Kruskal-Wallis test: $\chi^2 = 1.729$, *df* = 3, *p-value* = 0.631), suggesting a uniform trend in *Labyrinthula* sp. infection. The results suggest that for 2015, infection in the survey locations is proportionally low and consistently low across the habitats.

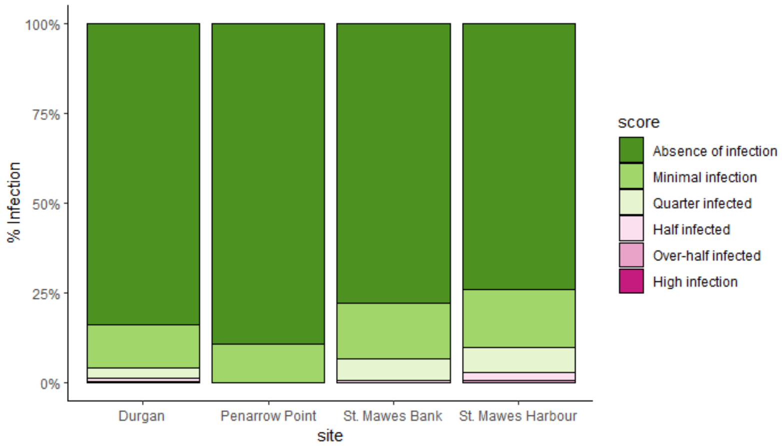

Figure 48: Stacked bar chart of levels of infection (1 low to 5 high) at Fal and Helford seagrass beds for 2015.

The 2021 data show a contrast, which is observable in the ranked infection burden results in figure 51. The contingency table results were found to be significant (Pearson's Chi-squared test: χ^2 = 14.16, *df* = 4, *p-value* = < 0.05). With the significant association between seagrass beds and infection burden in 2021, this indicates that this trend is consistent between the beds throughout Fal and Helford SAC at the time of the survey.

Table 9: Contingency table for 2021 seagrass infection presence/absence at Fal and Helford SAC.
Proportions of counts of infection per leaf in parenthesis.

Seagrass bed	Presence	Absence	Totals
Polgwidden Cove (Durgan)	1743 (0.785)	477 (0.215)	**2220**
Flushing	837 (0.739)	295 (0.261)	**1132**
Penarrow Point	686 (0.790)	182 (0.210)	**868**
St. Mawes Bank	1202 (0.751)	398 (0.249)	**1600**
St. Mawes Harbour & Carricknath Point	1384 (0.774)	404 (0.226)	**1788**
Totals	**5852**	**1756**	

Individual beds show slight differences the half- to highly infected category of the ranked infection burden scores. The samples showed that the highest amount of infection present was at St. Mawes Bank, where 2.81% of the seagrass sampled was ranked in this high-infection category. The lowest out of the seagrass beds was Penarrow Point, with 0.23% of the sample size in the higher-infection category. These marginal differences were verified by the statistical analysis, where no significant was found between the seagrass beds surveyed in 2021 (Kruskal-Wallis test: χ^2 = 0.159, *df* = 4, *p-value* = 0.997). The results for 2021 suggest that infection levels are higher in 2021 compared to 2015, and the level of infection burden in the seagrass sample areas was consistently high for all areas of the Fal and Helford SAC in 2021.

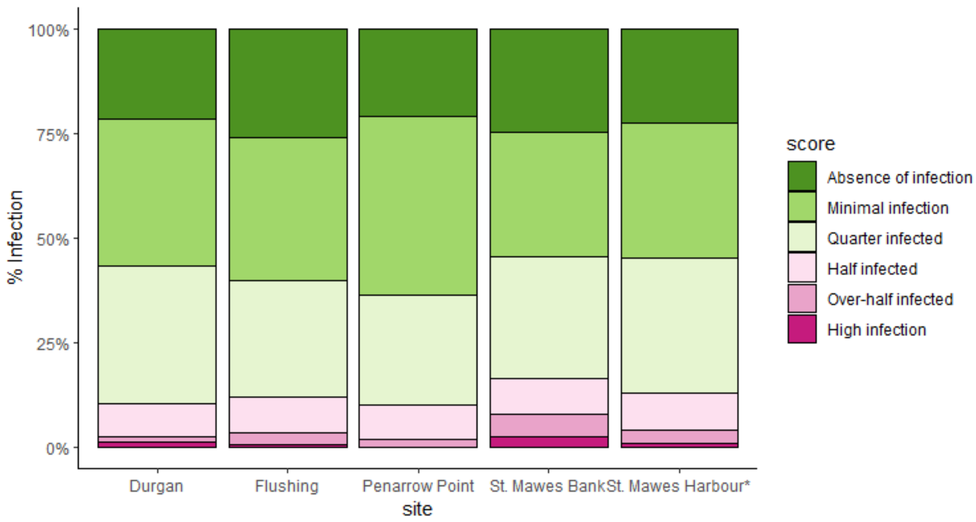

Figure 49: Stacked bar chart of levels of infection (1 low to 5 high) at Fal and Helford seagrass beds for 2021.

The 2024 data show another shift. Table 8 shows rate of infection burden being consistently lower again, like the 2015 levels. This result was significant (Pearson's Chi-squared test: χ^2 = 226.24, *df* = 4, *p-value* = < 0.05) for association between seagrass bed sites and the level of infection found in the samples.

Table 10: Contingency table for 2024 seagrass infection presence/absence at Fal and Helford SAC.
Proportions of counts of infection per leaf in parenthesis.

Seagrass bed	Presence	Absence	Total
Carricknath Point	867 (0.352)	1599 (0.648)	**2466**
Polgwidden Cove (Durgan)	475 (0.193)	1990 (0.807)	**2465**
Flushing	347 (0.256)	1010 (0.744)	**1357**
Penarrow Point	401 (0.283)	1016 (0.717)	**1417**
St. Mawes Bank	804 (0.365)	1397 (0.635)	**2201**
Total	**2894**	**7012**	

The results also indicate declines in the high level of infection burden by 2024. Flushing, with 0.29% had the highest level of high-infection. Analysis of this data, suggest there is still a consistency within infection present, however, with no significant difference found (Kruskal-Wallis test: χ^2 = 0.507, *df* = 4, *p-value* = 0.973). The combination of tests shows that levels of infection found are consistent between seagrass beds, and that infection affects the seagrass collectively across the seagrass beds within the SAC, rather than individual beds, based on the evidence available.

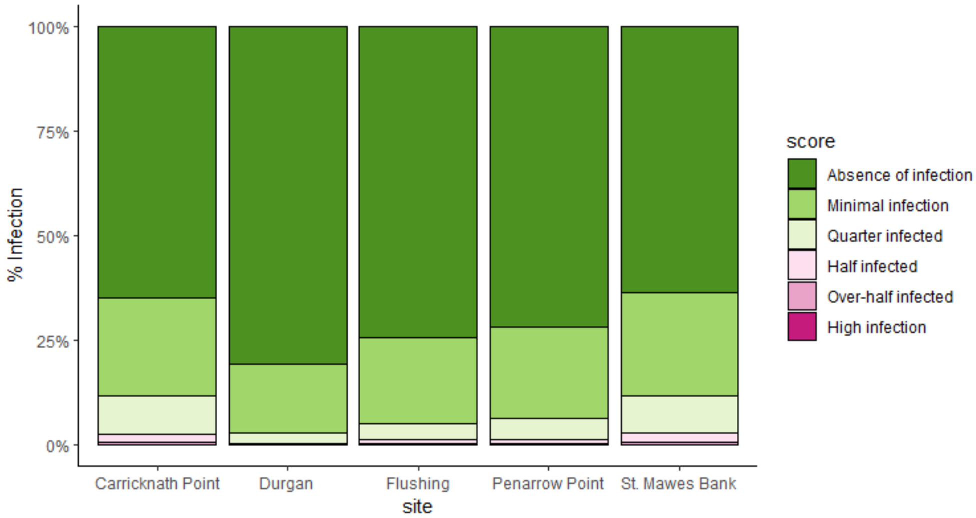

Figure 50: Stacked bar chart of levels of infection (1 low to 5 high) at Fal and Helford seagrass beds for 2024.

Comparisons between the years of the surveys is clearer when reviewed against the previous 2015, 2021, and 2024 result already shown. Infection burden from the samples taken on the same seagrass beds across the years, shown in figure 53, shows a peak in the levels of infection in 2021, with 2015 and 2024, appearing similar.

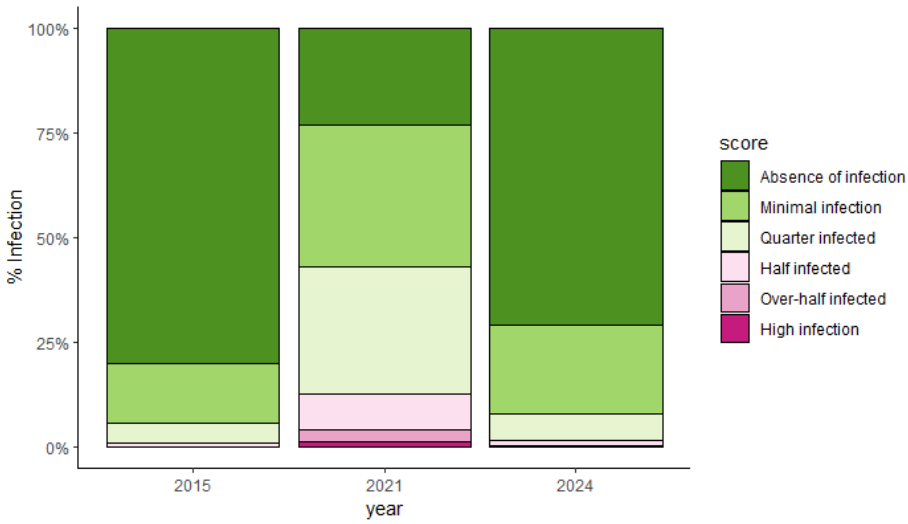

Figure 51: Stacked bar chart of the levels of infection (1 low to 5 high) for Fal & Helford SAC, from the samples taken in 2015, 2021, and 2024.

Analysis of the samples of infection burden in the total seagrass beds shows a significant difference between years (Kruskal-Wallis test: $\chi^2 = 7.546$, $df = 2$, $p\text{-}value = < 0.05$). The highest-level of high infection burden found from within the samples was 1.43% in 2021, while the lowest was 2015 with 0.04%. The difference in percentage coverage withing the ranking categories from the infection burden samples taken at Falmouth and Helford between 2015 was also significant (Dunn test: $z = -2.590$, $df = 83$, $p\text{-}value = < 0.05$). In 2015, 1.16% of samples were within the half-infected to high-infection category, and in 2021 11.51%, and whilst 2024 had 1.31%, the lowest out of the three years of survey, there was no significant difference between 2021 and 2024 (Dunn test: $z = 2.028$, $df = 83$, *p-value* = 0.128), and almost identical with no significant difference 2015 and 2021 (Dunn test*:* *z* = -0.678, *df* = 83, *p-value* = 1.000).

These results indicate health changes within the Fal and Helford SAC during the period of surveys. These effects were across all the seagrass beds, simultaneously, and when occurring mostly minimal in their infection burden impact.

Fal & Helford, Longest Leaf-Length (biomass)

Leaf length is used as a measurable proxy for biomass: with Penarrow Point showing difference in leaf-length between years (ANOVA: $F_{(2, 52.7)}$ = 46.017, *p-value* = < 0.05), with a significant difference found between 2015 and 2021 (t-test: $t_{(19, 236)}$ = NA, *p-value* = < 0.05), 2015 and 2024 (t-test: $t_{(19, 307)}$ = NA, *p-value* = < 0.05), and also 2021 and 2024's seagrass leaf-length (t-test: $t_{(236, 307)}$ = NA, *p-value* = < 0.05). This suggests inter-year variation in leaf length between beds is significant for seagrass at Penarrow Point. Polgwidden Cove (Durgan) also showed significance in the survey findings (Kruskal-Wallis test: χ^2 = 312.72, *df* = 2, *p-value* = < 0.05); with differences between 2015 and 2024 (Dunn test: *z* = -16.794, *df* = 750, *p-value* = < 0.05), 2015 and 2021 (Dunn test: *z* = -8.075, *df* = 750, *p-value* = < 0.05), and 2021 and 2024 (Dunn test: *z* = 11.309, *df* = 750, *p-value* = < 0.05), all significant. Model estimation showing the largest variation, being between 2015 and 2021.

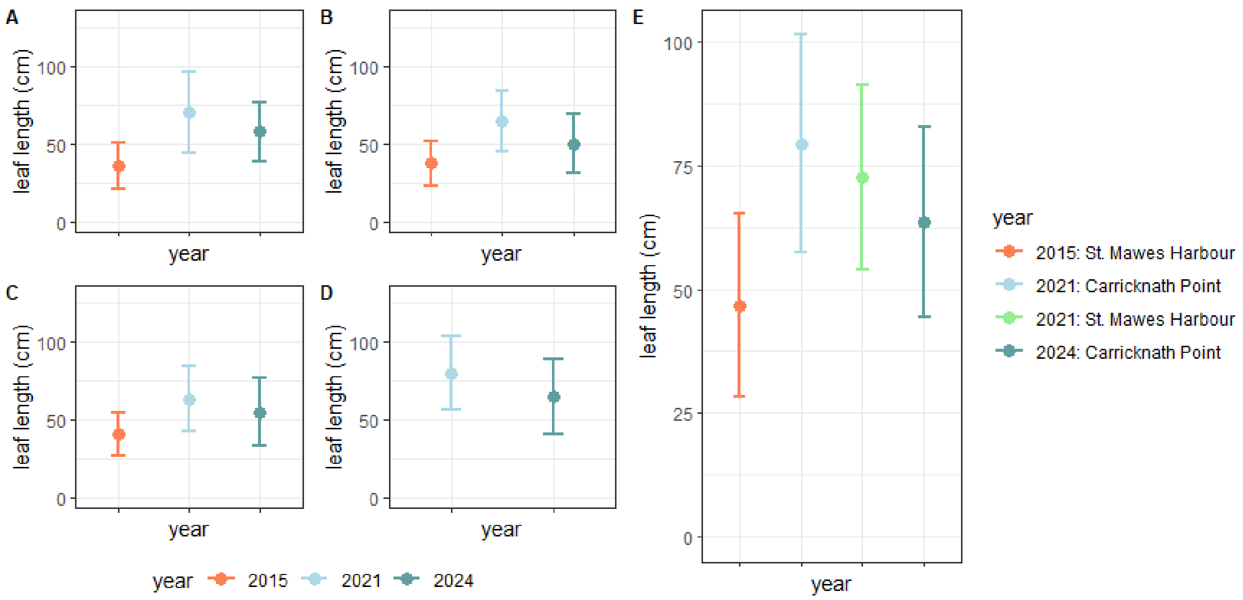

Figure 52: Leaf-length for Fal and Helford SAC for 2015, 2021, and 2024.
Plots shows Penarrow Point (A), Durgan (B), St. Mawes Bank (C), Flushing (D), and St. Mawes Harbour & Carricknath (E) mean length, with error bars of standard deviation, from the samples.

This tendency was also found in St. Mawes Bank which was also significantly different between years (ANOVA: $F_{(2, 791.39)}$ = 159.78, *p-value* = < 0.05); including 2015 and 2021 (t-test: $t_{(305, 395)}$ = -17.00, *p-value* = < 0.05), 2015 and 2024 (t-test: $t_{(305, 522)}$ = -11.431, *p-value* = < 0.05), and 2021 and 2024 (t-test: $t_{(395, 522)}$ = 5.889, *p-value* = < 0.05). Inter-year

variation was around a low in mean leaf-length in 2015 (40.92±13.92 cm), and a high in 2021 (63.53±20.97 cm).
Likewise, Flushing's seagrass bed, which was only surveyed in 2021 and 2024, was significant (t-test: t = 7.737, df = 614.07, $p\text{-value} = < 0.05$) between those years. Here the leaf-length was estimated to be greater in 2021(80.23 ± 23.96 cm) than the results showed for 2024 (65.20 ± 24.37 cm).

And finally, for St. Mawes and Carricknath, a significant difference found between years (Kruskal-Wallis test: χ^2 = 177.94, df = 3, $p\text{-value} = < 0.05$); 2015 and 2021 at St. Mawes Harbour (Dunn test: z = -7.200, df = 204, $p\text{-value} = < 0.05$), and 2021 and 2024 at Carricknath Point (Dunn test: z = 11.043, df = 924, $p\text{-value} = < 0.05$).

By mean leaf-length taken from the surveys of the seagrass, Penarrow Point has shown to be the lowest in 2015 (36.16 ± 14.56 cm), whilst the results from Flushing in 2021 were the highest, marginally higher than Carricknath Point in 2021 (79.36 ± 21.99 cm). The results showed 2021 was the highest overall, with only Flushing in 2024 (4th highest by average), and Durgan in 2021 (65.07 ± 19.42 cm) being other years that provided averages in leaf-length that were higher. Leaf length was lowest in 2015, with all the seagrass beds in the lowest categories and making up the four lowest leaf-length means. The tallest in 2015's mean leaf-lengths being St. Mawes Harbour (46.80 ± 18.36 cm). Considered together, these results show inter-year and inter-location variation in longest leaf-length, suggesting fluctuations in biomass over the survey periods but greater biomass / leaf length since 2015.

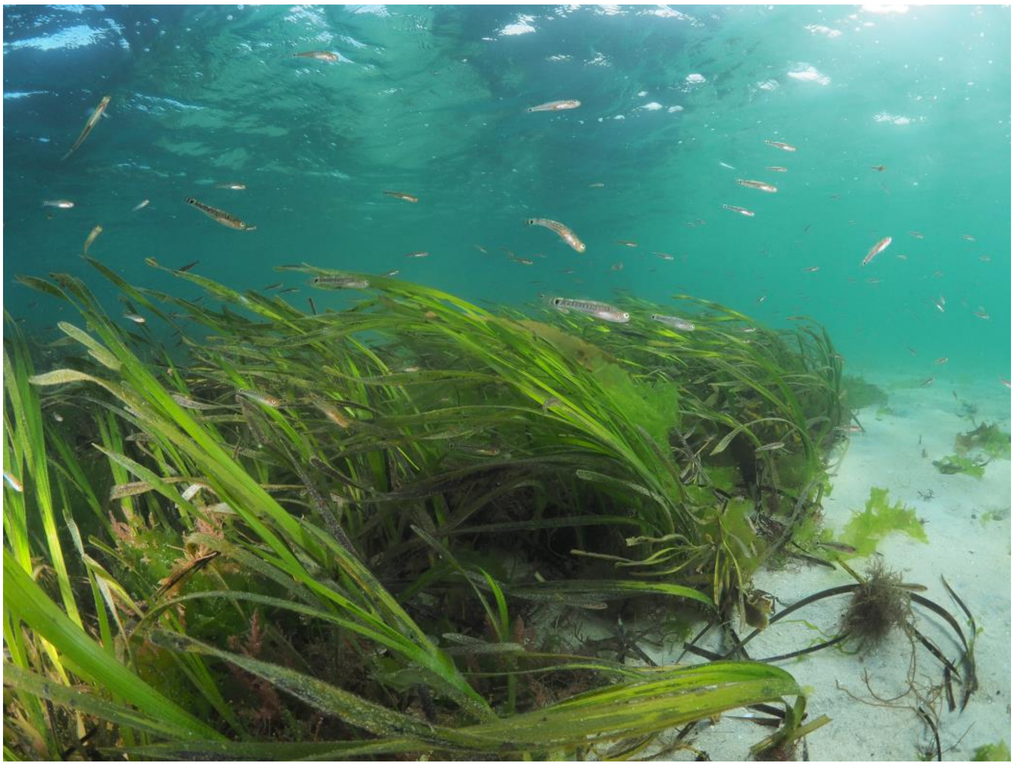

Figure 53: Juvenile fish in seagrass, Isles of Scilly
Image credit, Michiel Vos, Ocean Image Bank

Limitations and Quality Control

This study was presented with some constraints which impacted some of the methods, and therefore the results. Some limitations for ReMEDIES included the impacts of Covid-19 in 2020 and 2021 on monitoring schedules, leading to some gaps and delays in planned monitoring. Changes in restoration plans due to Covid-19 and limited accurate positioning data limited the ability to direct monitoring by the EA or Natural England dive team to precise areas of restoration at Jennycliff.

The goals of the study were to compare the available seagrass ecological metrics by years before and (as close to) end of ReMEDIES actions using the best available evidence. The current standard for measuring the extent of seagrass beds is to use hydro-acoustic methods, which includes data obtained by echosounder and side-scan sonar instruments which have been demonstrated to provide the accuracy required (Helminen and others, 2019: Muhamad and others, 2021: Ford and others, 2022), followed by (spline) interpolation of sampling points to create the layers representing the spatial distribution of seagrass (Nielsen and others, 2023).

Hydro-acoustic data was not available for all the years for all the SACs, only the sampling point data. As a result, the method applied here was to use a spatial model providing a contouring the differing extents by the sampling points only. Whilst this method has been used to spatially model seagrass in similar studies (Bunker & Green, 2019: Cai and others, 2024), it is unlikely to provide the same level of accuracy towards estimated extent that a combination of hydro-acoustic and sampling point derived layers would achieve. This is because the contour model only used the point data exclusively, so boundary to extent is less definitive over distances. As the model only uses sampling points to estimate the layers, differences in the distance between sampling points is likely to affect model precision too.

An overriding factor that affected the extent estimation specifically, was the lack of control sites. There were no control sites for 2 reasons:

1) As the environmental drivers of bed condition are likely to be include localised, site-specific variability would have required the monitoring of a seagrass bed close enough to represent similar conditions, when all local sites were part of the SAC suite and ReMEDIES project actions.
2) The timing of EA surveys is not consistent with each seagrass bed, and such consistently timed monitoring was not possible within the timeframe of the ReMEDIES project. With a longer duration, and had the project not been subject to shifts in planning due to Covid-19, this could have been considered. Regularly timed monitoring and an appropriate control site would have increased the certainty in the success of the project outcomes.

Furthermore, differences in sampling effort have caused some difficulties for result interpretation. For example, sampling area in EA-DDV for Cawsand Bay in 2018 and 2023 differ greatly. The 2018 EA survey covering an area of 208865 m^2 (21 Ha), whilst the 2023 covered an area of 350157 m^2 (35 Ha), including a higher number of sampling points in 2023. Complicated further by the buffer zone (outer edge of EA-DDV sampling points) in 2018 also containing samples of > 30% of seagrass at the northern point of the bed. The timings between surveys also provide challenges. Toombs Rock comparisons, for example, uses the EA-DDV field survey data between 04/07/2018 and 03/09/2023. This is also complicating the interpretation of scores of leaf infection and biomass. For example, NE-DS for Falmouth all took place over an almost three-month window from start to finish, with the earliest start in 22/06/2015 and the latest finish on 17/09/2021. These effort inconsistencies make the temporal-patterns of the seagrass, unclear, meaning judgments on extent increases/decreases and changes in seagrass condition in this report are based solely on direct evidence and measurements from the models, and to an extent, dismiss any uncertainties with sampling effort.

Both the mean distance between sampling points, and the root mean square error (RMSE) between the model and the sampling point data (as % cover of subtidal seagrass found) therefore is presented. Results of the error estimation process shows that the model error sits within the 10% threshold between the different estimated layers, as such is unlikely to have significantly overestimated coverage measurements within each modelled layer.

There is also the addition of a supplementary section (Addendum, page 62) which provides the error estimation and analysis for model validation for the total extent estimations. This is based on the full EA hydro-acoustic/echosounder results that were only fully available after the report data gathering and main report method creation and analysis process had occured. However, it is now supplied to give a more complete picture of the model validity alongside the RMSE for the coverage.

Results of the model validation process showed no significant differences found between the report model performance to the validation model, though Fal and Helford 2021’s was a weaker estimate than the other three extent estimations.

Together with the RMSE results for the coverage and the total extent cross-validation; there is a measure of confidence towards the estimated coverage within the models, with reasonable confidence in the estimated total extent of each bed, with a lower confidence towards the total estimated extent for Fal and Helford SAC in 2021.

Comparisons between the data presented were conducted using normal statistical processes. However, many of the statical comparisons were organised under nonparametric test including rank-based tests of small sampling populations which can increase the chances of type I and type II errors occurring (Knudson & Lindsey, 2014). Whilst machine learning based models (for example, linear models) would have been able to achieve better sensitivity, this was beyond the scope of this study. This is further discussed in the Recommendations section (page 61) of this report.

Summary

This report has synthesised data from multiple surveys and methods into a single document and provides insights into the status of the seagrass beds within two SACs within the EU LIFE recreation ReMEDIES project. The report's aim was to compare the seagrass status (extent, health and biomass) at the start of the ReMEDIES project to the status at the end using the best available evidence.

Spatial models used were to quantify the seagrass extent at different coverages above 5% (the minimum coverage set by OSPAR to constitute a seagrass bed), to assess changes at scale within the different beds and the larger area making the SACs of Plymouth Sound and Estuaries alongside Fal & Helford. The report also evaluated the biomass differences within the beds, and the darkening of leaves which is commonly used as a guidance for infection of *Labyrinthula* spp. Some of these results show increases in seagrass bed extent over the sampling periods, and decrease in seagrass bed extent over the sampling periods, but this overarching synthesis of the outcomes of conservation work by the ReMEDIES teams provides a status, based on the evidence available, offering a good understanding of the biological perimeters around the restoration efforts attempted at these sites that have worked towards improving the seagrass habitats in the protected sites.

Both the SACs have seen improvements in the modelled estimations of extent of total seagrass present. The largest being the Fal and Helford's estimated increase of 13 Ha, more than Plymouth Sound and Estuaries increase of 3 Ha.

The drivers for the Fal and Helford's large increase is difficult to assess and requires a lot of speculation. Much of this is likely a result of natural variation, and the lack of temporal analysis over the 6-year time gap here, makes this hard to dismiss. Polgwidden Cove (Durgan) and Bosahan, both within the Helford Passage seagrass bed complex, and St. Mawes Harbour seagrass bed, make up the largest contributions to this extent change. A study by Curtis (2015), found that the seagrass beds in St. Mawes Harbour had not changed significantly since 2012, and there had not been a decline in Polgwidden Cove (Durgan) since 2000. Suggesting a relatively stable seagrass state in these beds till at-least 2015. Nonetheless, the 2015 Curtis study placed St. Mawes Harbour beds (by amalgamating St. Mawes Harbour and Amsterdam Pont to Carricknath Point) at 23 Ha. This is consistent with the report spatial model of 22 Ha. Curtis also found Helford Passage beds (by amalgamating Passage Cove, Bosahan, and Polgwidden Cove (Durgan)) at 28 Ha for 2015. Which is also consistent with the spatial model used in this report, which estimated the same beds at 28 Ha. By then comparing to a focused study by Jenkin and others (2021), who reported the seagrass at St. Mawes at 30 Ha (the study amalgamating St. Mawes Harbour, Amsterdam Point to Carricknath Point and West of Carricknath seagrass beds). Plus, Helford Passage at 35 Ha (again amalgamating Bosahan, Passage Cove, and Polgwidden Cove (Durgan)), which, compared to this report's estimation of 32 Ha for St. Mawes and 35 Ha for the Helford Passage by the same amalgamations in 2021; all suggests this increase of 13 Ha is correct and the full drivers for this change would be valuable to identify. However, whilst natural variation or un-

identified environmental factors might be responsible for this change, it is also reasonable to point to VNAZ being installed at Greebe Beach next to Durgan in the Helford and seagrass markers off Carricknath Point installed by Falmouth Harbour Commissioners next to St. Mawes Harbour, giving certain protection.

For Plymouth Sound and Estuaries, much of this increase was from a single bed, Cawsand Bay, which had different levels of sampling coverage area between the two years of data collection used for this assessment. Another assessment of the seagrass at Cawsand Bay by Curtis (2012), which found a 12 Ha bed in Cawsand Bay – which is keeping within the baseline (and an increase temporally) of the 2018 data used in this report of 18 Ha - suggests that the change found by 2023 is correct. But this is not conclusive, as 2012 and 2018 might have not measured northern areas of seagrass which was sparse and below the > 5% coverage. It is clear, however, that this offers a consideration towards future mapping techniques, and the possibility that more extensive sampling in known seagrass localities is required, and sampling design considers beds that may be patchy and heavily fragmented, thus blurring the lines between where beds extent to at very low seagrass coverage levels. An optimal solution where possible being annual/biannual seagrass mapping that would account for minor changes in seagrass bed, or surveying areas identified by Habitat Suitability Mapping as being suitable for seagrass around known seagrass bed locations.

Areas where this increase has been most obvious, is in the areas of Percuil and Helford River seagrass beds in the Fal and Helford, and Cawsand and Firestine Bay in the Plymouth Sound and Estuaries SAC. What joins these sites together, is three out of the four are seagrass beds where either VNAZ or AMS systems, or both, are deployed as a seagrass protection tool. It has been reported that boat owners will comply where encouraged to avoid seagrass bed locations at points of the year (Twigger-Ross and others, 2021). It has also been shown that reducing environmental impacts was the main driver by harbour and port authorities for installing AMS systems (Stainthorp, 2024), which does provide mixed benefits (Solandt, 2022). Indeed, as ecological benefits from mooring impact mitigation techniques have been demonstrated externally (Luff and others, 2019: Unsworth and others, 2022: Seto and others, 2024); these results give good indications that the use of mooring management is an effective tool for seagrass recovery and restoration. Especially as the fourth site Firestone Bay is a designated swimming area, with buoys marking out the safe swimming zone, protecting the seagrass bed within it from engine-powered watercraft.

Some negative issues have been identified - most seagrass beds showed no significant difference in their coverage composition, the change in estimated increased extent of seagrass beds occurred with lower density coverage in some locations. This was especially notable in the beds within Plymouth Sound and Estuaries. For example, seagrass beds at Jennycliff (south) and Drakes Island, and possibly Cawsand Bay. And, very starkly at the Fal and Helford, for Swanpool and Gyllngvase seagrass. There is also evidence that suggests fragmentation is also occurring with these beds. Patch-scale effects such as fragmentation and increased edge effects are signs of seagrass degradation (Yarnall and others, 2022), which also affect seagrass-associated biological communities (Macreadie and others, 2009), highlighting the broader risks coupled with seagrass habitat reduction. Some of these changes are possible due to (natural) environmental factors. Weather and storm intensity has been shown to degrade seagrass beds (Oprandi and others, 2020), and the two SACs are at the in the southwest which receives prevailing maritime winds which brings in increasing rainfall and wind regulated by sea temperature variations (Phillips & McGregor, 2001), sea temperature that is increasing through Climate Change (Garcia-Soto and others, 2021).

Further assessment of the health of the beds is added in the infection burden data taken from the NE dive surveys. Plymouth Sound and Estuaries, from the data, has seemed to have a consistent but low-level (by ratio) volume of infection burden over the years surveyed. Though Drakes Island, along with loss of extent and fragmentation already noted, showed the largest amount of infection in Plymouth Sound and Estuaries; implying that Drakes Island is a 'at-risk' bed within the SAC. Fal and Helford SAC showed an amplified level of infection burden in 2021 that had reduced when monitored in 2024. And these were infection burdens that were uniform across all the beds surveyed, simultaneously rising and dropping in the volume of infection burden across the seagrass beds over time. The implications and causes of this are not clear but could also be attributed to natural variation in biological patterns related to the darkening or indeed related to localised nutrient loads and plant responses to turbidity changes and epifauna/epifloral dynamics. It is also important to consider the timings of the survey, and how this may have affected the darken stages and so the darkening levels recorded. The coverage data from the dive surveys which could indicate the density dependence to the process was not inspected to understand this better, as this was outside the scope of this report. But this does offer a knowledge gap that could be inspected in the future. The 'wasting disease' of subtidal seagrass has been attributed to *Labyrinthula* infection, and whilst recovery from infection does occur with beds improving after being affected (Godet and others, 2008), the disease was increasing throughout the world (Sullivan and others, 2013) but still represents an ongoing concern for seagrass within the SACs.

Biomass changes for the seagrass at the sites was also site-specific. Plymouth Sound and Estuaries showed minor variation between study periods in longest leaf-length, whilst the Fal and Helford showed variation. Caution should be applied when translating these results as the transect within-bed location and sample points for the observations were not factored within the analysis, and percentage cover and leaf-length are used as a proxy for biomass. On face-value, this does suggest differences in the growth rate of the two sites seagrass beds. This could be explained by growth related variations from localised physical factors, such as exposure to storms, turbidity, and water quality. If observing these as total biomasses per year, and as site-independent trends, the pattern could follow sea-surface temperatures or large-scale atmospheric pressure systems renowned as phenological mechanisms for marine productivity such as North Atlantic Oscillation (NAO). The effects from environmental variation could be an underling factor, as this has been shown to affect seagrass composition (Alonso Aller and others, 2019), which is also put into perspective through Climate Change.

Recommendations

Tools for conservation management of subtidal seagrass, as shown in this report, are varied and versatile, with no one single solution to achieve habitat recovery at the forefront. Whilst combining techniques has been indicated to be important, application of techniques is site specific and requiring in-depth prior understanding of the seagrass condition and pressures facing it. Pressure reduction, as demonstrated by some level of recovery in areas where VNAZ was implemented, is a vital tool for seagrass recovery. Pressure reduction allows natural recovery to occur and may allow potential active restoration to achieve greater success.

Obtaining evidence for seagrass condition by timely, consistent, and broader monitoring, would also be a beneficial improvement. Consideration towards the use of future mapping techniques and the possibility that more extensive sampling in known seagrass localities is required. Sampling design should consider beds that may be patchy, heavily fragmented, or very low coverage at their periphery, to monitor buffer zones of very sparse seagrass which may recover or expand.

An optimal solution for seagrass stocktaking could be:

(1) targeted annual/biannual seagrass mapping that would account for environmentally driven and temporal changes in seagrass bed composition.
(2) further study and focus on identifying any responsible environmental drivers of seagrass community dynamics and supporting ecosystem function, particularly the substrate ecology and microbiology. Investigation is required to establish 'good quality' environmental parameters and its role in supporting seagrass health and resilience.
(3) ensure areas where Habitat Suitability Mapping has identified as being suitable for seagrass and are on the periphery of known seagrass bed locations, are systematically checked and recorded for seagrass absence/presence during dedicated seagrass surveys.
(4) implementing targets on seagrass monitoring effort by bringing in partnerships and stakeholders as demonstrated during the ReMEDIES project. These could include IFCAs, Wildlife Trusts, Seasearch, Conservation NGOs, community groups and academia.

Minimising negative pressures is the most obvious and effective way to achieve results for seagrass recovery; facilitated by better seagrass protection including improved marine environmental quality, and workable processes for identifying and excluding (where appropriate) negative human-activities. Together, these can work towards better biological robustness and ecological connectivity for seagrass and seagrass habitat communities.

Addendum

The analysis of the report's (observed) spatial model error in comparison to the EA hydro-acoustic (expected) models has been added to support interpretation of results and assessment of the outcomes of the report. The tables show the element-wise absolute difference and the element-wise absolute percent difference between the two-seagrass extent measuring approaches, which provide a single-level comparison of the difference between the two model approaches and their contribution to the total error. The table also displays the single-level deviation by a plus (+) or minus (-). The statistical difference between the two models is also provided.

Seagrass Extent Estimation, Plymouth Sound & Estuaries, 2018

The 2018 Plymouth Sound and Estuaries report model against the EA hydro-acoustic data, gave a RMSE = 1579.25 m^2 (0.2 Ha) across the full estimated extent. The EA hydro-acoustic estimated the full extent as 314357 m2 (31 Ha), with the report model giving similar estimates of 308282 m^2 (31 Ha). The seagrass beds estimate for the report model showed the most error at Jennycliff (south), with 38% for the model being underestimated from the EA hydro-acoustic model figure. The largest overestimation error by the report model was Red Cove.

Table 11: Error estimation for Plymouth Sound & Estuaries SAC, 2018

Bed	Hydro-acoustic Model (m^2)	Report Model (m^2)	Absolute difference (m^2)	Absolute % difference	Deviation
Cawsand Bay	179231	179173	58	3.58%	-
Firestone Bay	3124	3165	41	1.31%	+
Cellar Cove	49035	47532	1503	3.07%	-
Drakes Island	43339	39174	4165	9.61%	-
Red Cove	18835	19123	288	1.53%	+
Jennycliff (north)	21	29	8	0.03%	+
Jennycliff (south)	6671	6432	239	38.10%	-
Tombs Rock	14102	13654	448	3.18%	-
Totals (m^2)	314358	308282			
Total Extent (Ha)	**31**	**31**			

There was no significant difference between the report model and the EA hydro-acoustic model proportions by individual beds (G-test: G = 33.271, χ^2_{df} = 49, *p-value* = 0.958), inferred by a maximum likelihood test, and there was no significant difference between the two-model groups (Wilcoxon test: V = 8, df = 15, *p-value* = 0.195).

Seagrass Extent Estimation, Plymouth Sound & Estuaries, 2023

The error estimation for Plymouth Sound and Estuaries for 2023, shows a RMSE = 5383.50 m^2 (0.5 Ha). The hydro-acoustic EA data gave a larger estimation of the total seagrass extent, at 330221 m^2 (33 Ha), whilst the lower report model gave a total extent of 317118 m^2 (32 Ha). Cawsand Bay's report model estimate showed an underestimation in comparison to the EA hydro-acoustic model estimate of the seagrass coverage, by 13162 m^2 (1.31 Ha). Toombs Rock showed the largest amount of overestimate by the report model, with 8.61% of the bed overestimated.

Table 12: Error estimation for Plymouth Sound & Estuaries SAC, 2023

Bed	Hydro-acoustic Model (m^2)	Report Model (m^2)	Absolute difference (m^2)	Absolute % difference	Deviation
Cawsand Bay	228829	215667	13162	5.75%	-
Firestone Bay	5503	5498	5	0.09%	-
Cellar Cove	49068	49098	30	0.06%	+
Drakes Island	34349	33898	451	1.31%	-
Jennycliff (south)	4925	4760	165	3.35%	-
Tombs Rock	7547	8197	650	8.61%	+
Totals (m^2)	330221	317118			
Total Extent (Ha)	**33**	**32**			

The difference statistically in the individual bed estimates was proportionally insignificant (G-test: G = 21.501, χ^2_{df} = 25, *p-value* = 0.664). The analysis of the difference between the two-model groups also showed no significant difference (Wilcoxon test: V = 7, df = 11, *p-value* = 0.563).

Seagrass Extent Estimation, Fal & Helford, 2015

For Falmouth and Helford SAC 2018, the RMSE = 10036.25 m^2 (1 Ha) over the full model. The EA hydro-acoustic model estimating the total extent of the beds at 803903 m^2 (80 Ha), with the report estimation being consistent with this prediction at 797588 m^2 (80 Ha). Penarrow Point to Trefussis was the largest overestimation, with 21162 m^2 (2.12 Ha) of the estimate by the report model deviating from the hydro-acoustic EA model. Helford Passage, which was an amalgamation of the three individual beds of Polgwidden Cove (Durgan), Bosahan, and Passage Cove, because of bed labelling differences, showed the least overestimation by the report model of only 0.02%.

Table 13: Error estimation for Fal & Helford SAC, 2015

Bed	Hydro-acoustic Model (m^2)	Report Model (m^2)	Absolute difference (m^2)	Absolute % difference	Deviation
Helford Passage	280886	280947	61	0.02%	+
Flushing	30199	26309	3890	12.88%	-

Bed	Hydro-acoustic Model (m²)	Report Model (m²)	Absolute difference (m²)	Absolute % difference	Deviation
Swanpool	58978	66310	7332	12.43%	+
Parbean	21790	13851	7939	36.43%	-
Penarrow Point to Trefussis	74874	96036	21162	28.26%	+
St Mawes Bank	80266	68386	11880	14.80%	-
St. Mawes Harbour	233169	224224	8945	3.84%	-
Gyllngvase	23741	21525	2216	9.33%	-
Totals (m²)	803903	797588			
Total Extent (Ha)	**80**	**80**			

Maximum likelihood test showed no significant difference (G-test: G = 33.271, χ^2_{df} = 49, *p-value* = 0.958) between the two models estimate proportions. The difference in the two model approaches as independent groups showed no significant difference (Wilcoxon test: V = 13, df = 15, *p-value* = 0.547).

Seagrass Extent Estimation, Fal & Helford, 2021

For Falmouth and Helford, 2021, the RMSE = 5438.98 m² (0.5 Ha) for the full model. The EA echosounder model estimated the total extent for the year of the 2021 survey at 890439 m² (89 Ha), whilst the report model placed it at 930745 m² (93 Ha). Most of the 2021 report model results were over estimated. The contribution of the error over the full estimate extent by the report model being 40306 m² (4 Ha), with Passage Cove being the highest single-level overestimation, with the report model overestimating the bed extent by 57.93%. St. Mawes Harbour was also overestimated, by just under 1 Ha.

Table 14: Error estimation for Fal & Helford SAC, 2021

Bed	Hydro-acoustic Model (m²)	Report Model (m²)	Absolute difference (m²)	Absolute % difference	Deviation
Porthallow	19064	17820	1244	6.53%	-
Bosahan	56645	64429	7784	13.74%	+
Flushing	39026	37672	1354	3.47%	-
Swanpool	16142	13815	2327	14.42%	-
Parbean	17955	16598	1357	7.56%	-
Passage Cove	7097	11208	4111	57.93%	+
Penarrow Point to Trefussis	76216	82197	5981	7.85%	+
Polgwidden Cove	274255	279828	5573	2.03%	+
St. Mawes Bank	73072	78055	4983	6.82%	+
St. Mawes Harbour	59012	68979	9967	16.89%	+

Bed	Hydro-acoustic Model (m^2)	Report Model (m^2)	Absolute difference (m^2)	Absolute % difference	Deviation
Gyllngvase	4720	4214	506	10.72%	-
Amsterdam Point to Carricknath Point	247235	255930	8695	3.52%	+
Totals (m^2)	890439	930745			
Total Extent (Ha)	**89**	**93**			

Whilst the Fal & Helford SAC, 2021 report model was the lowest preforming model in the report, statistical analysis showed no significant difference between the bed estimations as proportions by the maximum likelihood test (G-test: G = 59.638, χ^2_{df} = 121, *p-value* = 1). The analysis on the model group data was also not at the significant-level (Wilcoxon test: V = 63, *df* = 12, *p-value* = 0.064), though this was close to α = 0.05.

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Images and Figures

Tables

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