Studland Bay Marine Conservation Zone (MCZ): Subtidal Seagrass Monitoring Survey 2021

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Foreword

Natural England commissioned this report to inform condition monitoring of the Studland Bay Marine Conservation Zone (MCZ) subtidal seagrass feature, reporting specifically on the following attributes:

- Extent and distribution.
- Distribution: presence and spatial distribution of biological communities.
- Structure: biomass.
- Structure: non-native species and pathogens.
- Structure: species composition of component communities.

The findings will be used by Natural England advisers to conduct a condition assessment of the MCZ and will also be shared with all stakeholders with an interest in the site.

Natural England commission a range of reports from external contractors to provide evidence and advice to assist us in delivering our duties. The views in this report are those of the authors and do not necessarily represent those of Natural England.

Executive summary

In July 2021 a new cycle of monitoring the subtidal seagrass (*Zostera marina*) beds in Studland Bay MCZ, in Dorset, commenced. The site was previously surveyed in 2018 by the Environment Agency to record the seagrass extent and percent cover (Green, 2018). Natural England led the 2021 survey programme following the methods developed for surveying seagrass beds in Torbay. The data collected will allow the condition of the subtidal seagrass beds to be assessed against some of the targets for this feature of the MCZ.

The survey programme (including the 2018 Environment Agency survey) was designed primarily to monitor the following attributes of the subtidal seagrass bed:

- Extent and distribution: The seagrass bed has the target to either remain stable or increase in size.
- Structure and function: The bed quality (density, overall health) and species composition of characteristic biological communities (diversity and abundance of species in the habitat) with the target to remain healthy and not deteriorate.

It also aimed to provide data to support the assessment of the spiny (long-snouted) seahorse attributes:

- Quality and quantity of the habitat available with the same targets as above for the seagrass bed attribute.
- Population size and age / sex ratios with the target to enable the population to thrive by maintaining these parameters.

The data collected in 2018 and 2021 provide the first step toward recording statistically robust and repeatable monitoring data against which future data can be compared. Historical data available from a variety of different survey programmes and organisations recorded prior to these surveys only permit qualitative comparisons and are discussed in this report.

The following key findings were determined:

- The monitoring sites selected were based on the 2018 Environment Agency survey data of bed extent and percent cover. With one exception the density and distribution (patchiness) of the seagrass at each monitoring station was broadly in line with the results of that survey.
- Where seagrass does occur in the bay, overall mean percentage cover values were in the range of 24-81%. Long, luxurious stands of seagrass provide habitat for a range of species, many of which have been recorded in previous surveys.
- Seagrass wasting disease, *Labyrinthula* sp. was observed across the sites monitored. The lowest infection scores were from the most northern site and the highest from the most southern.
- Non-native species were recorded throughout the bay and were mostly observed to be in low abundance. The exception was the tetrasporophyte phase of harpoon weed, *Asparagopsis armata*, which occurred in high numbers in some areas.
- Anchor and mooring damage to the seagrass was apparent throughout the areas monitored, leaving bare sand patches, exposed rhizomes and detached plants. Not all bare sand patches could be attributed to anchor or mooring impacts although the cause of some damage observed was unequivocal.
- No seahorses were observed during the survey, but the divers were operating outside the areas considered most likely to support them.

The following recommendations were made for future surveys:

- The planned repetition of the 2018 Environment Agency survey (Green, 2018) in 2022 and every three years thereafter will go some way toward monitoring the bed status in a comparable and robust manner and will improve the resolution in data and understanding of potential seagrass bed dynamics.
- The present survey programme does not address the issue of small-scale patchiness (<50 m resolution). Consideration could be given to the idea of surveying transects to produce and monitor patchiness ratios at selected points throughout the bed.
- Increased aerial surveys (drone) could be used to monitor anchor / mooring damage within the seagrass bed, or part thereof.
- Future surveys can be made more efficient by reducing the post-survey sample processing.
- Assigning divers to specifically collect data on the seagrass bed biological communities will enhance our understanding of the diversity within Studland Bay and enable quantification and statistical analysis of the data.

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1 Introduction

1.1 Site overview and designated features

Studland Bay was designated as a Marine Conservation Zone (MCZ) on 31st May 2019 and contributes to the UK's suite of MCZ sites and overall MPA network. The features (including Habitats and Species of Conservation Importance and Broad-scale habitats (BSH)) for which the site was designated, are listed below.

The MCZ was designated (under the Marine and Coastal Access Act) for the following habitats and species:

- Intertidal coarse sediment
- Subtidal sand
- Seagrass beds (*Zostera marina*)
- Long-snouted (spiny) seahorse *Hippocampus guttulatus*

A description of the seagrass beds within Studland Bay is provided within Natural England's Conservation Advice for the site (Natural England, 2022). "Seagrass beds are primarily found within the south and southwest corners of Studland Bay down to about 4m [Figure 1] and have been shown to have an important role in sequestering atmospheric carbon (Green *et al*., 2018). The seagrass beds support a high diversity of fish, including pipefish, wrasses and undulate ray (*Raja undulata*) and provide a nursery area for commercially important fish and shellfish, such as black bream (*Spondyliosoma cantharus),* pollack (*Pollachius pollachius*), cuttlefish (*Sepia officinalis*), sole (*Solea solea*) and plaice (*Pleuronectes platessa*). Studland Bay is currently the only known breeding location in the UK for the indigenous spiny (or long-snouted) seahorse (*Hippocampus guttulatus*). In addition, the short-snouted seahorse (*Hippocampus hippocampus*) and all six species of pipefish have been recorded here, including the rare Nilsson's pipefish (*Syngnathus rostellatus*) (Seasearch Dorset, 2015). The beds are also an important food source for overwintering wildfowl such as brent geese (*Branta bernicla)*."

For full site details please access the Conservation Advice package for the site using the following link: [Natural England Conservation Advice for Marine Protected Areas: Studland Bay MCZ - UKMCZ0072.]

OSPAR (2009) states that to "qualify as a *Zostera* 'bed', plant densities should provide at least 5% cover (although when *Zostera* densities are this low, expert judgement should be sought to define the bed). More typically, however, *Zostera* plant densities provide greater than 30% cover." The seagrass in the southwest corner of Studland Bay has been previously reported as providing plant cover of approximately 50-70% (Plastow, 2009; Axelsson *et al*., 2010; Seastar, 2012) and therefore qualifies as a seagrass bed.

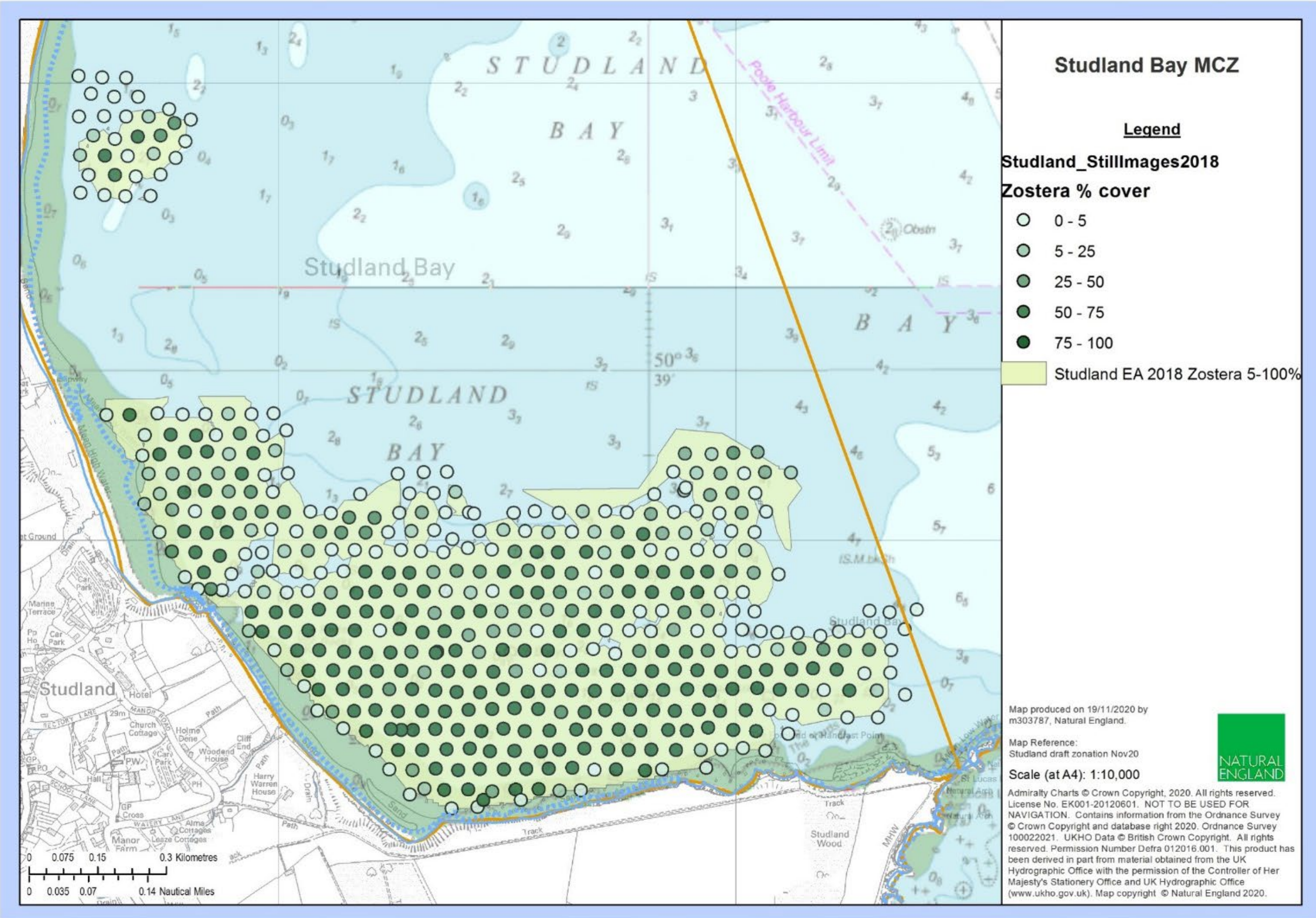


Figure 1: Latest seagrass extent mapped in 2018 by the Environment Agency (Green, 2018), reproduced by Natural England under the Open Government Licence.

Generally, each MCZ has one overarching Conservation Objective (CO) that applies to all of the features being protected; that they be protected in ‘favourable condition’. To achieve this objective, the general management approach required for a feature in an MCZ will either be for it to be maintained in favourable condition (if it is currently in this state), or for it to be recovered to favourable condition (if it is currently in a damaged state) and then to be maintained in favourable condition

Generally for a habitat, favourable condition is defined as:

- its extent is stable or increasing; and
- its structures, functions, quality and the composition of its characteristic biological communities are such as to ensure that it remains in a healthy condition and does not deteriorate (Defra, 2013).

For a species, favourable condition means that the quality and quantity of the species’ habitat, and the composition of the species’ population in terms of number, age and sex ratio, are such as to ensure that the population is maintained in numbers which enable it to thrive. For some highly mobile species this definition will be adapted to reflect that the species is only present in the MCZ for part of its life-cycle and/or for a particular purpose (e.g. mating, egg-laying). For further details see Defra (2013).

The primary purpose of this document is to report on the monitoring fieldwork undertaken during a five-day period between 19th and 23rd July 2021. The surveys were led by Natural England staff and supplemented by external consultant marine biologists.

1.2 Project background and aims

The Studland Bay Marine Conservation Zone Habitat Protection Strategy (Marine Management Organisation, 2021) states that "Natural England is responsible for reporting on the condition of the designated features of the MCZ every six years. To that end, Natural England works with partners to carry out monitoring and establish the best available evidence base to assess the condition of site features. Depending on resources, formal monitoring of sensitive features like seagrass occurs more regularly than other features because of its vulnerability and comparatively rapid changes that can occur. The Marine Management Organisation (MMO) will use this, and other pertinent ecological data, alongside activity data whilst reviewing the effectiveness of management.

The extent and quality of the seagrass is monitored using a range of techniques including remote sensing (e.g. acoustic ground discrimination), echo-sounding, drop-down video and diver-collected data. Quality is monitored using a variety of metrics, including: seagrass percentage cover; number of shoots present; maximum shoot length; presence of epiphytes and infection; and evidence of anthropogenic impacts. This data is used alongside activity data to inform reviews of the Natural England Conservation Advice for the site, which can be subsequently updated.

The monitoring within Studland Bay MCZ, undertaken in July 2021, was designed to acquire high quality data of suitable resolution to allow key quality attributes of the seagrass bed (*Zostera marina*) MCZ feature to be assessed and to produce this follow-up report. This report provides evidence to contribute to a future condition assessment allowing Natural England to monitor for any changes in the seagrass cover and density; where available, comparisons with other existing data are also made.

The present monitoring programme specifically assessed seagrass density, percent cover, leaf length, epiphytic cover, disease and reproductive status at a number of sites throughout the bay. It did not assess extent and distribution which was last monitored in 2018.

The data collected also provided information on the presence of any non-native species and pathogens as well as providing some supplementary information on the presence of associated biological communities within the beds.

Specific aims of this report are to:

- Provide supplementary evidence to inform condition of subtidal seagrass in the site.
- Assess insofar as possible the specified attributes (seagrass extent and distribution, habitat and community structure and function, seahorse population) to allow for a baseline to be established. This information will be used to allow Natural England to undertake a condition assessment of the subtidal seagrass bed feature of Studland Bay MCZ in the future (see Section 4).
- Consider any spatial variation observed, notable communities or exposure to environmental or anthropogenic factors.

- Apply appropriate statistical analyses of data to enable the assessment of each attribute target in the future. Where possible, if previous data allows, provide a comparative piece of analysis.
- Appraise the sampling design/intensity through post-survey power analyses.
- Evaluate the effectiveness of data collection methods, techniques and technical equipment.
- Provide opinion on the integrity of the feature, which will be considered by Natural England in future condition assessments.
- Record information about seahorses if found whilst undertaking survey work.

Natural England will review this evidence when carrying out its overall feature condition assessment.

1.3 Previous monitoring surveys

A number of surveys using a range of techniques have been undertaken previously in the seagrass beds of Studland Bay. Reports from a number of these have been used to inform the Results and Discussion sections of this report and are summarised in Table 1.

Table 1: Summary of monitoring surveys of the Studland Bay seagrass beds and referenced in this report.

Year(s)	Methods	General aims (not exhaustive)	Reference
2008-9	Tagging	Seahorse surveys	Garrick-Maidment *et al.* (2010)
2009 -2011	Transects and quadrats to monitor anchor scars and shoot density	Investigate impacts from anchoring and mooring	Axelsson *et al.* (2010); Seastar (2012)
2009	Video sledge and side scan	Investigate impacts from anchoring and mooring	Collins *et al*. (2010)
2014	Quadrats (n = 3)	Collect measurements of shoot density, percent cover, leaf length, epiphytes and measurements of C, N and P	Jones & Unsworth (2016)
2013; 2014; 2015	Diver records	Record marine life	Seasearch Dorset (2013; 2014; 2015)
2016	Core sampling (n = 3) and quadrat surveys (n = 9)	Sediment carbon measurements with additional quadrats to record shoot density	Green *et al*. (2018)
2018	Drop camera and echo sounder	Map bed extent and seagrass percentage cover	Green (2018)

2 Methods

2.1 Dive operations

The dive surveys were carried out between 19-22 July 2021. A Swanage-based chartered hard boat, the *Mary Jo*, an Offshore 32 hard boat, category 2 MCA registered vessel, acted as the diving platform. The vessel operated from Swanage Pier. The diving work comprised a team of six divers plus one dedicated non-diving supervisor and a surface stand-by on the vessel. Due to the enclosed nature of Studland Bay and shallow water (<8m), none of the sites were restricted by a need to dive at slack water and were diveable at all states of the tide. However, the need to divert any water users, boats etc increased around low water and from late morning onward.

All diving was carried out under the Diving at Work Regulations (1997). All health and safety matters relating to the diving undertaken was governed by this legislation, the accompanying Scientific and Archaeological Approved Code of Practice (ACOP), and the Rules and Guidance for Scientific Diving in the Statutory Nature Conservation Bodies (Holt, 2015). In accordance with these regulations all divers were qualified to HSE Pt IV or equivalent CMAS 3*.

Natural England produced and supplied a Diving Project Plan detailing diving operations, site-specific information, risk assessment and emergency procedures. The plan detailed the sites to be dived on each day of field work and the times of low / high water when diving operations would take place.

The scientific diving team used SCUBA diving equipment and air. Diver pairs were equipped with through-water surface-to-diver voice communications with a diver-to-surface beep return (one per buddy pair as a minimum). This communication system provided the primary communication and recall facility.

2.2 Dive surveys

The seagrass bed within Studland Bay was surveyed by divers at eight different sampling stations. A selection of sampling stations were identified prior to the survey based on a range of depths and anticipated seagrass densities, the latter based on Environment Agency (EA) echosounder and drop camera survey data collected in 2018 (Green, 2018) (Figure 2).

The exact sites surveyed during the field work week were dependent on the prevailing conditions i.e. weather, tides, vessel activity. At each sampling site the skipper placed a shot line as close as possible to the site target position and the exact GPS position (WGS84) of the shot was recorded (Table 2). Site 5a is named such as it was repositioned between proposed sites 5 and 6, neither of which could be surveyed due to either shallow water or a total lack of seagrass.

Figure 2: 2021 seagrass monitoring sites in Studland Bay marked with yellow pins. Areas of seagrass show up as darker areas against the sand (Source: Google, ©2021 CNES / Astrium, Maxar Technologies. Image dated 16 July 2021).

Table 2: Positions of the eight sampling stations in Studland Bay, Dorset surveyed between 19-22 July 2021.

Site number	Latitude (WGS84)	Longitude (WGS84)
1	[illegible]	1° 55.994'W
2	50° 38.558'N	1° 56.359'W
[illegible]	50° 38.666'N	1° 56.573'W
4	50° 38.766'N	1° 56.211'W
5a	50° 38.860'N	1° 56.800'W
7	50° 39.268'N	1° 56.928'W
8	50° 38.752'N	1° 55.959'W
10	50° 38.674'N	1° 56.359'W

Three buddy pairs dived each site to record data and collect samples at pre-defined locations. On entering the water, pair 1 contacted the surface to confirm seagrass was present. As long as seagrass was present, pairs 2 and 3 then entered the water at 5-minute intervals to give the previous pair time to descend and attach their tape measure to the shot, thereby avoiding ‘congestion’ of divers.

Each survey station was defined as a circular area of 30 m radius divided into three working areas – one per diver pair (Figure 3). These working areas were defined approximately as:

- Pair 1: 0° and 120°
- Pair 2: 120° and 240°

- Pair 3: 240° and 360°

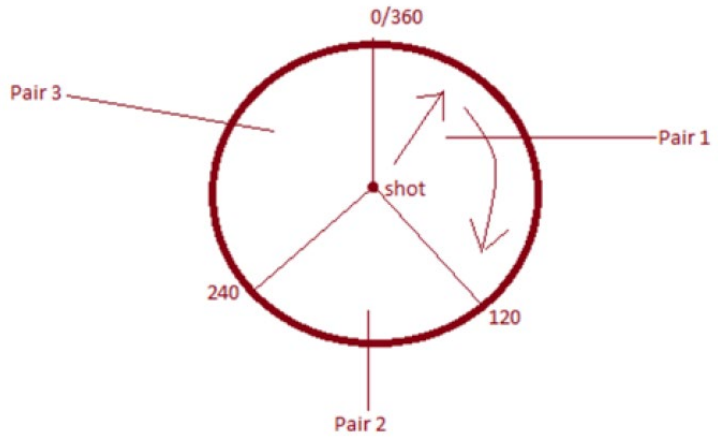


Figure 3: Approximate representation of the layout of each seagrass survey station.

The actual working areas per diver pair were pre-determined by the random calculated vectors along which divers should survey the seagrass. Graduation of sample bearings ensured buddy pairs were not attempting to work on the same bearing at the same time.

Each buddy pair carried out the following methodology:

- One diver attached the tape measure to the shot line, the pair then headed from the shot on a compass bearing and distance as stated on their first sample bag. Upon reaching the distance stated they placed the bottom left corner of a 0.25 m^2 quadrat down at the predetermined distance on the tape measure.

- Diver 1:
    - Photographed the quadrat.
    - Recorded % cover of seagrass in a 0.25 m^2 quadrat to the nearest 5%.
    - Recorded sediment type.
    - Recorded total % cover of other algae.
    - Recorded presence of any non-native species.
    - Counted the total number of shoots within the 0.25 m^2 quadrat.

- Diver 2:
    - Placed a 0.0625 m^2 quadrat adjacent to the 0.25 m^2 on the opposite side of the tape measure (i.e. right bottom corner aligned with left bottom corner of larger quadrat).
    - Cut all seagrass shoots within the quadrat whilst ensuring that shoots were cut low on the stem so that the plant stayed intact but not too low that the rhizome would be damaged. This allows the shoot to regrow.
    - Placed the shoots in the labelled plastic sample bag, tied and placed in a mesh bag.

After completion of each quadrat, each diver pair returned to the shot for a new bearing and distance, written on the next sample bag. The sample bags and survey form were all pre-labelled so that all diver pairs worked in a clockwise direction from their start point, ensuring that there was no overlap between pairs. If the next quadrat was within a short distance of the previous there was no need to return all the way to the shot. At each site, a total of 25 quadrats were surveyed and sampled (i.e. 8-9 quadrats per buddy pair).

Where possible, site transects were videoed using GoPro or Olympus compact cameras. Photographs were taken of each of the quadrats. Time allowing, images were also taken to illustrate the biota present in the seagrass beds, the variability in density and epibiota and the community and physical structure of the beds.

If divers completed the quadrat survey and had sufficient time and air they were able to conduct a short search for seahorses. Licence no. L/2019/00144/3 was issued by the Marine Management Organisation, under The Wildlife and Countryside Act 1981 (As Amended) Section 16: Power to grant licences, to intentionally disturb the species *Hippocampus guttulatus* and *Hippocampus hippocampus* whilst occupying place of shelter or protection, for the purposes of scientific (research) or educational purposes. To undertake this survey the divers unclipped their tape measure from the shot line and deployed a delayed surface marker buoy.

2.3 Post-dive sample analysis

All the shoots sampled from each 0.0625 m^2 quadrat were analysed post-dive at the end of each diving day to ensure no degradation of the samples. Shoots were assessed for:

- Maximum leaf length
- Degree of infection with *Labyrinthula* sp.
- Abundance of epiphytes
- Presence of invertebrate eggs
- Presence of flowering plants

Following training to ensure consistency of measurements and visual assessments, divers took each shoot collected and measured its longest leaf length to the nearest cm. On each shoot, each intact leaf was assessed to estimate the percentage cover of *Labyrinthula* sp. infection and epiphyte cover on a scale of 0-5 (Table 2). Culturing and isolation methods were not employed to prove *Labyrinthula* sp. infection, therefore infection of the plant by *Labyrinthula* is inferred, based on the methods in Burdick *et al*. (1993). A photograph showing infection and epiphyte cover on a *Zostera* sp. plant is shown in Figure 4. Sample processing was conducted in pairs, recording data onto hard copy data sheets. Data were then entered onto a spreadsheet following completion of sample processing.

Table 3: Scoring scale used for recording level of leaf infection and epiphyte cover.

Description – leaf infection / epiphyte cover	% Affected	Score
Uninfected / bare	0	0
Minimal	0-2	1
Up to a quarter	3-25	2
Up to half	26-50	3
Over half	51-75	4
Almost all	76-100	5

Figure 4: Amphipod tubes on the surface of a *Zostera* leaf (left hand side) and infection (black patch on right hand side).

2.4 Quality assurance

Scientific divers from Natural England were joined by two experienced contract marine biological surveyors providing an experienced diving team with excellent marine biological expertise to identify species *in situ* from quadrats and to work up specimens that required further identification.

At the commencement of the fieldwork the survey-specific ‘recording rules’ were discussed to ensure surveyors applied the rules consistently. In addition to this there was a briefing each evening to discuss site-specific and methodological details in preparation for each site visited the following day.

2.5 Statistical analysis

2.5.1 Data analysis

Univariate analyses were carried out in Minitab 16. Correlations were investigated using the Pearson product moment correlation coefficient to test for relationships between pairs of variables.

Scatter plots and box plots were created in Microsoft Excel. The boxplots illustrate the mean, median, interquartile ranges and variation in the data. With regard to the whiskers, these extend up from the top of the box to the largest data element that is less than or equal to 1.5 times the interquartile range (IQR) and down from the bottom of the box to the smallest data element that is larger than 1.5 times the IQR; these may or may not also be the maximum and minimum values. Values outside the whisker range are considered as outliers and are represented by dots.

2.5.2 Power analysis

The power of a statistical test to detect change is an important consideration in the design and execution of any experiment or monitoring programme. The collection of too few samples might mean that incorrect conclusions are reached if data cannot demonstrate significant differences when they are known to exist (Type II errors), whilst the collection of too many samples can be a waste of resources. Power analysis therefore is important in predicting future survey and analytical costs while ensuring that data collected are fit for purpose.

The power analysis results presented for this survey are based on the outputs from the MONITOR programme which uses simulation procedures to evaluate how each component of a monitoring program influences its power to detect change (Gibbs & Ene, 2010). The programme is devised with population monitoring in mind and allows the user to define the planned sampling design. Outputs indicate the power to detect specified levels of change over any specified time range based on the known or estimated population mean and standard deviation. Further explanation of the tests is provided alongside the outputs in Section 3.8.

2.5.3 Historical data

None of the data from previous surveys available was suitable for statistical comparison with that collected during the 2021 monitoring due to historical variation in the techniques, survey effort and different project aims. For qualitative comparison purposes, parameters such as shoot density and percentage cover from previous surveys have been cited alongside the 2021 data, but no further analysis has been undertaken.

The quantitative data collected in the present survey do provide a solid baseline against which future surveys can be compared assuming that the methods and sites used here are repeated. Interpretation of any comparison with future surveys would be aided by repeated extent surveys such as that by Green (2010).

3 Results and Discussion

The overall objectives of the survey were to collect high-quality data to:

- allow key attributes of the seagrass bed feature to be assessed;
- provide supplementary evidence to inform a condition assessment;
- to compare with existing data where possible;
- provide an indication on the condition of the feature to allow Natural England to undertake a formal condition assessment.

This results section reports the known extent and distribution of the seagrass bed, its percentage cover and density, and additional data on bed structure and health. Finally, a power analysis considers how effectively the data can determine changes over time, changes related to management measures and the efficiency of the sampling methodology. Summary descriptions and example images of each of the eight monitoring sites within Studland Bay are provided in Appendix A.

3.1 Extent and distribution

Monitoring of the extent and distribution of the seagrass beds in Studland Bay was not an aim of the 2021 diving surveys. Extent and distribution were last monitored and reported by the Environment Agency in 2018 (Green, 2018). Figure 5 summarises the results of that survey and shows the changes in seagrass percentage cover across the site in five graded categories of low to high percentage cover. The data demonstrate that the densest areas of seagrass are located in the most sheltered and inshore parts of Studland Bay, which face away from the prevailing south-westerly winds.

Although the resolution in Figure 5 is limited by the frequency and methodology of sampling (1 m^2 quadrats at 50 m intervals, n = 454) it provides a good visualisation with high data confidence of the seagrass distribution and extent, highlighting larger areas of patchy cover. Given the resolution of the data (sample intervals of ~50 m), Figure 5 is unable to accurately illustrate patchiness that is present at smaller scales (<50 m), which is visible through aerial imagery and was observed by the survey divers in July 2021 (see Sections 3.2.1, 3.7 and 4.1.2).

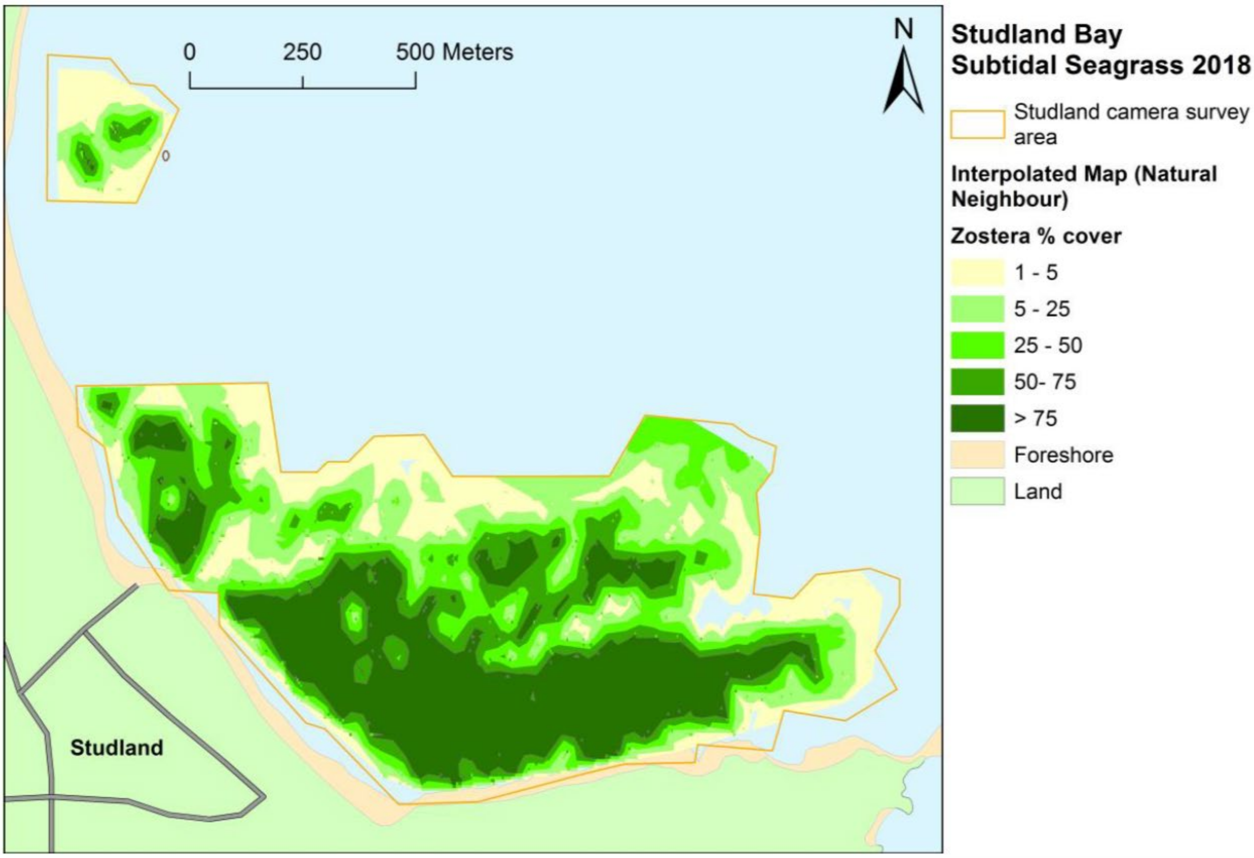


Figure 5: Interpolated map (using Natural Neighbour algorithm) of subtidal seagrass density from the 2018 drop-camera survey of Studland Bay (Green, 2018). Reproduced under the Open Government Licence.

3.2 Seagrass bed structure: *in situ* quadrat data

In this section the results of the 2021 surveys are presented and temporal comparisons are given for each parameter where previous data are known to exist.

In addition to exposure, water depth and substrate can be key environmental drivers that influence the presence and distribution of seagrass (Borum *et al*., 2004). Increasing water depth is associated with a decrease in light levels, thereby limiting the ability of seagrass to grow and establish beds. The data obtained during the 2021 quadrat monitoring showed some weak, positive correlations of % cover and shoot density with depth, contrary to what would be expected. Seagrass % cover was weakly positively correlated with increasing depth (R = 0.187, p = 0.009) as was shoot density per m^2 (R = 0.370, P < 0.001) (Figure 6). This apparent reversal of the recognised relationship with depth is most likely owing to the non-random selection of the sampling sites in this instance toward areas likely to support seagrass (within the known bed boundaries) and thereby not extending sampling to greater depths across a depth profile. Sites 4 and 8 are most likely responsible for skewing the data toward the positive relationship owing to the dense stands of seagrass at these sites, which were also the deepest ones surveyed and possibly less disturbed by wave action (Figure 6); removing them from the dataset resulted in similar correlation coefficients for both % cover and shoot density in a negative direction ($R = -0.318$, $p < 0.001$; $R = -0.245$, $p = 0.003$, respectively).

There was no correlation of % algal cover and sampling depth ($R = -0.038$, $p = 0.597$). All sediments were recorded as either sand, fine sand, muddy sand or coarse sand. These records were made on a visual and therefore somewhat subjective basis by each surveyor and therefore should not be subjected to any statistical analysis. Despite this, there was no obvious difference between seagrass cover or density data with how the sand types were categorised. Seastar (2012) noted from their surveys within Studland Bay that two natural deeper channels existed with coarser sediments and an absence of seagrass; these locations were not surveyed in 2021 but were visible within the bed.

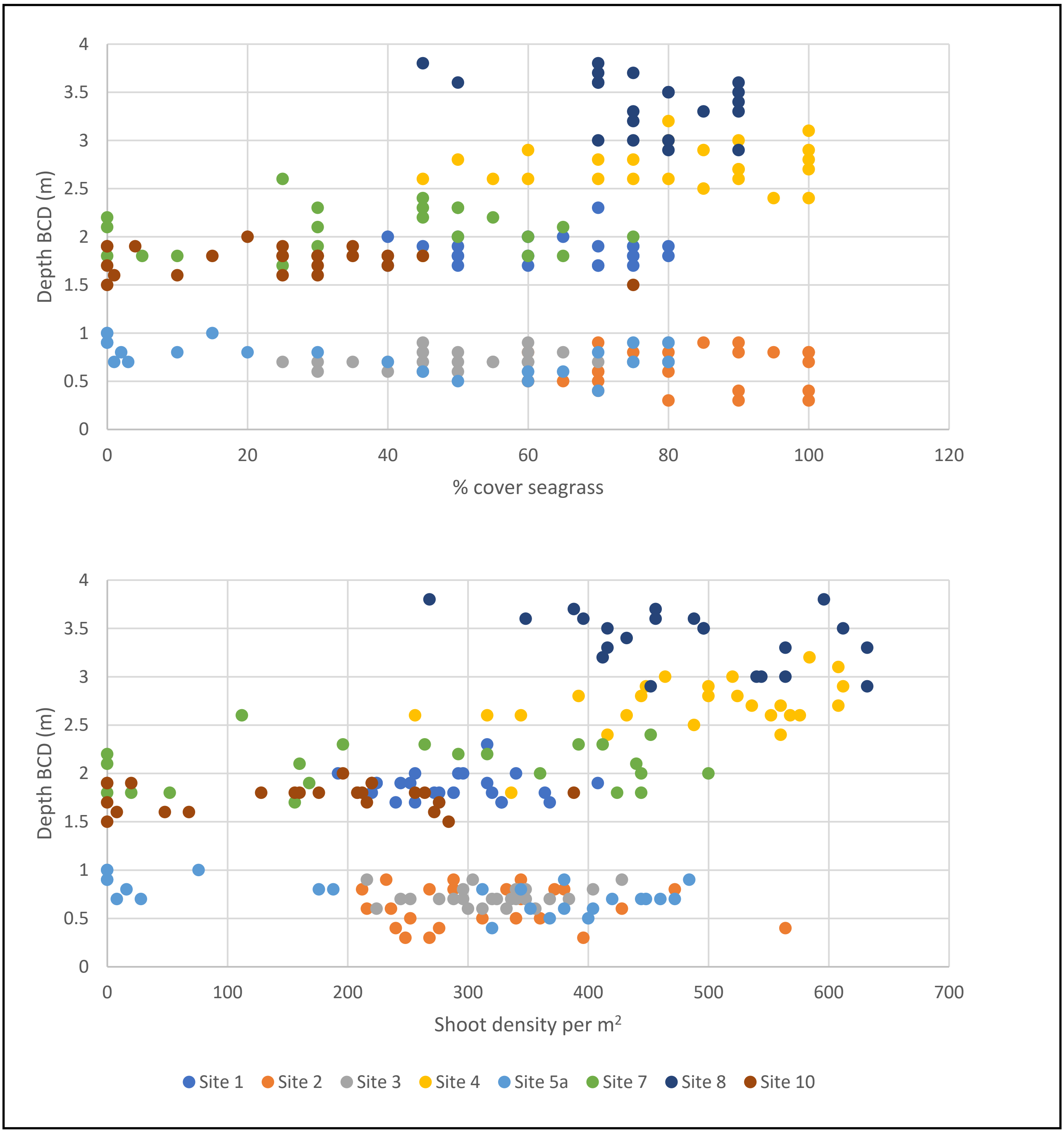


Figure 6: % seagrass cover (above) and shoot density per m^2 (below) plotted against depth below chart datum (BCD) from eight sites in Studland Bay, Dorset, July 2021.

3.2.1 Seagrass % cover

The mean percentage cover of seagrass assessed at each sampling location in the bay ranged between 24% and 81% (Figure 7). The mean percentage cover across all sites was 57%.

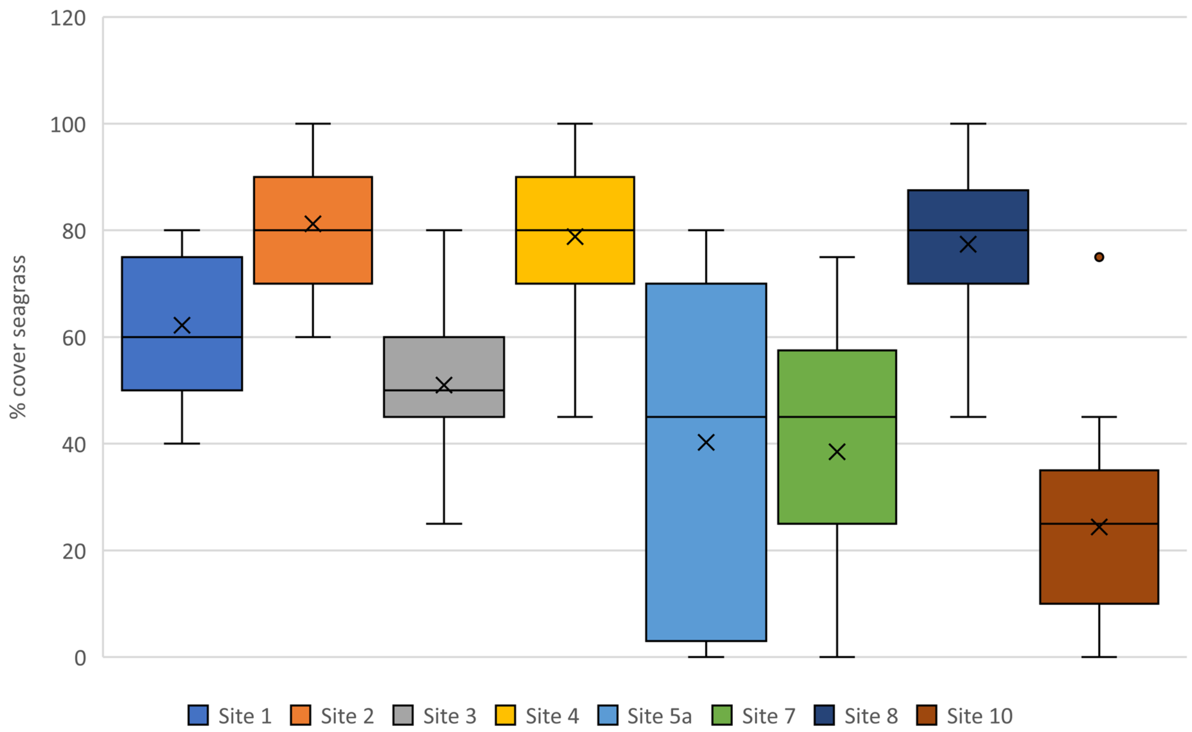


Figure 7: Boxplot of percentage cover seagrass assessmentswithin 0.25 m^2 quadrats (n=25) at each of eight sites within Studland Bay, Dorset in July 2021. Boxes show mean (x), median (-), interquartile ranges (boxes), whiskers and outliers.

The EA data and Google Earth imagery illustrate the patchy distribution of seagrass within Studland Bay (Figure 5 and Figure 2). The data in Figure 7 help to demonstrate which sampling sites within Studland Bay were either more patchy or more uniform in terms of seagrass cover and appear to align well with the more extensive drop camera survey data (Figure 5). Data for % cover at sites 1, 2, 3, 4 and 8 were less variable than those for sites 5a, 7 and 10; the latter all recording quadrats with seagrass ranging from 0% to 70-80% cover. Whilst there were no quadrats with absent seagrass at site 3, five quadrats were recorded with seagrass cover as low as 25-40% which were the next lowest % cover values of the other sampling sites.

Although not directly comparable from a statistical perspective, the mean % seagrass cover values were assessed using 0.0625 m^2 quadrats during early autumn at different sites within Studland Bay between October 2009 and October 2011. Those data compare favourably with the 2021 survey data, with mean % cover values ranging between 5 – 90% per quadrat and the overall mean values per survey ranging between ~49-62% (Seastar, 2012).

3.2.2 Seagrass density

The mean density of seagrass shoots per quadrat was multiplied up to provide values per m^2. The mean shoot density recorded at each sampling location in the bay ranged between 159 and 486 shoots per m^2 (Figure 8). The mean density across all sites was 322 shoots per m^2.

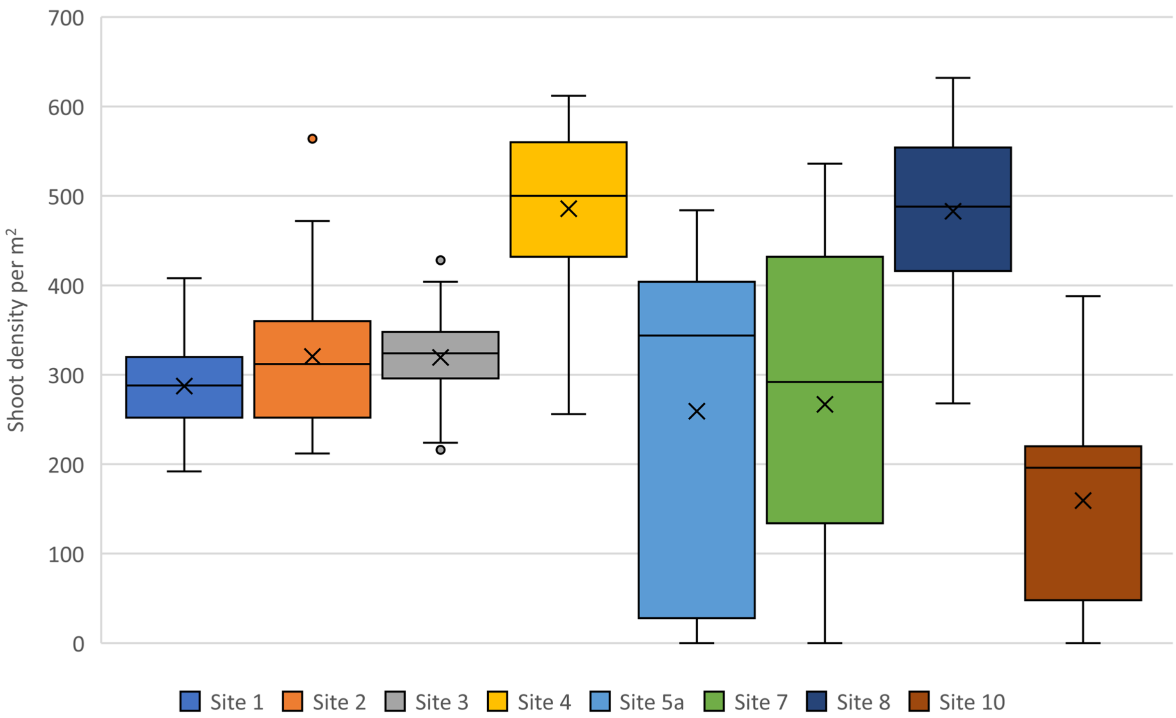


Figure 8: Boxplot of seagrass shoot density per m^2(n=25) at each of eight sites within Studland Bay, Dorset in July 2021. Boxes show mean (x), median (-), interquartile ranges (boxes), whiskers and outliers.

The data in Figure 8 exhibited a very similar pattern to those for % cover (Figure 7), again reflecting the patchy distribution and variable density of seagrass within the bay (Figure 5). Not surprisingly, there was a strong correlation between seagrass % cover and shoot density per m^2 ($R = 0.820$, $p < 0.001$) (Figure 9).

As with the % cover data, shoot density data from sites 1, 2, 3, 4 and 8 were less variable than those for sites 5a, 7 and 10; the latter all recording quadrats with seagrass densities ranging between 0 to ~390-540 shoots per m^2 – arguably more patchy than the other sites. As with the % cover values, sites 4 and 8 exhibited the highest densities of seagrass whilst values for site 3 were more similar to sites 1 and 2. The highest densities recorded were at site 8 with 632 shoots per m^2 recorded from two quadrats. Compared with other seagrass beds in the southwest (e.g. the Fal and Helford estuaries, Torbay, Plymouth Sound), the mean shoot densities recorded were very high, with the other southwest surveys often recording mean densities well below 120 shoots per m^2 (Curtis, 2015; Bunker & Green, 2019; Field, 2019). Densities recorded from Poole Harbour are comparable with those from Studland Bay, even reaching up to ~780 shoots per m^2 (Envision, 2015). Determining the reason for these differences and similarities is beyond the scope of this study but might be attributable to a range of environmental factors including a site's degree of shelter from wave energy or to local nutrient levels. Further investigation would be needed to reach any firm conclusions since seagrass growth may be increased at low-moderate nitrogen (N) or phosphorus (P) levels but inhibited at higher concentrations, with the source of the N or P also playing a significant role i.e. from the water column or sediment (Touchette & Burkholder, 2000). Further reference to nitrogen levels in Studland Bay is made in section 3.7.

Overall, the seagrass bed at Studland Bay exhibits obvious spatial variation in terms of shoot density and percentage cover, both of which are well-correlated with one another.

The greatest variation within sites was observed at sites 5a and 7, the two more northerly locations with the most patchy distribution. Both the Environment Agency (2018) data and available aerial imagery (Figure 5) illustrate the variation in density and patchiness across the site very effectively.

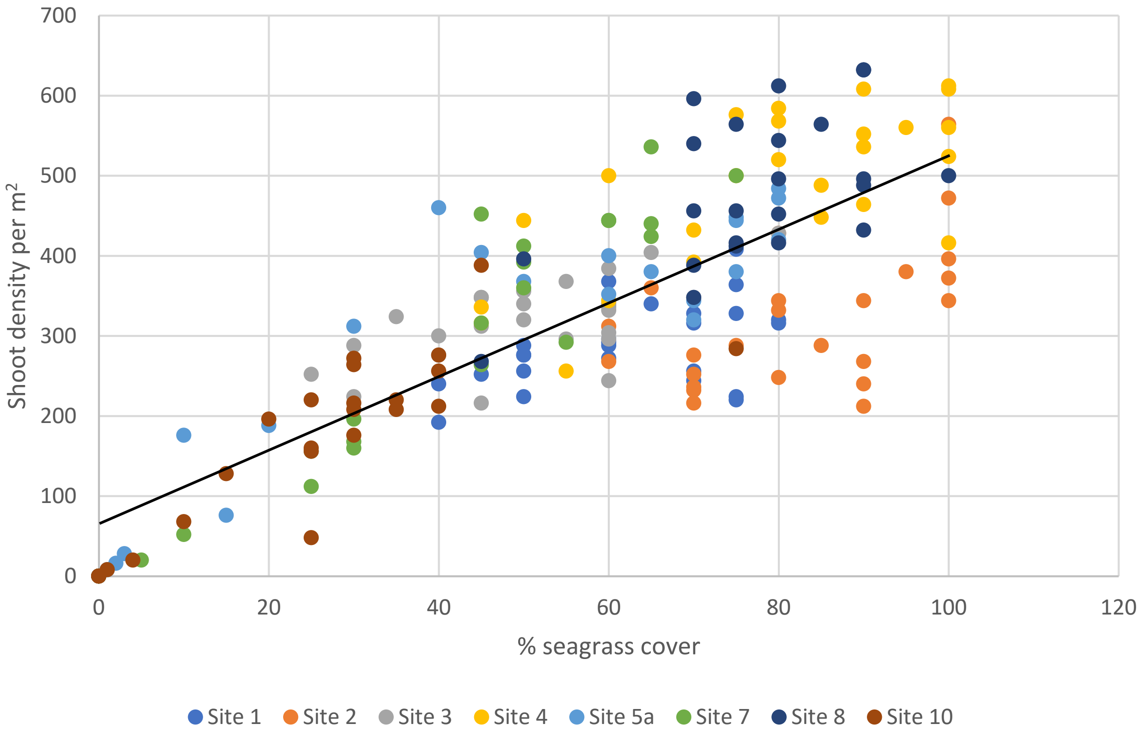


Figure 9: '% seagrass cover' plotted against 'shoot density per m^2'from eight sites in Studland Bay, Dorset, July 2021. Trendline shows the significant and strong correlation (R = 0.820, p < 0.001).

Although not directly comparable statistically owing to a different survey season, the mean shoot densities were assessed using 0.0625 m^2 quadrats during early autumn at different sites within Studland Bay between October 2009 and October 2011. Data from those surveys recorded shoot densities a little lower than the 2021 survey data, with overall mean densities per survey ranging between ~160 – 240 shoots per m^2 (Seastar, 2012). Use of smaller quadrats might have resulted in these different values or they may reflect genuine spatial / temporal differences in shoot density. Samples collected as part of wider studies in 2014 and 2016 recorded shoot densities within the bay of 144 and 212 shoots per m^2 respectively (Jones & Unsworth, 2016; Green *et al*., 2018). It should be noted that the sampling intensity and area covered in those studies was significantly lower than the present study, n=3 and n=9 quadrats respectively. The use of different sampling apparatus, different survey seasons and sometimes small sample sizes in the previous studies cited here, mean the data should not be directly compared with the July 2021 data and should not be used to infer any increase in seagrass density.

3.2.3 Algal cover

The mean percentage cover of algae assessed at each sampling location in the bay ranged between 6% at site 5a and 62% at site 2 (Figure 10). The mean percentage cover across all sites was 22%. Site 2 had the highest overall % algal cover values compared to the other sites, perhaps related to being in the most sheltered part of Studland Bay compared with the other sampling locations. With data from site 2 excluded, the mean % algal cover was reduced to 16%.

Surveys undertaken during early autumn between 2009 and 2011 recorded algal cover anecdotally either per five quadrats surveyed, or for wider transect survey areas inside and outside ‘no anchor zones’. Typically the % cover values cited ranged between 0 – 20 %, with values of 10% cited most frequently (Seastar, 2012). It is not possible to make any meaningful statistical comparison with these data owing to the different survey methods used.

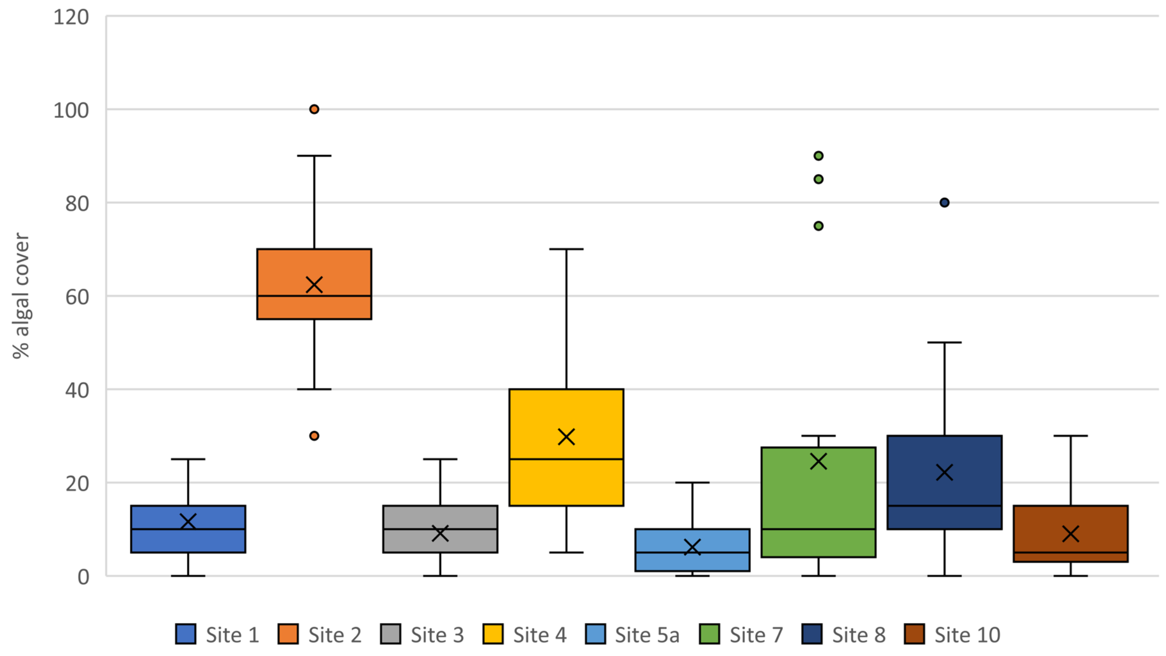


Figure 10: Boxplot of % algal cover (n=25) at each of eight sites within Studland Bay, Dorset in July 2021. Boxes show mean (x), median (-), interquartile ranges (boxes), whiskers and outliers.

3.3 Seagrass bed structure: quadrat sample data

3.3.1 Leaf length and health

To provide some indication of the health of the seagrass plants within the bay, the longest leaf length on each shoot was measured and scores of 0-5 were assigned to each leaf on each shoot to indicate both the degree of infection with *Labyrinthula zosterae* and the level of epiphyte cover. These parameters are summarised in Figure 11 to Figure 13 and provide an indication of the variability throughout the bay.

The single longest leaf length was 124 cm from site 7, the most northerly of the sampling sites. However, the sites with the longest mean lengths overall were sites 1 and 2 with 46.4 and 45.7 cm respectively. Excluding outliers, sites 1 and 2 also had the greatest range of leaf lengths suggesting the greatest age structure of the seagrass plants. These two sites were also the most southerly in the survey area and most protected from prevailing south-westerly winds. The shortest mean leaf lengths were from sites 3 and 5a with means of 27.1 and 21.6 cm respectively. Site 5a was in an area of very patchy seagrass with potentially greater exposure than the more southerly sites (see Figure 2 and Figure 7). Both sites 3 and 5a were close inshore; it is not clear whether there is a different amount of boating / anchoring activity compared with site 2 which had some of the longest mean leaf lengths but is also known to accommodate large numbers of vessels during summer months. Overall there were no clear spatial gradients of leaf length with distance from shore or from south to north.

The overall mean leaf length for the seagrass in Studland Bay was 37.0 cm (±17.2 stdev). This is shorter than the mean length of ~42 cm reported by Jones & Unsworth (2016) although their samples were collected from only three quadrats as opposed to 200 so may not have represented the bed as thoroughly as the present data set. Autumnal surveys between 2009 and 2011 gave *in situ* mean leaf lengths of 38 – 49 cm within Studland Bay depending on the location (Seastar, 2012). The later sampling date might have allowed for greater growth of the seagrass before the time of sampling or the *in situ* measurement technique might have skewed the data toward longer shoots if smaller plants were overlooked. Alternatively, different environmental factors such as temperature or turbidity may have influenced these differences between the sampling years.

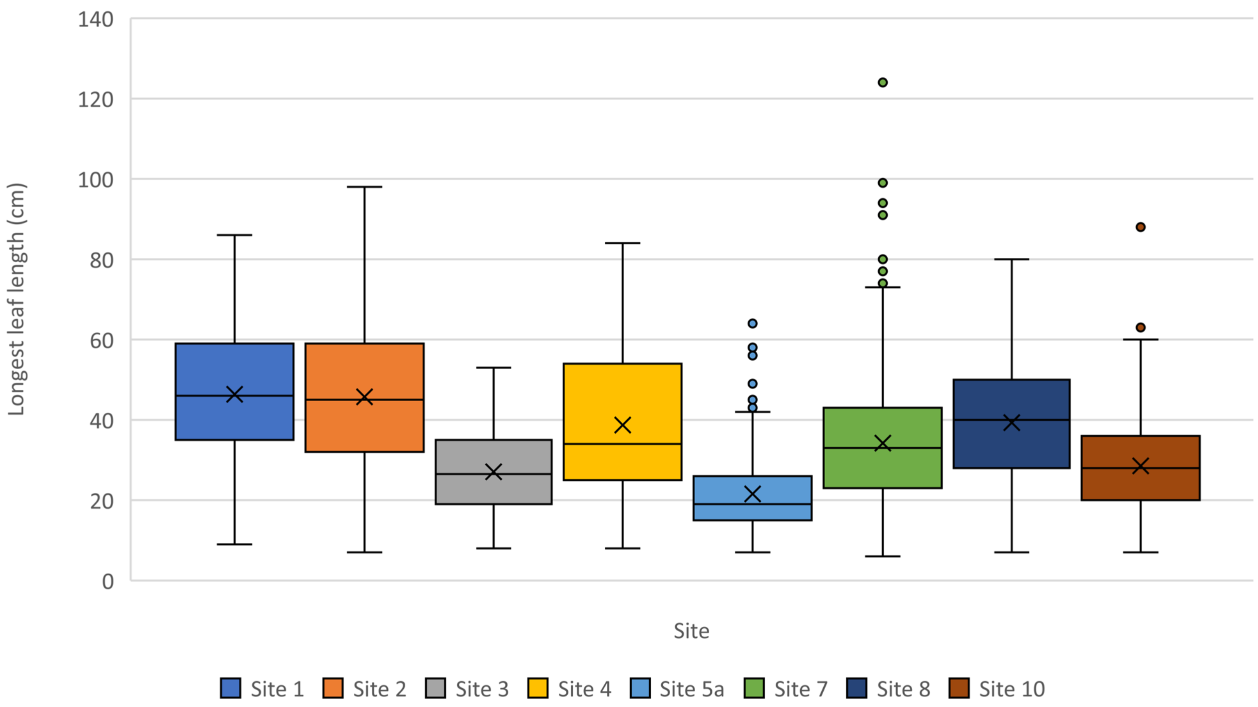


Figure 11: Boxplot of longest leaf lengths per shoot (n=25) at each of eight sites within Studland Bay, Dorset in July 2021. Boxes show mean (x), median (-), interquartile ranges

(boxes), whiskers and outliers.

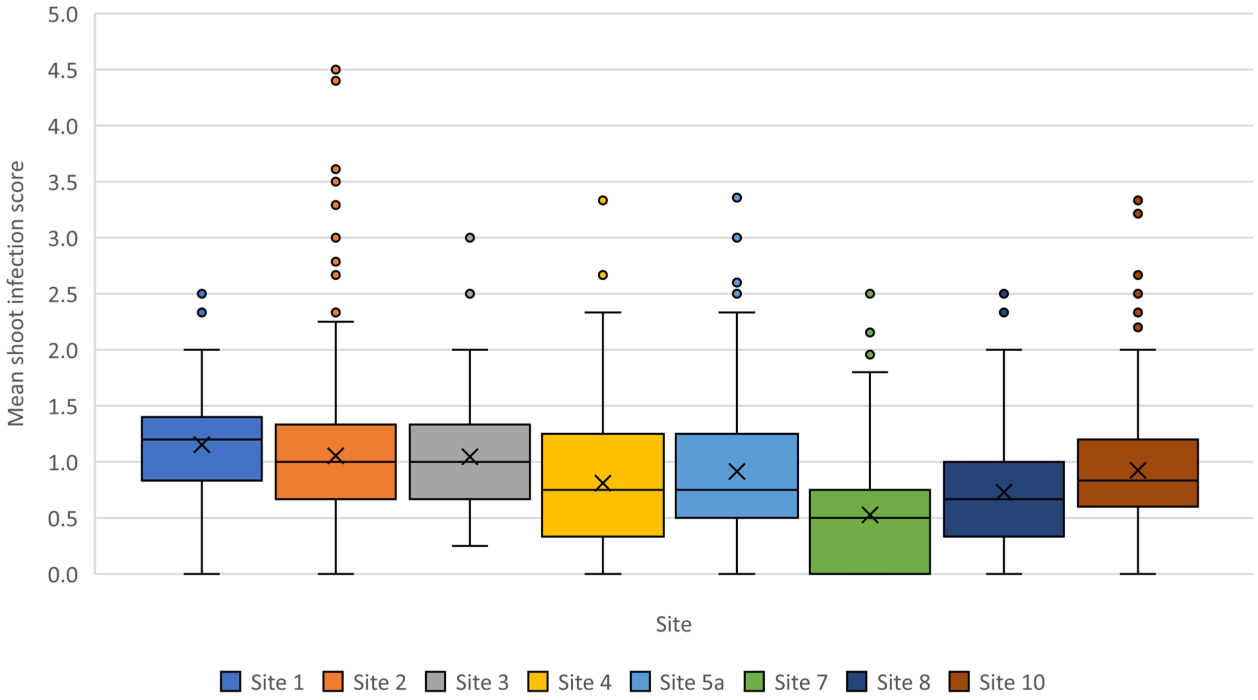


Figure 12: Boxplot of overall shoot infection scores (n=25) at each of eight sites within Studland Bay, Dorset in July 2021. Boxes show mean (x), median (-), interquartile ranges (boxes), whiskers and outliers.

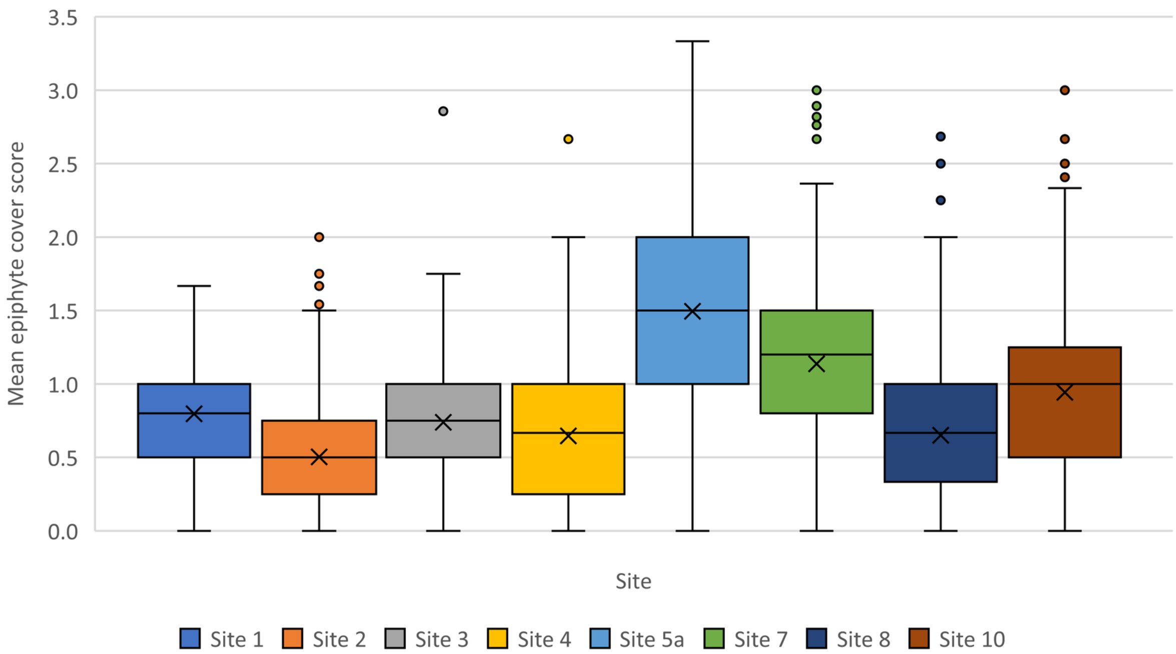


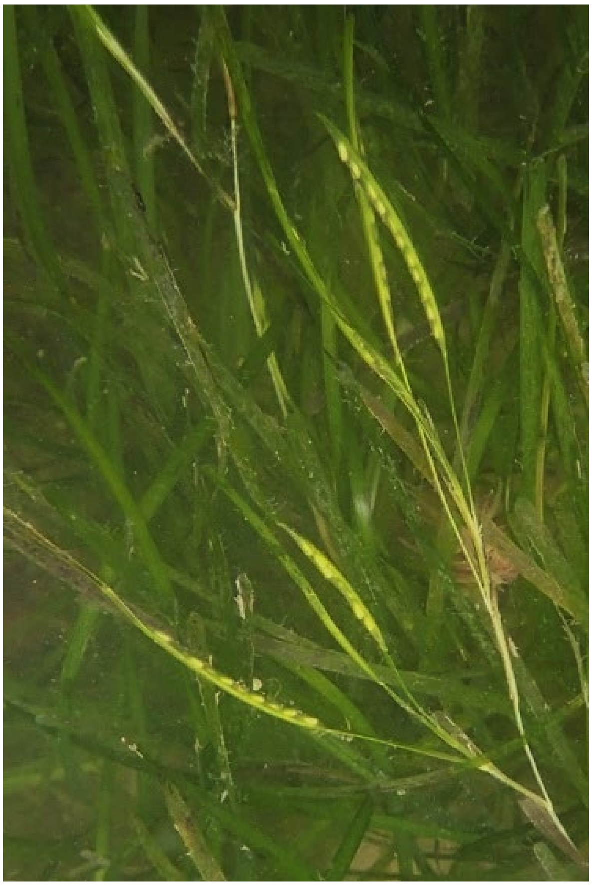
Figure 13: Boxplot of overall epiphyte cover scores (n=25) at each of eight sites within Studland Bay, Dorset in July 2021. Boxes show mean (x), median (-), interquartile ranges (boxes), whiskers and outliers.

The mean infection scores were low for all the sites, ranging between 0.5 and 1.2 (Figure 12). The lowest mean infection score of 0.5 was from site 7, the most northerly and exposed sampling site in the bay. The next lowest mean infection scores were from sites 4 and 8 at 0.8 and 0.7 respectively; these two sites were located in the outermost reaches of the main seagrass bed within the bay but also recorded the greatest shoot densities (Figure 8). The highest mean shoot infection scores were recorded at sites 1 and 2, the most southerly sampling locations. Increased sampling would be required to confirm whether a spatial gradient of infection exists within the bay influenced perhaps by either exposure, shoot density or some other environmental factor. The overall mean infection score for the seagrass in Studland Bay was 0.9 (±0.5 stdev).

Overall mean epiphyte cover scores per shoot were similar for the most southerly sampling sites, ranging between 0.5 and 0.9 (Figure 13). The mean scores were higher at sites 5a and 7 at 1.5 and 1.1 respectively, these two sites being north of the main seagrass bed. The overall mean epiphyte cover score for the seagrass in Studland Bay was 0.8 (±0.6 stdev). Epiphytic growth included filamentous algae, snakelocks anemones, fine hydroids, turf and encrusting bryozoans and amphipod tubes.

3.3.2 Flowering plants

Seagrass, specifically *Zostera marina* has been recorded to flower and seed subtidally in the Solent between May and July (Tubbs & Tubbs, 1983). It is unclear whether this phenology is applicable to the beds at Studland, but it is possible that the surveys were undertaken in the latter part of the flowering season for the area. All the flowering stems (spathes) observed were judged to be at developmental stage 4 (Infantes & Moksnes, 2018), which is a late developmental stage with mature seeds (Figure 14).



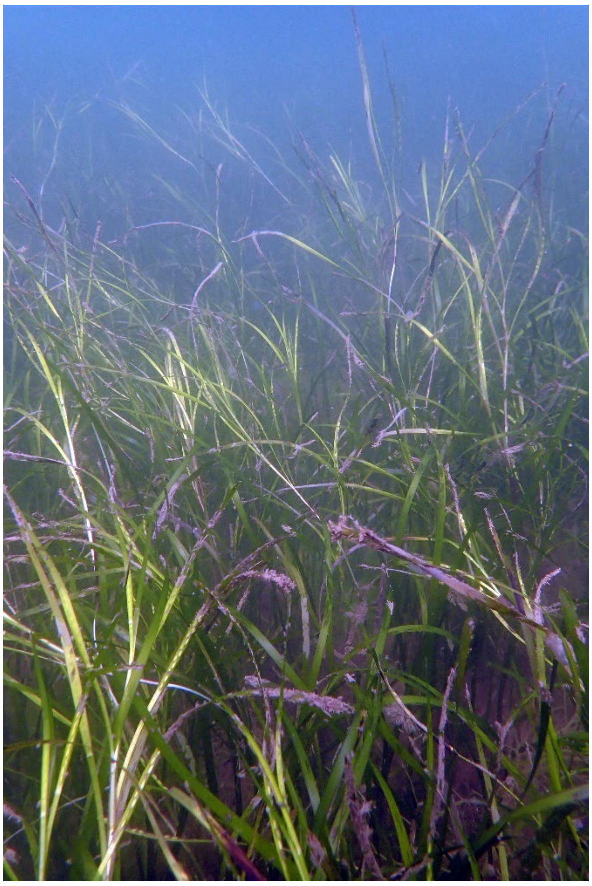


Figure 14: Flowering stems visible on *Zostera marina* (left) and a dense stand of flowering plants at site 7 (right) in Studland Bay, Dorset, July 2021.

Table 4 shows the total number of plants sampled and assessed from the 200 quadrats surveyed including the number and percentage of flowering plants at each sampling site. Overall, 1.5% of the plants were flowering at the time of survey. No comparable data are available from the bay for flowering plants. The percentage recorded during this survey was lower than 5% flowering plants reported for Plymouth Sound for a comparable July survey in 2018 (Bunker & Green, 2019) but higher than the three plants observed in Torbay in September 2019 (Field, 2019).

Table 4: Number and percentage of flowering seagrass (*Zostera marina*) plants sampled from 0.25 m^2 quadrats within Studland Bay, Dorset in July 2021.

	Site								**Total**
	1	**2**	**3**	**4**	**5a**	**7**	**8**	**10**	
Total plants	464	500	194	338	367	410	904	300	**3477**
Total flowering plants	4	11	1	1	7	8	10	10	**52**
% flowering plants	0.9	2.2	0.5	0.3	1.9	2.0	1.1	3.3	**1.5**

There was no obvious spatial distribution pattern of flowering plants within the bay from the eight sites sampled. Anecdotal observations during the quadrat surveys suggested the occurrence of mature flowering stands of seagrass were highly clumped where they occurred within each sampling location.

3.3.3 Presence of eggs on leaves

Eggs of various mollusc species (including cuttlefish) and potentially from polychaete worms were recorded on seagrass leaves during sample processing. Table 5 shows the total number of plants sampled and assessed from the 200 quadrats surveyed including the number and percentage of those with eggs on the leaves at each sampling site. There was no obvious spatial pattern to the data from the eight sites sampled within the bay.

Table 5: Number and percentage of plants with leaves supporting eggs of other species (mainly molluscs) sampled from 0.25 m^2 quadrats within Studland Bay, Dorset in July 2021.

	Site								**Total**
	1	**2**	**3**	**4**	**5a**	**7**	**8**	**10**	
Total plants	464	500	194	338	367	410	904	300	**3477**
Total with eggs	18	12	11	18	21	17	16	9	**122**
% with eggs present	3.9	2.4	5.7	5.3	5.7	4.1	1.8	3.0	**3.5**

3.4 Seahorse observations

No seahorses were recorded during the 2021 surveys.

3.5 Other incidental species observations

Other incidental species recorded during the survey were noted or photographed but were not quantified in any way due to survey time constraints. These included but were not limited to:

- Snakelocks anemone, *Anemonia viridis*
- Lugworm, *Arenicola marina*
- Sand mason worm, *Lanice conchilega*
- Fan worm, *Sabella pavonina*
- Slipper limpet, *Crepidula fornicata*
- Clam siphons, cf. *Ensis ensis*
- Netted dogwhelk, *Tritia reticulata*
- Snail, *Rissoa* sp.
- Bryozoan, cf. *Scrupocellaria* sp.
- Bryozoan, *Electra pilosa*
- Amphipod tubes
- Shore crab, *Carcinus maenas*
- Decorator crab, *Macropodia sp.*
- Masked crab, *Corystes cassivelaunus*
- Brittlestar, *Ophiura albida*
- Sea squirt, *Aplidium* sp.
- Sea squirt, *Botrylloides* sp.
- Leathery sea squirt, *Styela clava*
- Greater pipefish, *Syngnathus acus*
- Deep-snouted pipefish, *Syngnathus typhle*
- Black goby, *Gobius niger*
- cf. Couch's goby, *Gobius couchi* (site 8 – see Appendix A)
- Sand/common goby, *Pomatoschistus* spp.
- Two-spot goby, *Gobiusculus flavescens*
- Pollack, *Pollachius pollachius*
- Pout / pouting / bib, *Trisopterus luscus*
- Sand smelt, *Atherina presbyter*
- Corkwing wrasse, *Symphodus melops*
- Long-spined scorpionfish, *Taurulus bubalis*
- Conger eel (juv.), *Conger conger*
- Tunicates, *Ascidiella* sp.
- Harpoon weed, *Asparagopsis armata*
- Wireweed, *Sargassum muticum*
- Sea lettuce, *Ulva* sp.
- Coralline algae, *Jania* sp.

- Cuttlefish eggs
- Sponge *Leucosolenia* sp.

Some sites were characterized by particular taxa such as *Rissoa* snails, *Anemonia viridis*, *Scrupocellaria* sp., *Leucosolenia* sp. or Amphipod tubes, all of which could occur in high numbers as epiphytic growth on the seagrass leaves, with patchy distributions throughout the survey locations (Figure 15). Other areas had very clean leaves (see site descriptions in Appendix A).

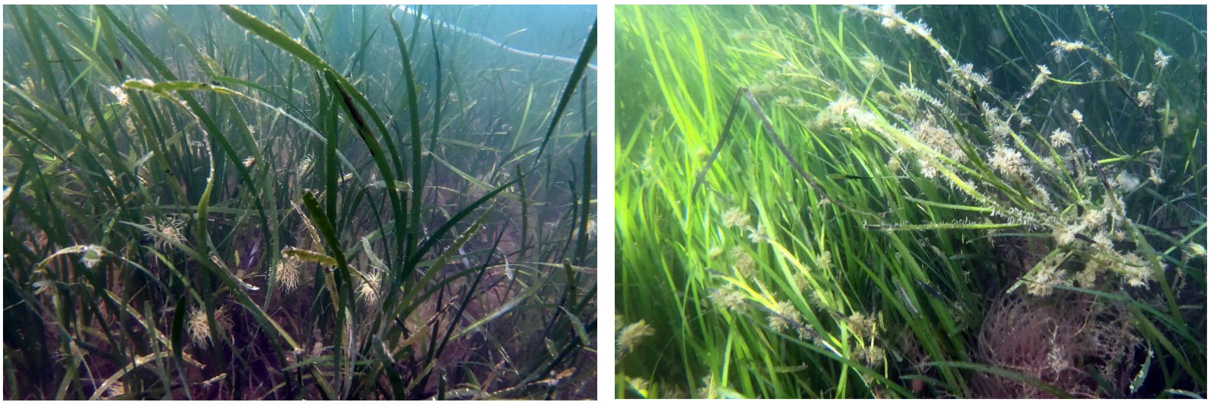


Figure 15: High abundances of snakelocks anemones (*Anemonia viridis*) (left) and bryozoans (cf. *Scrupocellaria* sp.) (right) growing on the seagrass at site 2, Studland Bay, Dorset in July 2021.

The data gathered do not permit any conclusions to be drawn regarding the exact nature of this patchiness or gradation from one area to another, but they do show a diversity of different taxa colonising the seagrass bed. All these taxa were recorded during previous surveys of the seagrass beds in 2009-2011 suggesting some degree (although unquantifiable) of community stability over the preceding 10-12 year period (Seastar, 2012). Given the nature of the community taxa data recorded during this survey and others previously, statistical comparison between the time periods is not appropriate.

3.6 Non-native species

Non-native species were not observed in high abundances during the survey and sampling work but several species were recorded. These included:

- Slipper limpets, *Crepidula fornicata* at sites 2, 5a and 7.
- Sea squirt, *Botrylloides* sp. at sites 2, 3, 5a and 7.
- Leathery sea squirt, *Styela clava* at site 7.
- Harpoon weed, *Asparagopsis armata* (Falkenbergia stage) at sites 4, 7 and 8.
- Wireweed, *Sargassum muticum* at site 7.

Images of each of the non-native species recorded are shown in Figure 16. Seastar (2012) reported the known presence of *C. fornicata*, *A. armata* and *S. muticum* within the bay from its own and previous surveys; the species have also been reported from Seasearch surveys (Seasearch Dorset 2013; 2015). The tunicate (sea squirt) taxa *Styela clava* and *Botrylloides* sp. have been reported from Seasearch surveys in Studland Bay in 2014 along with *Botrylloides diegensis* (Seasearch, 2014). Differences in survey methods and effort do not allow any statistical comparisons of data sets and it cannot be assumed these taxa have not been present in Studland Bay MCZ prior to 2021.

Figure 16: Non-native species observed during the surveys of seagrass in Studland Bay, Dorset, July 2021. Left to right, top to bottom – *Crepidula fornicata*, *Botrylloides* sp., *Styela clava*, *Sargassum muticum* and *Asparagopsis armata* (Falkenbergia).

Anthropogenic influences

Seagrass beds can be sensitive to various anthropogenic activities. Direct measurement of such influences was outside the scope of the monitoring surveys in July 2021 although some can be considered here.

"In subtidal situations, nutrient enrichment may lead to excessive growth of opportunistic epiphytic algal species, or blooming species such as, *Ulva*, *Chaetomorpha* and *Ectocarpus* on seagrass beds, potentially compromising the health and viability of seagrass by overlying and smothering them" (WFD-UKTAG, 2014). Studland Bay has been reported to have nitrogen (N) and phosphorus (P) levels above the global average, with the N:P ratio being highly elevated which suggests a nutrient imbalance at the site (Jones & Unsworth, 2016). Although recognised as anthropogenic in nature (Jones & Unsworth, 2016), identifying the exact source(s) of these elevated nutrient levels was beyond the scope of this study. Light levels in the bay are considered sufficient based on the C:N ratios reported by Jones & Unsworth (2016).

In July 2021, the only direct observable impact obvious to the surveyors was that from boat anchors. There was no overlap of survey quadrats and recognised mooring chain scars within the seagrass. Anchoring in the bay can reach high levels during summer months (see Seastar, 2012) with anchors, chains and swing moorings all having the potential to cause physical damage to the seagrass bed structure. During quadrat sampling at site 8, two surveyors were working immediately adjacent to a recreational craft at anchor (Figure 17). On retrieval of the anchor (after the divers had moved away from the area) seagrass shoots and rhizomes were dislodged. Dwindling air supplies on that occasion prohibited any images from being obtained although the damage was clearly visible to the divers.

Figure 17: Anchor and chain from a recreational craft observed during the survey of site 8, Studland Bay, Dorset, July 2021.

Further recent damage was visible at sites 10 and 4, the latter showing freshly exposed roots and rhizomes above the sediment surface (Figure 18), this was very likely from anchoring activity. At site 4, a furrow at least 6-7 m long through the seagrass and seabed sediment was left behind, very likely resulting from the retrieval and dragging of an anchor (Figure 19). A short video was obtained of part of the furrow observed at site 4 (similar to that observed at site 8). From the video it was possible to create a small 3D model of part of the damaged seagrass to illustrate the structural damage to the seagrass bed (Figure 19); the full model can be viewed via this link to Sketchfab. The cause of exposed rhizome mats observed at site 5a was not readily attributable to any particular activity with absolute certainty although there are few other likely causes of physical disturbance other than anchoring in the area.

Damage to the seagrass widely recognised to occur within Studland Bay originates from both swing moorings and anchor chains as larger vessels rotate around the anchor point causing a chain to scour the seabed. This causes mechanical damage which can uproot seagrass shoots and rhizomes and bury seeds too deep in the sediments for them to germinate successfully (see Collins *et al*., 2010 and Seastar, 2012 for further information). Typically, from swinging boats around a fixed mooring point, the scars left behind in the seagrass can be somewhat circular in nature – several are clearly visible in the Google Earth satellite imagery from 16th July 2021 around fixed moorings (Figure 20). Impacts from the dragging of smaller anchors, or those deployed temporarily, as opposed to permanent anchor chains is likely to cause damage on a smaller scale; the significance and cumulative impact of this is less clear in terms of the overall effect on seagrass bed integrity and patchiness.

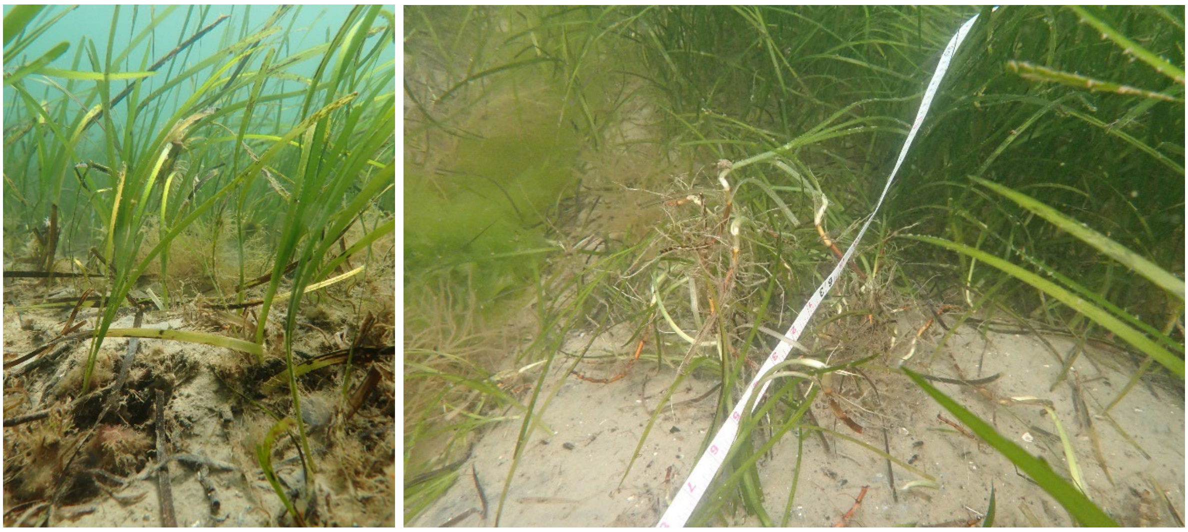


Figure 18: Observations of disturbed shoots, roots, rhizomes and sediments at site 10 (left) and site 4 (right) from anchor dragging through seagrass in Studland Bay, Dorset, July 2021.

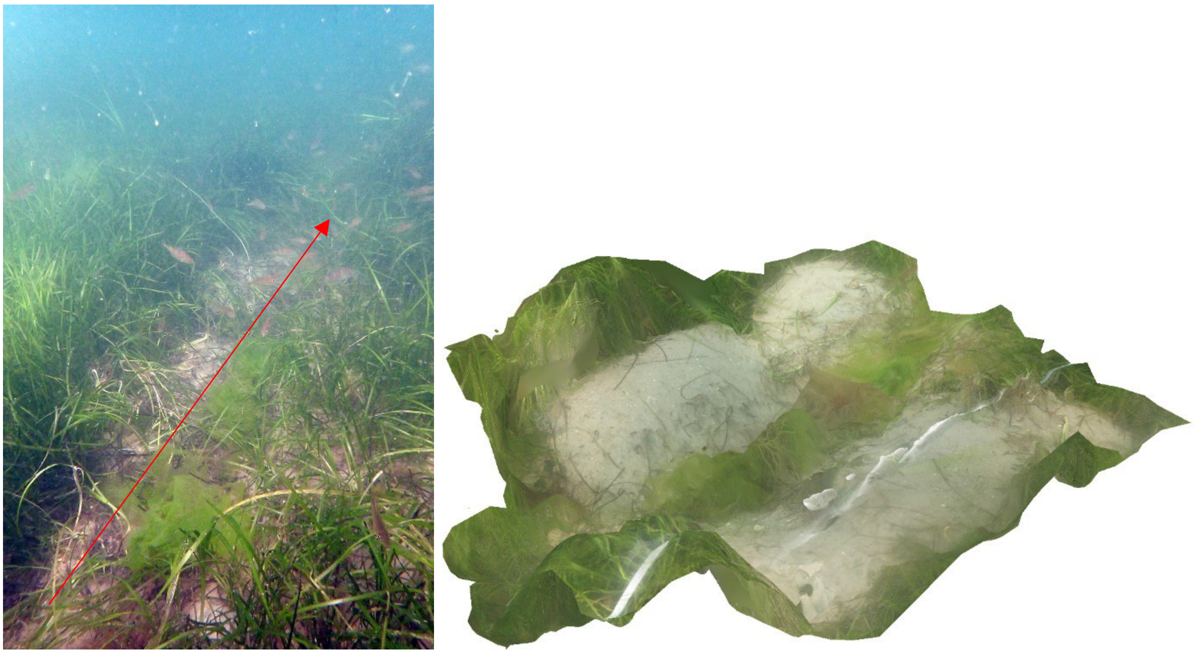


Figure 19: A furrow left behind in seagrass and sediments (likely to be from anchor dragging through seagrass) in Studland Bay, Dorset, July 2021. Images show (left) the narrow furrow extending beyond the limit of visibility (red arrow) and (right) a 3-dimensional model of a section of the furrow (the full model can be viewed via this link to [Sketchfab]()).

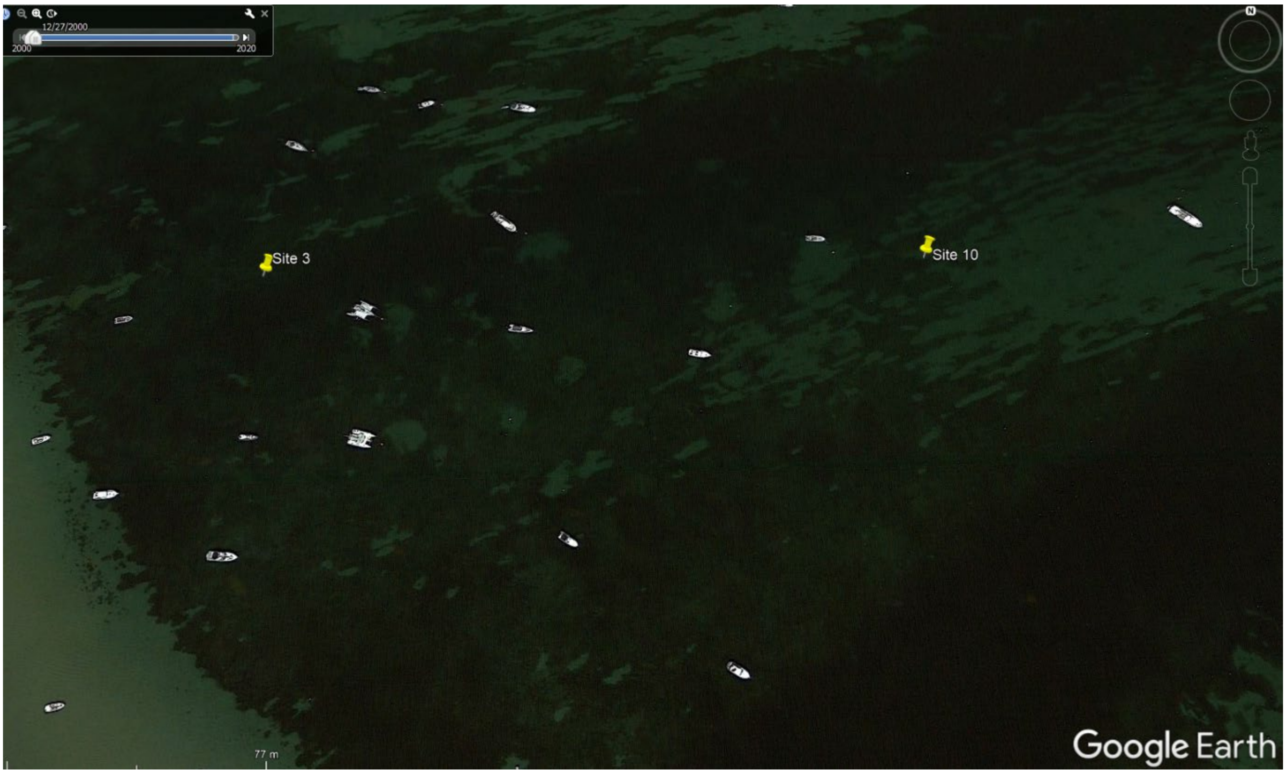


Figure 20: Google Earth satellite image of Studland Bay on 16th July 2021 showing scarring in the seagrass around fixed moorings close to sites 3 and 10 (Source: Google, ©2021 CNES / Astrium, Maxar Technologies).

During the survey of site 5a on 20th July 2021, a snorkeller swam around the survey area gathering video footage around the start point from the surface; this was a relatively *ad hoc* exercise to see if the images obtained could be used to produce a single image of the wider area. From the video it was possible to extract and stitch some (but not all) of the individual frames to give a wider view of the seagrass bed. The technique was semi-successful and could work well with better pre-planning of the area to cover and the use of divers rather than snorkellers. Bare sand patches were visible amongst the seagrass in the model produced (Figure 21). Whilst there were no fixed moorings within the survey area the site was within an area of patchy seagrass where vessels are known to anchor (Figure 22). Whilst it cannot be said with full certainty that these particular sand patches resulted from anchor damage the surveyors did note erosion and exposed rhizomes around some of the seagrass bed edges within site 5a which clearly resulted from some form of direct physical disturbance (Figure 23). Collins *et al*. (2010) describe how such disturbance could lead to increased erosion in a seagrass bed from increased shear stresses around the exposed rhizomes following anchor damage.

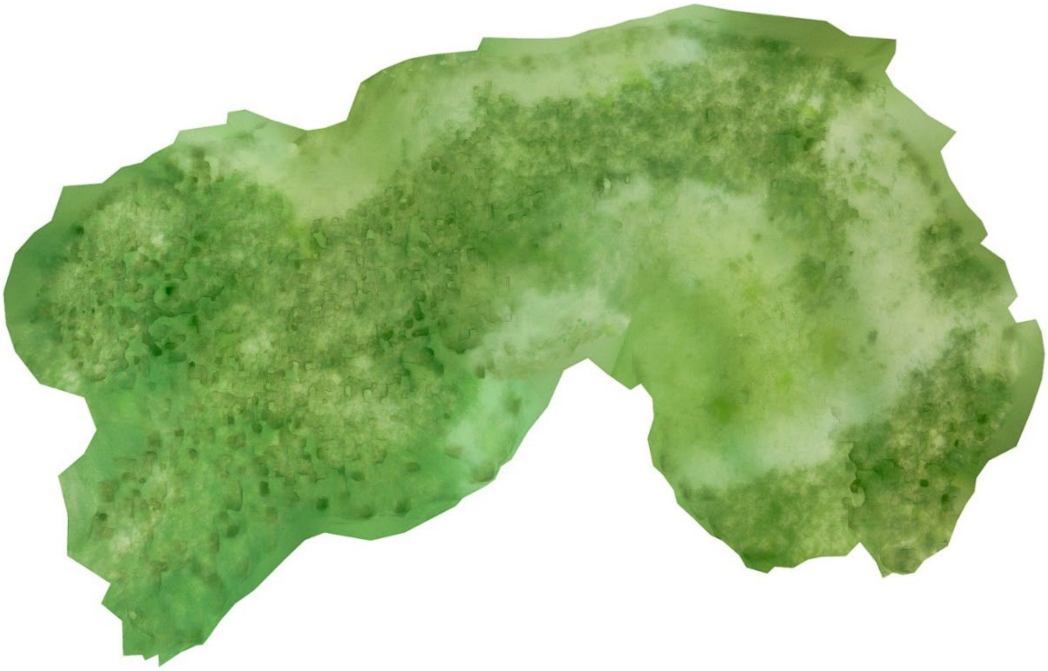


Figure 21: Stitched images from GoPro video stills obtained adjacent to the centre point of survey site 5a in Studland Bay, Dorset on 20th July 2021 (the full model can be viewed via this link to Sketchfab). The image shows areas of continuous seagrass cover and areas of bare sand with scattered dead seagrass and the green alga *Ulva* sp. NOTE: No scale available but the image likely represents an area 5-6 m long x 3 m wide.

Figure 22: Google Earth image of the patchy seagrass bed at site 5a, July 2021. (Source: Google, ©2021 CNES / Astrium, Maxar Technologies. Image dated 16 July 2021).

Figure 23: Exposed rhizomes and dead / dying seagrass at the edge of a bare sand patch at site 5a, Studland Bay, Dorset.

3.8 Statistical Power

As stated in Section 2.5.2, the power analyses of the univariate shoot density data were completed using the MONITOR programme (Gibbs & Ene, 2010), outputs from which indicate the power to detect specified levels of change over a specified time range based on the known or estimated population mean and standard deviation. In the power analysis outputs presented here, the data used were those from the 2021 quadrat surveys. The power analyses were run on the default assumption of wanting a 90% chance of detecting a real change with the significance level of that change set at the standard 5% level i.e. power = 0.9. Where a power level of 0.9 was not achievable, power levels of 0.8 were investigated. These parameters can be altered in future if there was reason to accept the findings of less powerful monitoring designs or to set significance thresholds at lower levels. Although there are no established conventions, it is common practice to seek power estimates exceeding 0.80 (Cohen, 1988), i.e. a monitoring program with power estimates in excess of 0.80 would detect trends, should they occur, >80% of the time. A starting default value of 0.9 desired power is used here to give confidence that any population changes could be captured. “In a monitoring context [a significance (p) level of] 0.1 or 0.2 is perfectly reasonable depending on the seriousness of missing important trends versus the costs of exploring false detections. Justifying significance level is a critical part of designing monitoring programs” (Gibbs & Ene, 2010). Two monitoring designs were entered into the MONITOR interface depending on the hypotheses that might wish to be tested:

- Firstly, data were analysed on the basis of wanting to monitor for time trends in the mean shoot density per sampling site (regression) over three or four survey events during a ten-year period. This was based on the assumption that annual monitoring at the site is unlikely but any future data would still need to be able to determine whether density was increasing or decreasing. In this case a "route regression" approach was used, whereby trends in sample measurements are determined for each site, and then averaged across all sites.
- Secondly, the power analyses were conducted on the basis of comparing the ‘mean shoot density per sampling site’ values on a ‘before / after’ basis (i.e. to determine if potential management measures might be effective) to test a hypothesis that an increase in seagrass density might occur.

3.8.1 Time trends

The ability to detect directional change over time is important to be able to determine if the seagrass bed structure is changing in a positive or negative way. Natural factors such as wave action, temperature and turbidity can influence bed structure in any given year, so the ability to determine directional change must account for this variability and, also contain sufficient time points to be confident that a trend is present. Of the three parameters recorded during the *in situ* quadrat surveys (% seagrass cover, shoot density per m^2 and % algal cover) the shoot density data are the least subjective, being based on actual counts rather than visual estimates. Therefore these data were used for the power analysis as they are not subjected to any (or minimal) recorder variability.

The WFD-UKTAG (2014) document notes the variability inherent in seagrass beds in terms of cover or density may be as high as 30%. Where data allow, it is suggested to use a five-year rolling mean value for shoot density to reduce noise and identify longer term trends; in such cases variation of ~15% is considered “as tolerable evidence of natural variation and decreases in extent of >15% should be viewed suspiciously” since a 30% reduction when using rolling means could mask underlying trends (WFD-UKTAG, 2014). The authors assume “*suspiciously*” suggests changes due to anthropogenic causes, rather than natural variation. On this basis, power analysis was conducted to determine the power to detect positive or negative trends in the Studland seagrass bed shoot density of 15%, 30% or 40% over a period of approximately one decade.

Table 6 shows the power of the present monitoring programme design to detect time trends in the seagrass shoot density data. The changes in shoot density indicated are the overall changes over the full time period. The results suggest the power to detect changes of ±30% or higher is consistently high (>0.9) no matter the sampling interval. The power to detect changes of ~15% over the full 9-11 year period was more variable and required sampling at least every three years to obtain power values >0.8, which may be regarded as less reliable than those >0.9. Unsurprisingly, the more frequent the sampling, the greater the power to detect change.

Table 6: Statistical power to detect trends in shoot density per m^2 of varying magnitude over three or four different survey occasions at the 5% significance level.

Sampling interval	% change in mean shoot density per m^2	Power
Every 5 years, over an 11-year period x3 surveys Yrs 0, 5 and 10	±15% ±30% ±40%	0.723 0.907 0.974
Every 3 years, over a 10-year period x4 surveys Yrs 0, 3, 6 and 9	±15% ±30% ±40%	0.845 0.911 0.980
Every 2 years, over a 9-year period x5 surveys Yrs 0, 2, 4, 6 and 8	±15% ±30% ±40%	0.852 0.931 0.991

Should the seagrass shoot density change by 15% year-on-year, the power of the monitoring design was very high (>0.990) no matter which of the survey interval options were selected, in fact even with three survey events over an 11-year period it would be possible to detect changes of ±10% with power values consistently >0.990.

Whilst changes in shoot density can be informative about the state of seagrass within a bed, it does not provide information on expansion or contraction of the overall bed, particularly given that the eight monitoring sites were selected based on the known presence of seagrass (albeit with a range of densities). Monitoring of the bed extent and boundary by drop camera or multibeam methods is likely to provide the best indication of this parameter although the same considerations must be given to determining actual trends in bed extent from natural variation and viewing anything <15% in either direction as unlikely to be indicative of a trend (WFD-UKTAG, 2014).

3.8.2 Management effects

The second way to assess statistical power is to consider the ability of the present monitoring design to detect 'before/after' effects of potential management measures to protect the seagrass. In this instance there may be one or more years' monitoring data pre-management following by subsequent years post-monitoring. Considering the same monitoring intervals as those in Table 6 with a single 'pre-management' year and two or three 'post-management' monitoring years, the present design only had power values of 0.176 to 0.205 to detect a change of 30% in shoot density.

The 'Optimise' function in MONITOR was used to investigate what combinations of survey interval, numbers of sites and variations of acceptable power and significance might enable some detection of significant change in shoot density following changes to management measures. In summary:

- Ten annual surveys (3 ‘before’ / 7 ‘after’ management measures) have a power of 0.904 to detect a 70% change in density or a power of 0.810 to detect a 50% change in density within the present eight monitoring sites.
- Six annual surveys (3 ‘before’ / 3 ‘after’ management measures) have a power of 0.830 to detect a 30% change in density using 14 monitoring sites at the 10% significance level.
- No other iterations tested of survey frequency, number of monitoring sites and varying the significance level provided any level of power likely to be deemed acceptable in terms of effort and cost.

The low power of the survey design to detect before / after changes of such a magnitude is in line with similar findings for the seagrass beds in Torbay (Field, 2019), likely because of the high variability between sampling sites within the bed. It may be more cost-effective to use annual remote sensing techniques to monitor for changes in bed extent and % cover over time in relation to management changes rather than more frequently repeated diver measures, in this instance. Use of % cover from drop cameras as a proxy for density could be a useful compromise given the cost and time for diver surveys and the smaller areas divers can cover; correlation of density and % cover has been shown to be excellent (Figure 9) albeit both from diver records. Less frequent diver surveys e.g. every five years would still retain the power to detect temporal trends in density which may add confidence to other changes observed in extent and overall % cover. Furthermore, divers can gather useful additional data on non-native species, community composition, and can collect samples to assess overall plant health and size structure.

3.8.3 Sampling efficiency

A final aspect to examine in terms of statistical power was the sampling efficiency of the surveys. From every sampling point, all the shoots within a 0.0625 m^2 quadrat were collected; n = 3,471 from the entire survey. From each shoot the longest leaf length was measured (n = 3,471) before every leaf (n = 14,122) was then assessed for infection and epiphyte growth. As a general rule of thumb, the ideal minimum (or large enough) sample size for most statistical analyses is n = 30. Naturally the processing of these samples and subsequent data entry required considerable time, effort and cost from the survey team with additional staff contracted in to undertake the data entry. Since the number of leaves assessed for infection and epiphytes is not independent of the number of shoots, the optimisation of ‘n’ should apply to the number of shoots assessed per quadrat rather than the number of leaves.

The 3,471 shoots sampled across the 200 quadrats gave an average count of 17 shoots per quadrat. However, the shoots were not equally distributed across quadrats or sample sites with the number per quadrat ranging from 0-59 and the number per site ranging from 194 at site 3 to 904 at site 8. The impact of reducing the number of shoots sampled was investigated in two different ways:

- Shoot data were deleted at random from each quadrat sample to leave a maximum of n_{max} = 20 per quadrat. Naturally some quadrat samples contained fewer shoots than these values and were left unchanged. This approach meant the overall shoot sample size was reduced to 2,647 from the original 3,471 – a reduction of 824 shoots.

- Minitab was used to randomly generate a sample of 1,000 shoot samples from the full dataset. Assuming an equitable distribution of the data, this would average five shoots per quadrat, or 125 shoots per sampling site (still a very high 'n'). In reality the random sample was proportionate to the original distribution of the data meaning the total shoot samples per site ranged between n = 42 at site 3 to n = 280 at site 8.

The data sets for n = 2,647 shoots and n = 1,000 shoots generated by these two selection processes were then compared to the original n = 3,471 shoots by means of 2-tailed *T*-tests to check for significant differences in mean shoot lengths, *Labyrinthula* sp. infection and epiphyte cover for the Studland Bay seagrass bed.

Reduction of the sample sizes to n_{max} = 20 resulted in no significant overall difference of the mean longest leaf length, or the infection score but was marginally significantly different for the mean epiphyte cover score per shoot (Table 7).

Table 7: T-test comparisons of original survey samples to a reduced data set where n_{max} = 20 shoots per quadrat.

Parameter	n	Mean	Stdev	SE	df	*T*-value	*p*-value
Longest leaf length	3,471	37.0	17.2	0.29	6116	0.49	0.625
	2,647	36.8	17.2	0.33			
Infection score	3,471	0.870	0.548	0.0093	6116	-1.80	0.072
	2,647	0.896	0.554	0.011			
Epiphyte score	3,471	0.825	0.559	0.0095	5678	-2.26	0.024
	2,647	0.858	0.562	0.011			

One potential problem with the above approach to sample selection is that if infection rates and epiphyte cover are patchy within the seagrass bed this approach may skew the data toward better representation of less dense areas of seagrass as these samples where $n_{original}$ <20 remain unchanged compared to those with $n_{original}$ >20 and may have been a cause behind the significant difference observed.

Reduction of the original data set to n = 1,000 shoots overall via random sample selection maintains some degree of the original proportion representation of each sampling site in the original dataset, whilst being approximate to 4-5 shoots assessed per quadrat at all bar two sites. Consequently, this approach resulted in no significant differences being found to occur between any of the three parameters compared in Table 8.

Table 8: T-test comparisons of original survey samples to a reduced data set where n = 1,000 shoots across all sample sites.

Parameter	n	Mean	Stdev	SE	df	*T*-value	*p*-value
Longest leaf length	3,471	37.0	17.2	0.29	1605	0.58	0.564
	1,000	36.7	17.4	0.55			
Infection score	3,471	0.870	0.548	0.0093	1601	0.59	0.556
	1,000	0.859	0.556	0.018			

Parameter	n	Mean	Stdev	SE	df	*T*-value	*p*-value
Epiphyte score	3,471	0.825	0.559	0.0095	1619	0.93	0.353
	1,000	0.806	0.558	0.018			

Not only did this approach to reducing sample size to n = 1,000 result in no overall change to the population estimates of these parameters but the same patterns of spatial differences/similarities between the sites remained as apparent as with the original data where n = 3,471 (Figure 24). The same non-significant results were obtained even when the sample size was reduced to just 240 shoots overall (approximately 30 shoots per sample site, or 1-2 per quadrat on average) (Table 9).

Table 9: T-test comparisons of original survey samples to a reduced data set where n = 240 shoots across all sample sites.

Parameter	n	Mean	Stdev	SE	df	*T*-value	*p*-value
Longest leaf length	3,471	37.0	17.2	0.29	270	0.76	0.449
	240	36.1	17.9	1.2			
Infection score	3,471	0.870	0.548	0.0093	283	1.42	0.156
	240	0.824	0.482	0.031			
Epiphyte score	3,471	0.825	0.559	0.0095	271	1.27	0.204
	240	0.776	0.571	0.037			

When developing the index to assess and monitor wasting disease in *Zostera marina*, Burdick *et al*. (1993) suggested that “for the population estimate to fall within 1 standard error of the mean of 20 shoots more than 95 % of the time, 14 shoots must be indexed.” Although they based this statement on the use of percentage data rather than the 0-5 scale, it is perhaps not surprising that sample sizes of 1,000 or even 240 were more than adequate in the present study to represent the parameters of leaf length, infection score and epiphyte cover score for the Studland Bay seagrass bed.

For this approach of processing fewer shoot samples to be representative of the Studland Bay site in subsequent surveys, all shoots still need to be collected from each 0.0625 m^2 quadrat to avoid any sampling bias, i.e. only selecting the larger, easier shoots to cut. Then a random approach to shoot selection post-dive needs to be applied before measurements and infection / epiphyte assessments are undertaken. The random selection of five shoots per quadrat sample would likely generate a total sample size somewhere between 500 and 1,000 shoots with only a maximum of 40-45 shoots to process per sampling site per diver pair post-dive, being more than powerful enough to provide a confident representation of the plants present in the bay. Further benefits include divers being less fatigued each day and substantial cost savings can be made as it will be unlikely that additional staff will be required for data entry and to assist with sample processing.

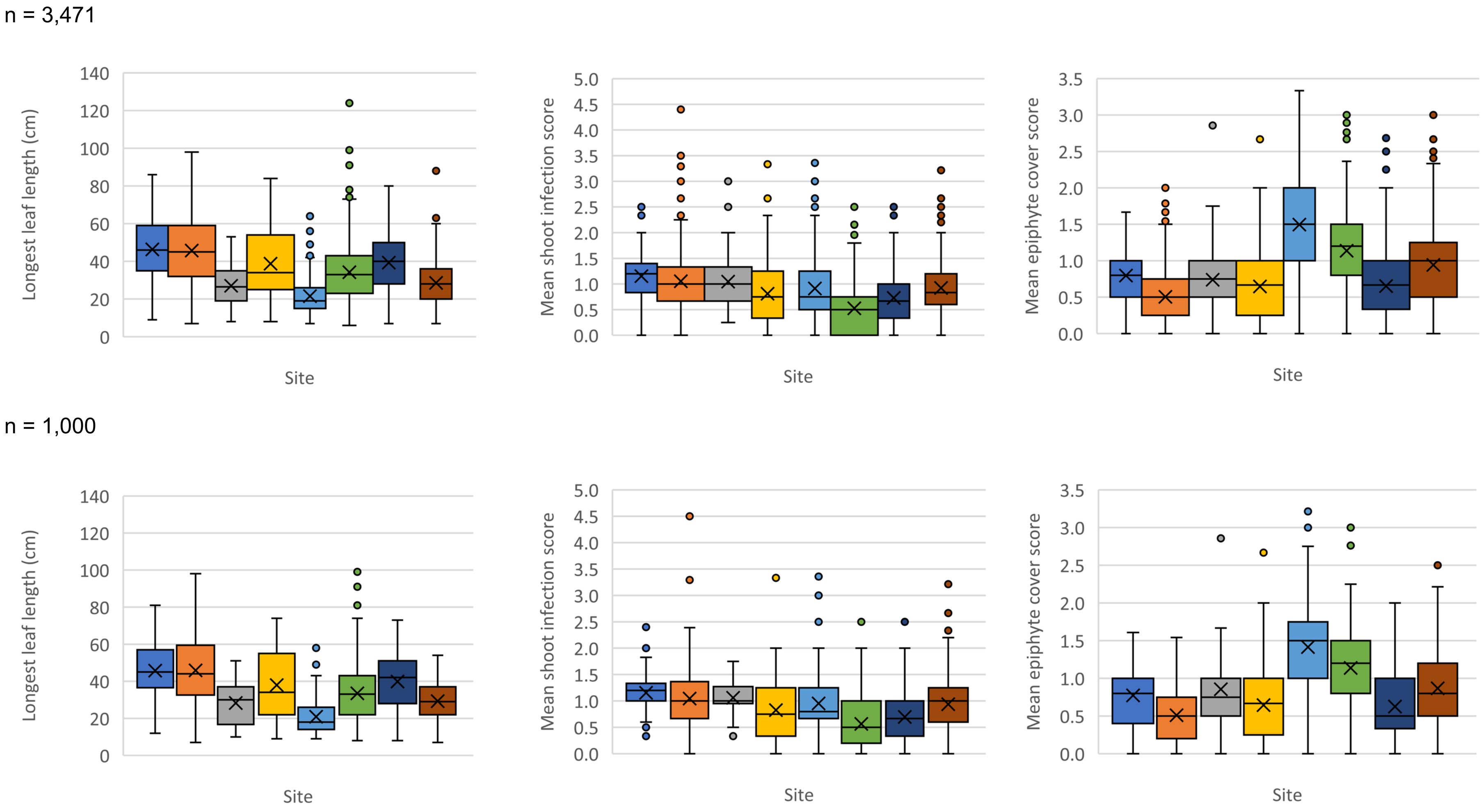


Figure 24: Boxplots of longest leaf lengths per shoot, mean shoot infection scores and mean epiphyte cover scores across all eight sampling sites in Studland Bay, Dorset in July 2021. Top plots show data for all shoots processed (n = 3,471). Lower plots show data for a randomised subsample (n = 1,000) from the original data collected, showing near identical spatial patterns across the sites to the data above. Boxes show mean (x), median (-), interquartile ranges (boxes), whiskers and outliers.

3.9 Effectiveness of data collection methods, techniques and technical equipment

The methods used in the present survey to gather data on the parameters presented in Sections 3.2 and 3.3 are considered effective in providing a robust baseline, comparable with future monitoring events. A methods-review and quality assurance session prior to the start of the field work combined with post-survey reviews of the data collected early in the survey week was fundamental in ensuring all tasks were fully understood and undertaken consistently by the survey team members.

The in-water work during the survey was straightforward and the equipment required for the tasks was practical to use. Pre-dive briefings for each dive pair and careful preparation of sample bags and distance / bearing information meant each diver pair could complete their tasks easily and without error.

4 Condition assessment

To date, no formal condition assessment has been undertaken for the Studland Bay MCZ by Natural England, primarily due to the site only being designated in 2019. As part of this report the authors were asked to make any comments regarding their assessment on the condition of the 'subtidal seagrass beds' feature of the MCZ, based on the following attribute targets:

Table 10: Designated features, their attributes and conservative objectives in Studland Bay, Dorset that can be assessed in full or in part using data from the 2018 and 2021 seagrass surveys.

Designated feature	Attribute	Conservation Objective
Subtidal seagrass beds *Zostera marina/angustifolia* beds on lower shore or infralittoral clean or muddy sand	Extent: presence and spatial distribution. Structure and function: quality and composition[1] of characteristic biological communities[2] including diversity and abundance of species forming part of or inhabiting the habitat. Represented by species composition of characteristic biotope SS.SMp.SSgr.Zmar.	Maintain in or bring into favourable condition. Where favourable condition means: **Extent** is stable or increasing. **Structure and functions**, quality and biological community composition are sufficient to ensure that its condition remains healthy and does not deteriorate Maintain species composition - presence and abundance of composite species should not deviate significantly from baseline.
Spiny (long-snouted) seahorse, *Hippocampus guttulatus*	Habitat: quality and quantity. Population structure: population number, age and sex ratio.	Maintain in or bring into favourable condition. Where favourable condition means: **Habitat**: as above for subtidal seagrass beds so as to enable the species to thrive. **Population**: the population within the MCZ is supported in numbers which enable it to thrive by maintaining the number, age and sex ratio of its population.

[1] Species composition of communities includes a consideration of both the overall range of species present within the community, as well as their relative abundance. Species considered need not be restricted to sessile benthic species but could include mobile species associated with the benthos.

Species composition could be altered by human activities without changing the overall community type. Within each component community, species composition and population structure should be taken into consideration to avoid diminishing biodiversity and affecting ecosystem functioning within the habitat (JNCC 2004).

[2] For the purpose of assessing the condition of the seagrass feature, the 'communities' are described as biotopes using the Marine Habitat Classification (Connor *et al*., 2004).

The survey methodology devised by Natural England for the 2021 monitoring programme was the first since the designation of the Studland Bay MCZ. No directly comparable data sets exist and consequently comparisons with previous work are qualitative in nature. The same applies to the seagrass bed extent data collected by the Environment Agency in 2018 (Green, 2018) just prior to MCZ designation. Both data sets provide a baseline against which quantitative comparisons can be made in future to determine any changes in habitat and community parameters over time.

4.1 Anthropogenic impacts

Section 3.7 describes the anthropogenic impacts observed during the monitoring survey which included varying degrees of damage from moorings and anchors. Reports from previous studies also highlighted potential nutrient imbalances within the bay with nitrogen levels above the global average (Jones & Unsworth, 2016). Jones *et al*. (2018) scored a number of seagrass beds on a scale of 0-5 in terms of 'anthropogenic influence' and subsequent 'perceived health score' with a score of 5 being the most impacted. Industry, tourism, agriculture, catchment and population were all factors recognised to result in higher nitrogen levels (measured in the leaf tissue). Of the 11 seagrass beds sampled throughout the UK, the Studland Bay seagrass bed had the second highest nitrogen loading and scored '4' for anthropogenic impact level and was given a 'moderate' health status in light of these anthropogenic impacts (Jones *et al*., 2018). The study did not attempt to identify the precise causes of high N levels at each bed but stressed the need for site-specific actions to address the sources of nutrient pollution within each site's management plan.

A habitat protection strategy is currently in place to address the issue of anchor and mooring damage within the seagrass bed (MMO, 2021).

Raw sewage has previously been reported to overflow, on occasion, down the steam that flows directly out into South Beach (Garrick-Maidment, 2020). The authors are uncertain whether this remains an issue.

4.2 Extent: presence and spatial distribution of seagrass bed habitats

The 2018 EA seagrass extent data also compare well visually with Google Earth satellite imagery from July 2018 (Figure 25). The equivalent imagery from July 2021 appears to show a lower and more patchy area of seagrass cover compared with 2018 (Figure 25). Earlier Google Earth data are not of sufficient quality to enable further comparison but imagery from the Channel Coast Observatory suggests that the seagrass extent in the bay in June 2013 might have been greater still, raising the question as to whether the bed extent might have reduced since that date.

However, aerial imagery such as that in Figure 25 can be taken as indicative of seagrass extent **only** if it is ground-truthed at or very close to the time of image capture; dark areas considered to be seagrass could in fact be other algae growing in the area or washed in by tidal / wind action. Therefore, in isolation aerial imagery may not reliably demonstrate changes in seagrass extent and gives no information on shoot health and the associated community. Video drop down, and side scan sonar methods provide a greater certainty compared to aerial photographs. Ground-truthing combined with aerial imagery can give a very accurate measure of seagrass extent and distribution. Although the satellite images in Figure 25 originate from comparable June / July dates between 2013 and 2021 (minimising the potential for seasonal variation), further ground-truthing surveys were not undertaken to ensure are advised to ascertain whether or not this decline is real, perceived or simply part of interannual variation across the site. Nonetheless, the data suggest the bed extent should be monitored carefully to improve understanding of any trends in changes of its extent.

27th June 2013

6th July 2018

16th July 2021

Figure 25: Google Earth satellite imagery of Studland Bay, Dorset in June 2013 (top), July 2018 (middle) and July 2021 (bottom). 2021 monitoring sites are marked with the yellow pins. Areas of likely seagrass show up as darker areas against the sand (Sources: TOP – Channel Coast Observatory, ©2021 NNRCMP CCO National Network of Regional Coastal Monitoring Programmes. Reproduced under the Open Government licence. MIDDLE & BOTTOM - Google, ©2021 CNES / Astrium, Maxar Technologies).

The 2018 Environment Agency drop camera survey calculated the seagrass area with >5% cover as 82.26 ha. in Studland Bay (see Section 3.1). The area of 1-5% cover was calculated as 19.14 ha. (Green, 2018). These extents were greater than those estimated from previous surveys (Collins, 2010; Pearce, 2009 – both cited in Green, 2018) but cannot be compared directly due to the different survey areas covered (larger in 2018). The Finding Sanctuary Final Report (p382, Table II.3.15d) cites the area of seagrass in the bay as 91 ha. but does not provide specific information on survey dates or coverage and whether this applies only to areas with >5% cover (Lieberknecht *et al*., 2011). Even if the Finding Sanctuary data did not include seagrass at <5% cover the difference between the figures cited of 91 ha. and 82.26 ha. is less than the threshold decline of 15% that is recommended to be viewed "*suspiciously*" (WFD-UKTAG, 2014). Quantifying the level of potential change between 2013 and 2021 as suggested by Figure 25 was beyond the scope of this report. Whilst such a process would inevitably suffer from a lack of ground-truthing, the images suggest a possible decline in the seagrass bed extent has occurred, particularly in the northern part of the bed; further investigation would be justified and the EA seagrass extent and percent cover surveys planned for 2022 will help toward addressing this question.

Based on the information above, the authors do not feel it is possible at this stage to say whether the extent and distribution of the seagrass within Studland Bay has been maintained.

4.3 Structure and function: quality of seagrass bed habitats

Notable community: *Zostera marina*, seagrass beds - SS.SMp.SSgr.Zmar

Feature target: Maintain the habitat structure, functions and quality in (or bring into) favourable condition

Feature outcome: Structure, functions and quality are sufficient to ensure its condition remains healthy and does not deteriorate.

Density and percent cover values from previous surveys cannot be compared statistically due to the different methods used. Considering this, the values cited between surveys are broadly in range with the current study, suggesting that the overall density / percent cover of the seagrass, where it was surveyed, has not changed significantly and has likely been maintained. Further surveys would be required to determine trends in data either way and give higher confidence to this assertion which cannot currently be supported with statistical evidence. The planned extent and % cover survey in 2022 will help to address this question.

The EA survey (Green, 2018) of the bed extent and density suggests approximately 40-50% of the seagrass bed area at Studland may be below the density needed for healthy resilience, i.e. it is below the optimal level (beds with patchy and sparse areas creating cover <60%) to provide resilience to storms, disease or other natural / anthropogenic impacts (Borum *et al*., 2004). This in turn could lead to lower overall biodiversity (McCloskey & Unsworth, 2015). However, the main areas of this lower density are throughout the outer reaches of the bed and may be a natural degradation in density as the water depth or substrate changes, becoming sub-optimal for seagrass growth. The continued regular monitoring of extent and % cover will enhance understanding of any dynamics associated with these parameters and of any implications for seagrass health and bed quality.

Data on algal cover and leaf length compared favourably with that from previous surveys, albeit using different survey methods. No historical site data were available to make comparisons for infection levels and the number of flowering plants. The overall feel for the seagrass bed health, structure and function was good, with tall, dense luxurious stands of seagrass observed frequently throughout the fieldwork. Nonetheless, damage to the bed from moorings and anchors remained evident with disturbed patches observed throughout the areas surveyed; the nature of this specific survey meant that these disturbed areas were not quantified or assessed directly.

Despite the areas of healthy seagrass observed, the anthropogenic impacts listed above and in Section 4.1.1 (with their potential to disrupt processes such as carbon sequestration and coastal protection) coupled with the potential decline in bed extent suggests that overall the habitat structure, function and quality **needs to be brought into favourable condition**. The Habitat Protection Strategy (MMO, 2021) aims to achieve this.

4.4 Structure and function: species composition of component communities

Notable community: *Zostera marina*, seagrass bed biological community

Feature target: Maintain the community composition

Feature outcome: Composition of the characteristic biological communities (including diversity and abundance of species forming part or inhabiting the habitat) are sufficient to ensure that its condition remains healthy and does not deteriorate.

The biological community composition was not recorded in sufficient detail in the present survey to enable any statistical or quantitative comparisons with previous data, themselves often offering only limited and semi-quantitative accounts of species and non-native species within the seagrass bed (see Sections 3.5 and 3.6). The JNCC biotope description has many taxa not recorded in this study or from other surveys (JNCC, 2015). The abundance of non-native species observed during the 2021 surveys does not presently appear to be having a negative impact but no formal assessment of their abundance and distribution has been undertaken. Comparison with previous survey reports from divers in Studland Bay shows similar community taxa recorded over the preceding 10-12 years, suggesting some degree of **community stability has been maintained**. Without comparable quantitative or semi-quantitative data there can only be low confidence in this assertion.

4.5 Condition of feature: spiny, long-snouted seahorse, *Hippocampus guttulatus*

Notable species: Spiny, long-snouted seahorse, *Hippocampus guttulatus*

Feature target: Maintain in, or bring into favourable condition

Feature outcome: Maintain the quality and quantity of its habitat and maintain the number, age and sex ratio of its population, allowing the species to thrive

Habitat outcome – see Sections 4.1.2 to 4.1.4 above.

Population structure outcome – the assessment of the seahorse population levels and structure was beyond the scope of the present study and the data collected. No seahorses were observed during the 2021 seagrass survey.

Seahorses are recognised as being more likely to occur closer to shore within Studland Bay in areas deemed too shallow for the 2021 diving survey work. Regular surveys are undertaken under licence by the Seahorse Trust, who have reported the frequency and distribution of records within Studland Bay to Natural England (Garrick-Maidment, 2020). The Seahorse Trust considers the spiny seahorse population has declined over the last decade, with a perceived increase recorded in 2020. The report suggests a significant increase in seahorse sightings in 2020 compared to previous years but would benefit from applying corrections for survey effort and methodology to make the interannual data comparisons more robust. For example the number of survey hours undertaken each year (as cited in the report) varied between 14 and 302. The reported increase in sightings to 111 in 2020 represented 46 individual seahorses; this was a slight increase from the 40 individuals recorded in 2010. Comparison of these records would be helped further by applying adjustments for survey effort. Garrick-Maidment (2020) also describes the different survey methods used, stating each one to be highly successful. An assessment of the different efficacies of these methods would be beneficial and could be used to design an on-going monitoring programme using consistent or comparable methods with future sightings data adjusted per unit effort.

Although details on population structure are not provided by Garrick-Maidment (2020), it seems likely they would exist since most of the seahorse sightings aim to record the animal’s gender. On the basis of the data available to the authors, it is **not possible to state with certainty** whether the seahorse population structure in Studland Bay is stable, improving or declining. On-going collection of robust, effort-adjusted data using calibrated methods will help inform on any population trends or cycles in abundance that might occur.

4.6 Non-native species throughout the MCZ

The non-native species (NNS) recorded during the 2021 surveys are listed in Section 3.6. Quantitative records were not made so only their presence was noted, although in some areas of patchy seagrass certain taxa were relatively noticeable e.g. *Asparagopsis armata* (see Appendix A, Site 7). The tunicate (sea squirt) taxa *Styela clava* and *Botrylloides* sp. from the 2021 surveys have been reported from Seasearch surveys in Studland Bay in 2014 along with *Botrylloides diegensis* (Seasearch, 2014). All the NNS recorded in Studland Bay have been recorded elsewhere in Dorset previously (author’s pers. obs.).

Differences in survey methods and effort do not allow any statistical comparisons of data sets to determine whether or not there are increasing or decreasing trends of any of the NNS; a reliable assessment of overall change is unfeasible.

As these NNS were either already or very likely already present at the site a target of ‘**reduce**’ is recommended to be set for *non-native species* in the MCZ.

5 Future Survey Plans & Recommendations

Although the aerial and satellite images in Figure 25 originate from comparable June / July dates between 2013 and 2021 (minimising the potential for seasonal variation), further ground-truthed surveys are advised to ascertain whether or not any possible decline in seagrass extent or cover that they suggest is real, perceived or simply part of interannual variation across the site. As outlined in Section 4.1.2 the aerial images alone do not provide sufficient evidence to suggest one way or another if significant changes were occurring between 2013 and 2021. The planned repetition of the 2018 Environment Agency survey (Green, 2018) in 2022 and every three years thereafter will go some way toward monitoring the bed status in a comparable and robust manner.

As indicated in Section 3.8.2, remote sensing and drop-camera techniques may also be a more cost-effective method to monitor for changes in bed extent and % cover over time in relation to management changes rather than more frequently repeated diver measures, in this instance. Less frequent diver surveys, e.g. every five years, would still retain the power to detect temporal trends in density which may add confidence to other changes observed in extent.

Whilst the seagrass surveyed in 2021 was generally considered to be in good condition, signs of physical damage were apparent to all the surveyors. The present survey programme does not address the issue of small-scale patchiness (<50 m resolution) in sufficient detail to understand the potential impacts from physical disturbance caused by anchors, moorings or storm events. Consideration could be given to the idea of surveying transects to produce and monitor patchiness ratios at selected points throughout the bed. Alternatively a ‘condition ratio’ could be developed whereby patchiness is measured and combined with condition scores to produce a comparable metric which could be used to trigger management action if required.

Within-year monitoring could also offer valuable information on the dynamics of physical impacts to the bed and the speed of any subsequent recovery. Transects similar to or the same as those described above could address this and / or the capture of georeferenced aerial images could be trialled to monitor any seasonal changes. Anchor and mooring damage within the bay remain a contentious issue with some stakeholders with regard to the level and / or significance of such impacts. Aerial (drone) surveys could be completed quickly and efficiently several times over the spring and summer to map the level and extent of the present impacts and, over time, record the efficacy of any management policies introduced. Drone surveys can be carried out relatively quickly, causing minimal disturbance to beach and water users and offer a very cheap and effective method to quantify the extent of any impacts. Again, the data would assist with the site condition assessment.

As stated, the main focus of the surveys was to gather high-quality, quantitative data on the main parameters presented in Sections 3.2 and 3.3, and this was achieved. The data and observations presented in sections 3.4 to 3.7 should be considered incomplete and qualitative only however. Once the main survey tasks had been achieved during each dive, there was little additional time available for divers to make any detailed or semi-quantitative records of the biological communities associated with the seagrass beds. Obviously, a survey focussed on the main seagrass parameters in question cannot capture a full picture of the seagrass community and infaunal habitats. More thorough data collection in this regard however could provide a better picture of community dynamics and distribution throughout the bed (including for non-native taxa), would provide better evidence for assessing whether communities have or have not changed over time and would assist with the overall site condition assessment. Careful design of such transects could allow for statistical interpretation of the data both within and between monitoring events. Consequently, it might be of benefit to dedicate either an additional diver pair to each survey to carry out a Phase II MNCR-style survey around each monitoring station or allocate one survey day of the field week for all divers to survey transects within the bed to record community and habitat spatial variation.

The 3D photogrammetry techniques outlined in Section 3.7 with regard to recording anchor and mooring damage could be a useful consideration for demonstrating the physical disruption to the seagrass beds caused by these activities. With careful planning, this element could be factored into the present survey schedule with minimal disruption.

The adoption of measuring and assessing fewer shoots from the 0.0625 m^2 quadrats as outlined in Section 3.8.3 is highly recommended. The benefits to the project in terms of both budget and diver fatigue are not to be underestimated whilst having no impact on the quality of the data collected. Furthermore, this approach would free up resources, perhaps for allocation toward additional drop camera or aerial surveys as mentioned above.

It would be useful in future surveys to note for each quadrat whether it was associated with physical damage to the seagrass bed from a perceived or verified source.

6 Conclusions

Recommendations for future surveys are provided in Section 5.

Statistical comparisons of the 2021 and 2018 data cannot be made against previous surveys owing to differences in methodologies between surveys.

The current data on extent, distribution, density, percent cover, leaf length and shoot health provide a robust baseline against which to make quantitative comparisons in future monitoring surveys.

Available data from drop-camera and remote sensing surveys (Green, 2018) compare well with Google Earth satellite imagery from the same year. Further surveys are recommended to ascertain whether or not the extent and percent cover of the seagrass beds is changing, as suggested (but not proven) by aerial and satellite imagery since 2013.

Composition data for the biological communities associated with the seagrass beds in 2021 are considered incomplete, since this was not the main aim of the 2021 monitoring. Many of the biological taxa associated with the seagrass bed have been recorded in previous surveys suggesting some degree of community stability; without comparable data there can only be low confidence in this assertion.

Anthropogenic impacts from small boat anchors and permanent non-eco moorings were evident throughout much of the survey area as a mix of large bare sand patches, ‘ploughed’ furrows caused during anchor retrieval or loose clumps of seagrass plants (including rhizomes) above the substrate. Some of these impacts were directly observed to occur during the survey work.

Regarding pollution impacts, the nitrogen:phosphorus ratio in Studland Bay has been reported as being highly elevated, suggesting a nutrient imbalance at the site whilst light levels in the bay are considered to be sufficient (Jones & Unsworth, 2016). Further investigation of these parameters was beyond the scope of this study.

No seahorses were observed during the present surveys. The latest available report (Garrick-Maidment, 2020) suggests a recent recovery in their numbers in the bay. Further data and more rigorous analyses are required to determine whether or not this was a real and / or temporary increase. Only low to medium confidence should be attributed to the increased numbers recorded in 2020.

Further data are required to determine the current status of the designated features for the Studland Bay MCZ with high confidence. Only low confidence can be given to the assertion that the biological community remains stable and that the extent of the seagrass bed has not changed significantly.

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Appendix A – Site Descriptions

Description of the habitats/biotopes monitored

Studland Bay is a shallow embayment, sheltered from prevailing south westerly winds but more exposed to the east. It is an area of shallow sandy seabed which supports extensive areas of subtidal *Zostera marina* seagrass beds. It is a shallow site with maximum depths ~4 m BCD and subject to weak tidal streams.

All the sites surveyed represented the *Zostera* biotope: SS.SMp.SSgr.Zmar *Zostera marina*/*angustifolia* beds on lower shore or infralittoral clean or muddy sand (JNCC, 2015).

The following sections provide summaries of each site comprised of various observation made by the surveyors during the field surveys.

Site 1

- The site is situated in the southern part of Studland Bay in an area of dense *Zostera marina* providing 40-80% cover over muddy sand. 1.7-2.0m below chart datum.
- Long *Zostera* blades (mean = 46.4 cm, max = 86 cm). <1% observed to be flowering.
- Many algae caught/growing in understorey of the *Zostera* plants.
- Many snakelocks anemones *Anemonia viridis* attached to the *Zostera* throughout the bed.
- No non-native species were observed.

Figure 26: Typical view of snakelocks anemones (*Anemonia viridis*) in the seagrass bed at site 1, July 2021.

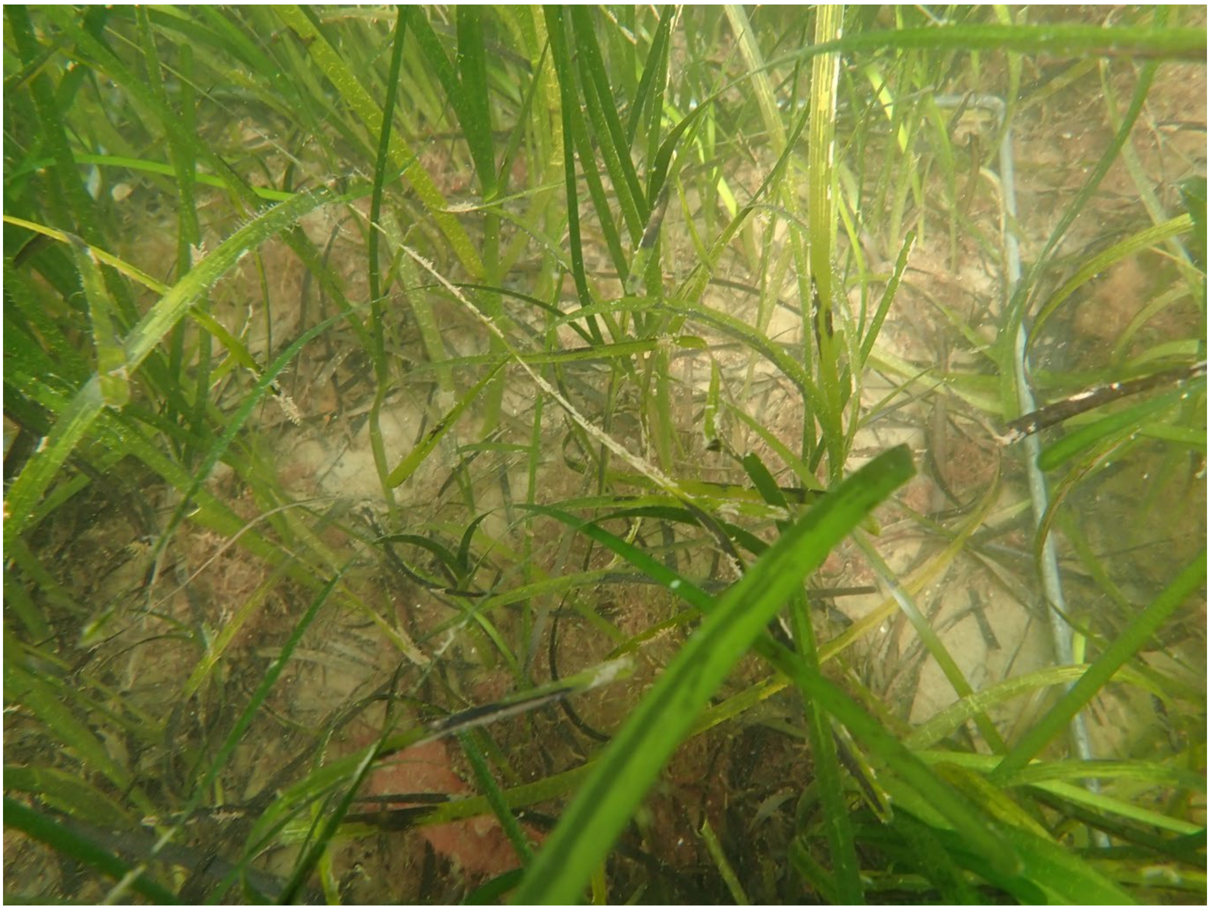

Figure 27: Seagrass and algae within a sampling quadrat at site 1, July 2021

Site 2

- The site is situated in the southwest corner of Studland Bay in an area of dense luxuriant *Zostera marina* providing 60-100% cover over sand and muddy sand (one surveyor noted a grey subsurface element suggesting reduced oxygen levels in the sediment). 0.4-0.8m below chart datum.
- Long *Zostera* blades (mean = 45.7 cm, max = 98 cm).
- Highest algal cover of all the sites with a mean of 62% cover caught/growing in understorey of the *Zostera* plants - *Jania* sp. particularly noticeable.
- Note on algal understorey: it was dense on this site and owing to the lack of solid substratum, very likely to be free-living. Unlikely to be drift algae because it was (for the most part) held too securely by the shoot bases of adjacent tall *Zostera*, plus the attachment of various clumps to squirts, occasional *Crepidula* and dead shells.
- Many snakelocks anemones *Anemonia viridis* attached to the *Zostera* throughout the bed and bryozoans, cf. *Scrupocellaria* sp.
- Non-native species observed was the sea squirt, *Botrylloides* sp. and the slipper limpet, *Crepidula fornicata*.

Figure 28: Topside view from site 2, July 2021

Figure 29: Bryozoans cf. *Scrupocellaria* sp. and flowering seagrass at site 2, July 2021

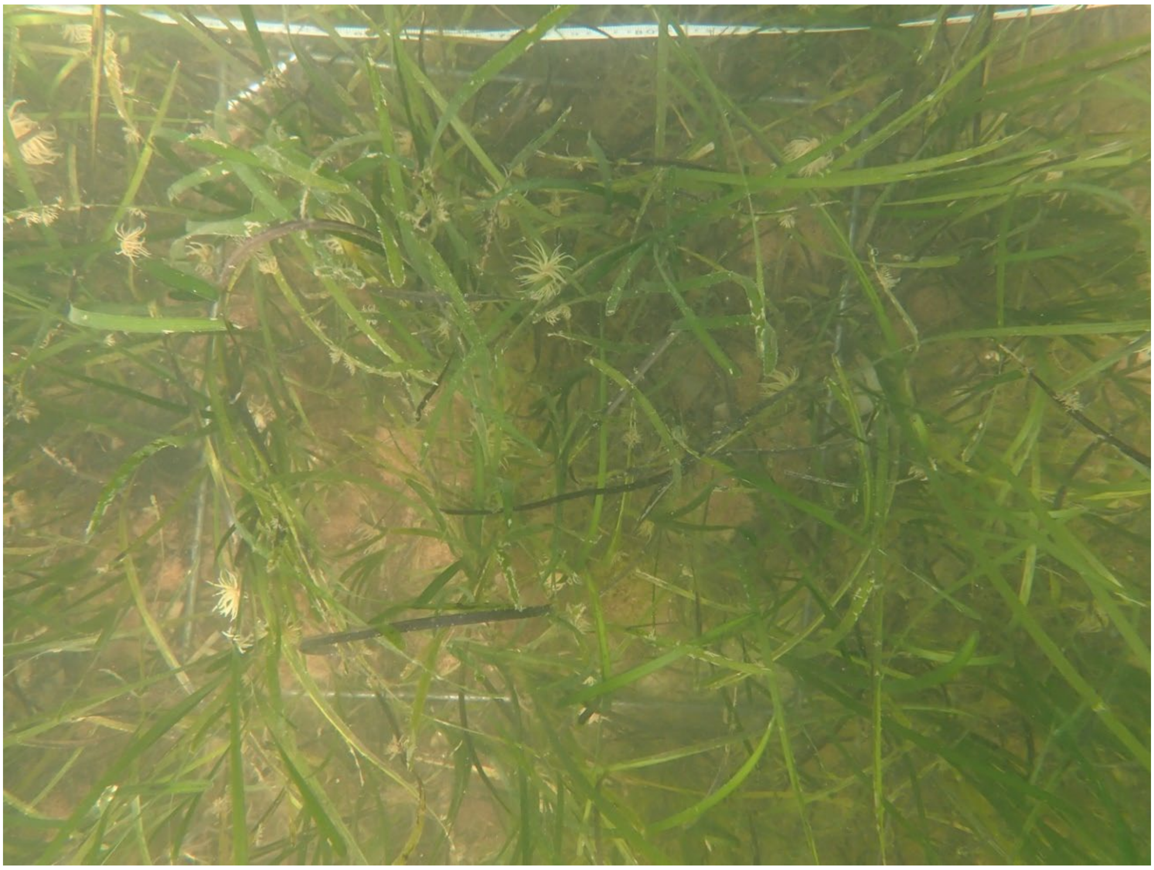

Figure 30: View of a quadrat with some leaves infected with *Labyrynthula* sp. at site 2, July 2021

Site 3

- The site is situated inshore in the southern part of the bay in an area of fairly dense *Zostera marina* providing 25-80% cover over sand and fine sand. 0.6-0.9m below chart datum.
- Short *Zostera* blades (mean = 27.0 cm, max = 53 cm).
- Low algal cover with a mean of 9% cover caught/growing in understorey of the *Zostera* plants.
- Some snakelocks anemones *Anemonia viridis* observed attached to the *Zostera* throughout the bed but fewer than sites 1 and 2. Frequent 'balls' of the sponge *Leucosolenia* sp. often alongside *Scrupocellaria* sp.
- In patches the gastropods *Rissoa* sp. were superabundant on the seagrass leaves. *Tritia* eggs were also seen.
- The gastropod *Hinia* sp. was frequently seen, and patches of the red alga *Gracilaria* sp. were prominent.
- The *Zostera* blades appeared 'cleanish' but black patches of *Labyrynthula* sp. were notable.
- Non-native species observed was the sea squirt, *Botrylloides* sp.
- Infaunal species such as the sand mason *Lanice conchilega* and lugworm mounds of *Arenicola marina* were present and obvious in areas of less dense seagrass. One area contained many *Arenicola* mounds and troughs with no *Zostera*. The *Arenicola* appeared to displace the *Zostera* and the furrows were filled with detrital *Zostera* leaves. In quadrat 4 an area of extensive *Arenicola* mounds changed the

whole appearance of the seagrass bed. Here, the seabed was furrowed and silted – the *Zostera* had long stems covered with amphipod tubes.

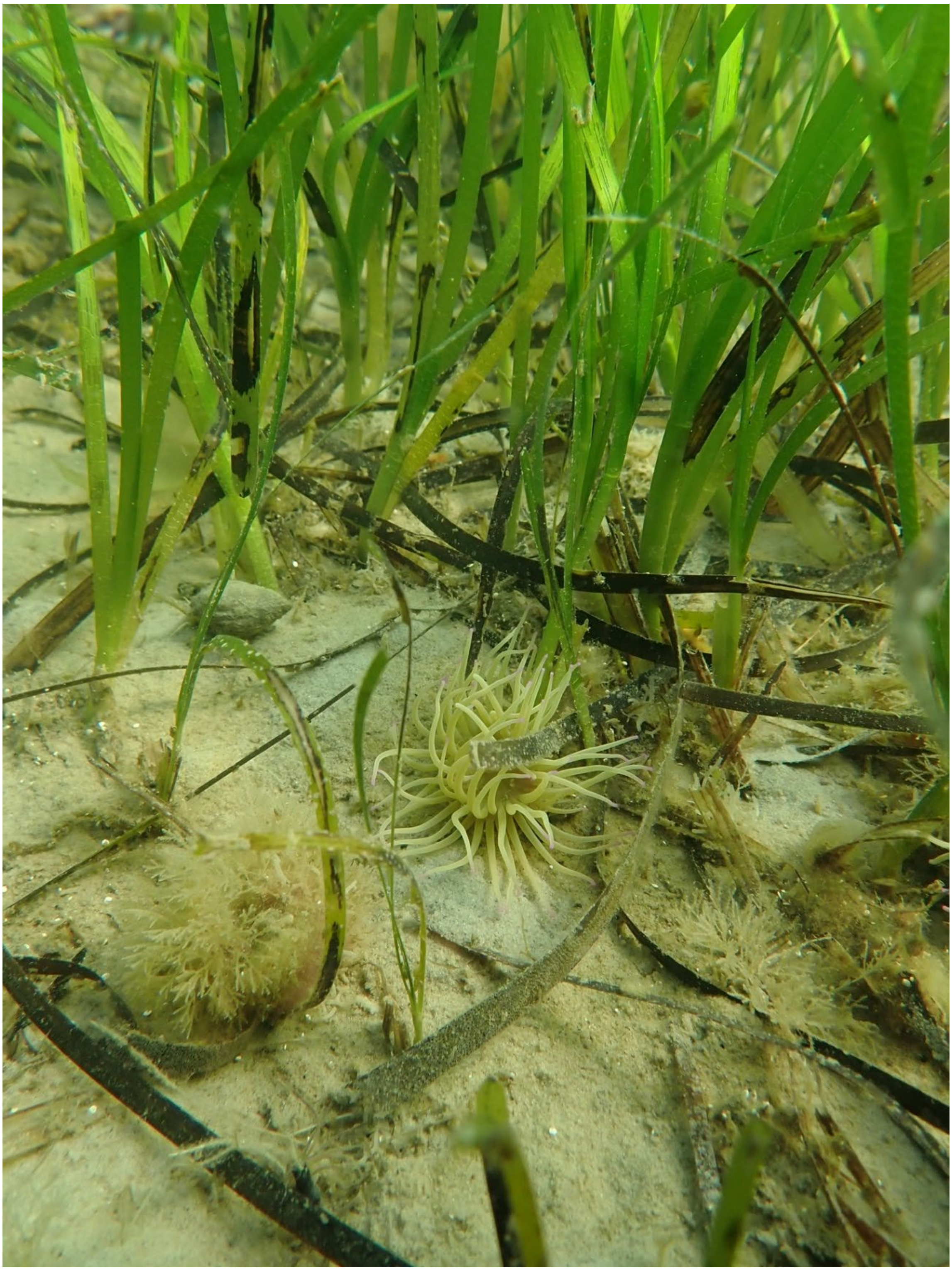

Figure 31: Snakelocks anemone and *Scrupocellaria* sp. amongst the seagrass, at site 3, July 2021.

Figure 32: Amphipod tubes on flowering seagrass at site 3, July 2021.

Figure 33: ***Arenicola*** **mounds at site 3, July 2021.**

Figure 34: ***Arenicola*** **mounds at site 3, July 2021.**

Site 4

- The site is the most northerly of the ‘southern’ sites in the bay with the highest mean density of *Zostera marina* providing 45-100% cover over sand and muddy sand. 2.6-3.2 m below chart datum.
- Long *Zostera* blades (mean = 38.7 cm, max = 84 cm).
- Surveyors described the *Zostera* in this area as ‘luxuriant’.
- Mean algal cover of 29.8% caught/growing in understorey of the *Zostera* plants. Species included *Ulva* spp. and harpoon weed, *Asparagopsis armata* (Falkenbergia phase – ‘pompoms’)
- In patches the gastropods *Rissoa* sp. were superabundant on the seagrass leaves. *Tritia* eggs were also seen.
- Some amphipod tubes were also seen on the ends of the taller *Zostera* blades.
- The deep-snouted pipefish *Syngnathus typhle* was observed in one quadrat.
- A lot of detrital *Zostera* leaves were present amongst the bed.
- Non-native species harpoon weed, *Asparagopsis armata* was observed.
- Anchor damage was apparent within areas of the bed.

Figure 35: Deep-snouted pipefish at site 4, July 2021.

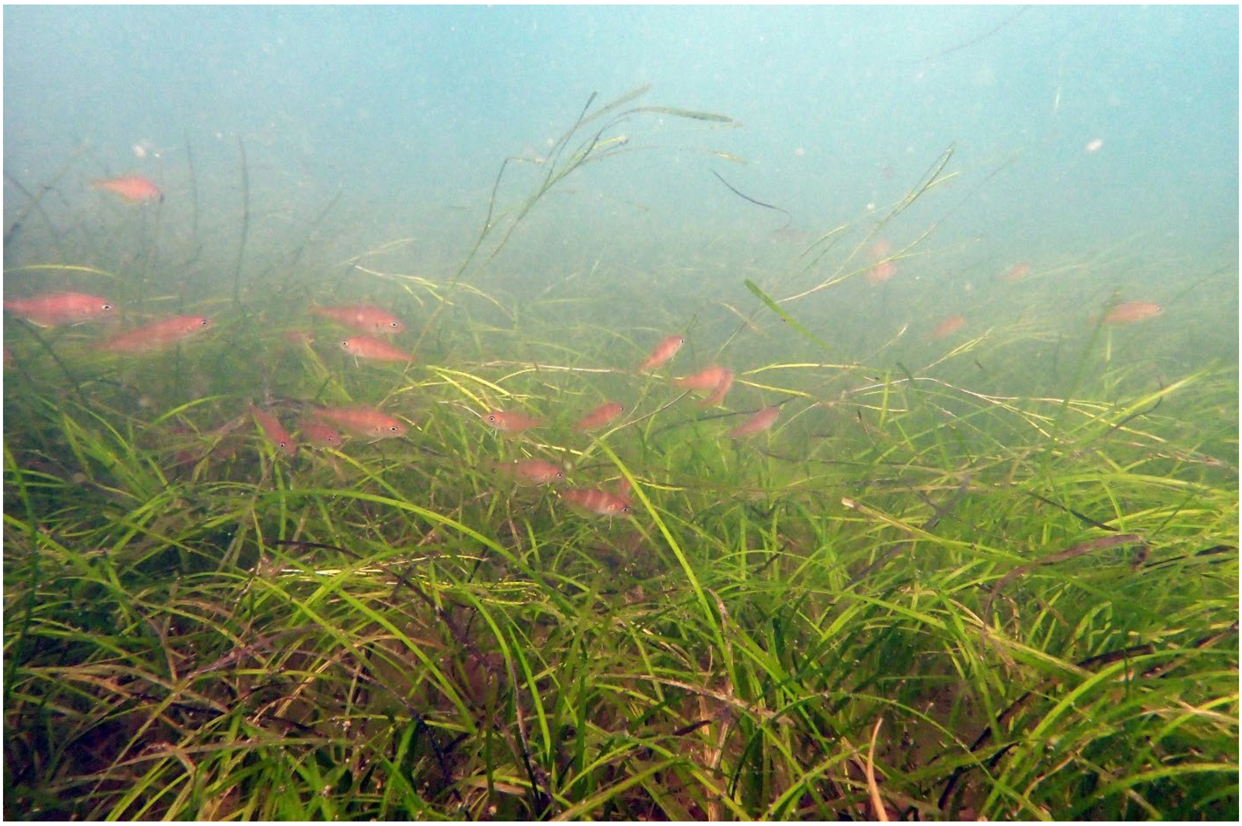

Figure 36: Juvenile bib/pouting among tall, flowering *Zostera* at site 4, July 2021.

Figure 37: Diver collecting samples of *Zostera* at site 4, July 2021.

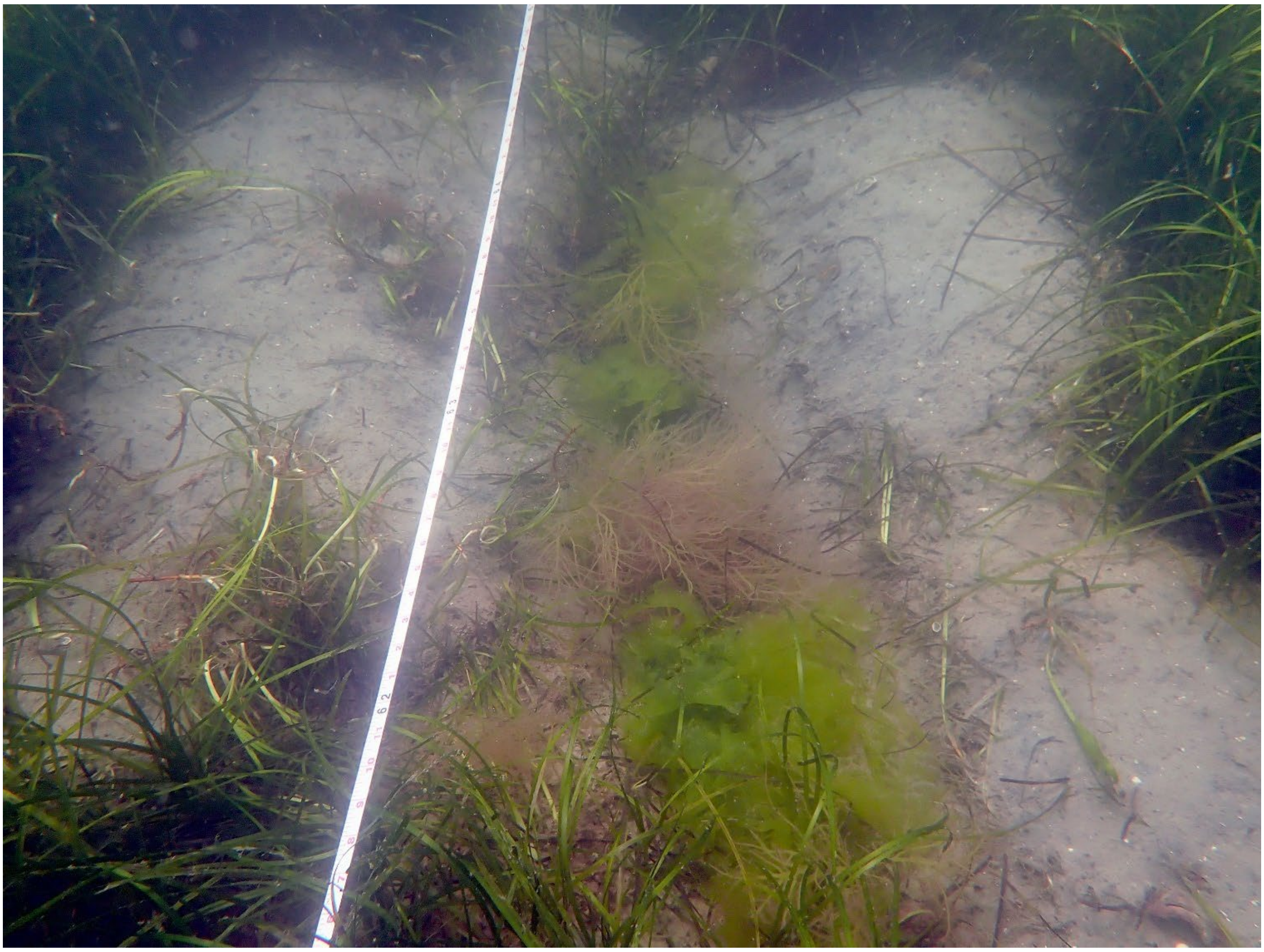

Figure 38: Anchor damage at site 4, July 2021.

Site 5a

- The site is situated northward of the majority of the sites and is opposite Middle Beach within the bay. *Zostera* here was patchy and highly variable in density. Percent cover was 0-80% cover over fine sand, sand and coarse sand. 0.4-1.0m below chart datum.
- The location of site 5a differed from the original site 5 or 6 to be surveyed owing to the water inshore being too shallow to safely survey with the boat traffic on site 5 and a lack of any seagrass at site 6. Site 5a was assigned on the basis that it was between the original two target locations and seagrass was present.
- Shortest mean leaf length of all the sites (mean = 22.6 cm, max = 64 cm).
- Several flowering stems observed (1.9% overall).
- Low algal cover with a mean of 6% cover caught/growing in understorey of the *Zostera* plants.
- Some snakelocks anemones *Anemonia viridis* observed attached to the *Zostera* throughout the bed but fewer than sites 1 and 2. Some amphipod tubes were also seen on the ends of the taller *Zostera* blades. *Scrupocellaria* sp. was notably absent.
- Exposed rhizome mat in places with bare patches – possible historical anchor damage.

- Razor clam, *Ensis ensis* siphons were observed in the sand patches between areas of seagrass.
- The deep-snouted pipefish *Syngnathus typhle* was observed in one quadrat.
- Non-native species observed were the slipper limpet, *Crepidula fornicata and the s*ea squirt, *Botrylloides* sp.

Figure 39: Deep-snouted pipefish at site 5a, July 2021.

Figure 40: Greater pipefish among seagrass, dead seagrass and algae at site 5a, July 2021.

Figure 41: Non-native *Botrylloides* sp. at site 5a, July 2021.

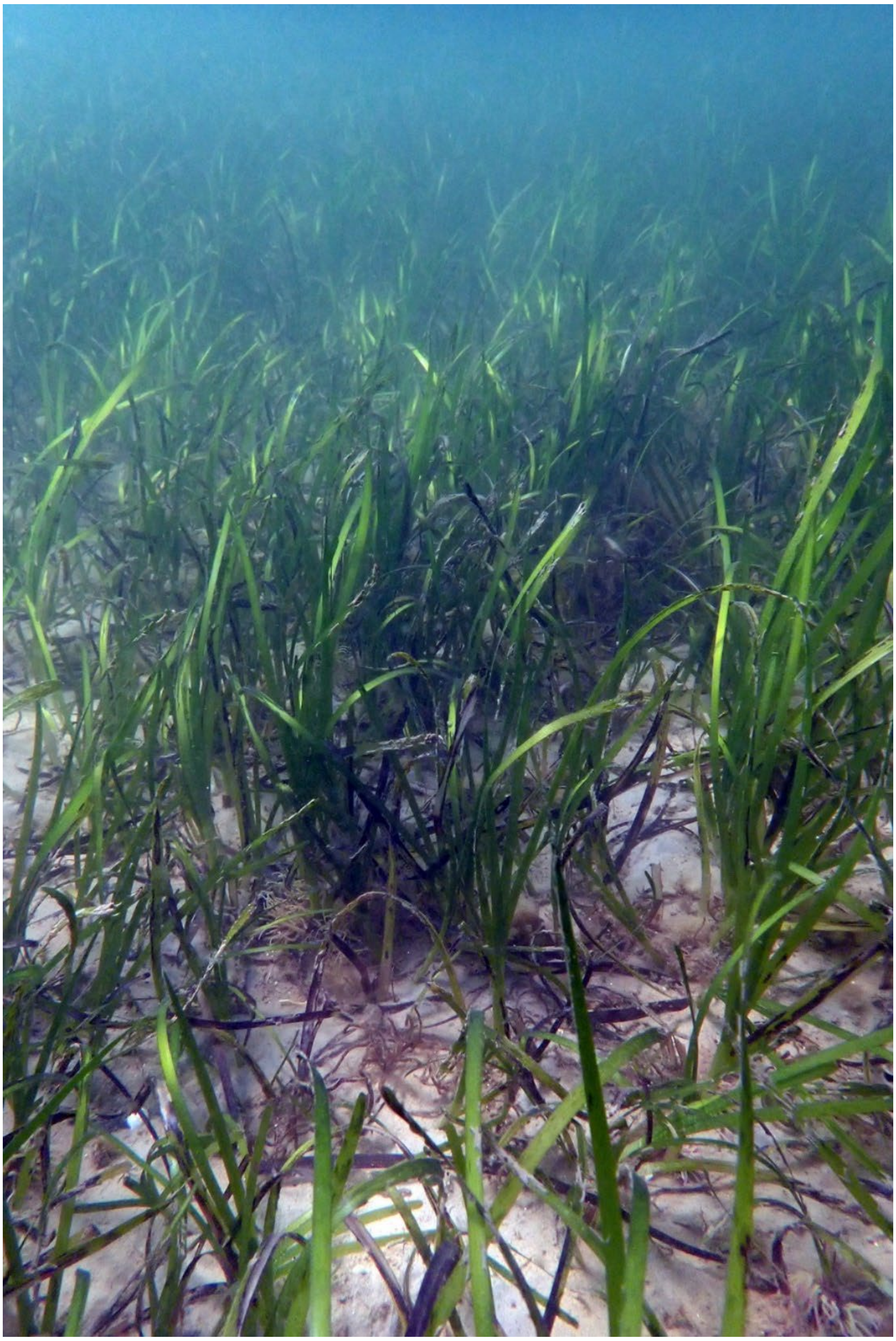

Figure 42: Short seagrass shoots at site 5a, July 2021.

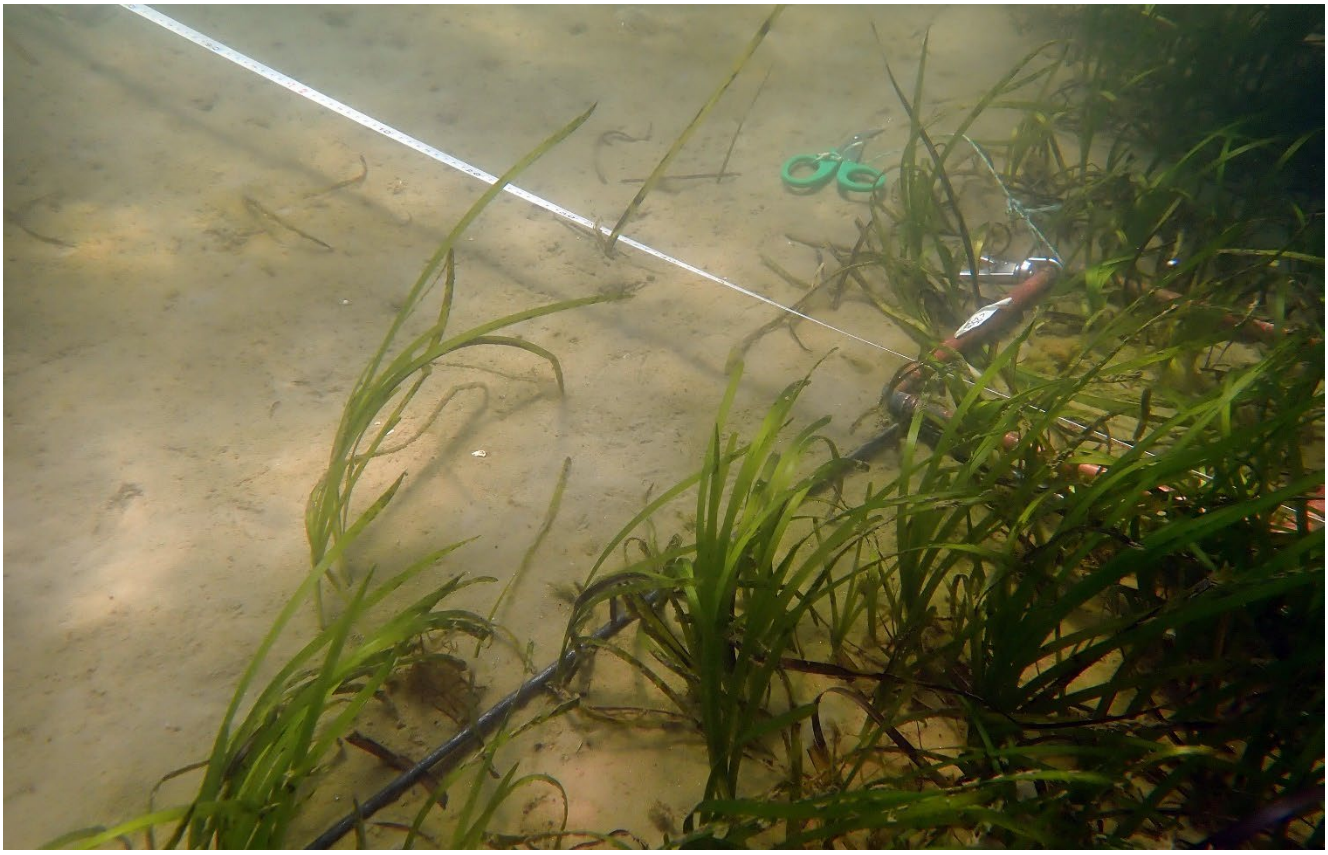

Figure 43: Patch of bare sand at site 5a, July 2021.

Site 7

- The site was the most northerly of all the survey locations in the bay. Comparable density and patchiness to site 5a with *Zostera marina* providing 0-75% cover over sand. 1.7-2.6m below chart datum.
- Short mean *Zostera* blade length but longest max length from flowering plants (mean = 34.2 cm, max = 124 cm).
- Mean algal cover was 24.6% with algae either caught or growing in understorey of the *Zostera* plants. Patches of the red alga *Gracilaria* sp. and the green alga *Ulva* sp. were prominent.
- Some amphipod tubes were also seen on the ends of the taller *Zostera* blades. *Scrupocellaria* sp. was sparse.
- The gastropod *Hinia* sp. was frequently seen.
- Non-native species observed were the sea squirt, *Botrylloides* sp. slipper limpets, *Crepidula fornicata,* leathery sea squirt, *Styela clava,* harpoon weed, *Asparagopsis armata* (Falkenbergia stage) and wireweed, *Sargassum muticum*.
- Interspersing the *Zostera* beds were patches of sand, themselves supporting sporadic *Zostera* plants.


Figure 44: Occasional *Scrupocellaria* sp. and signs of *Labyrynthula* sp. at site 7, July 2021.

Figure 45: Amphipod tubes on seagrass at site 7, July 2021.

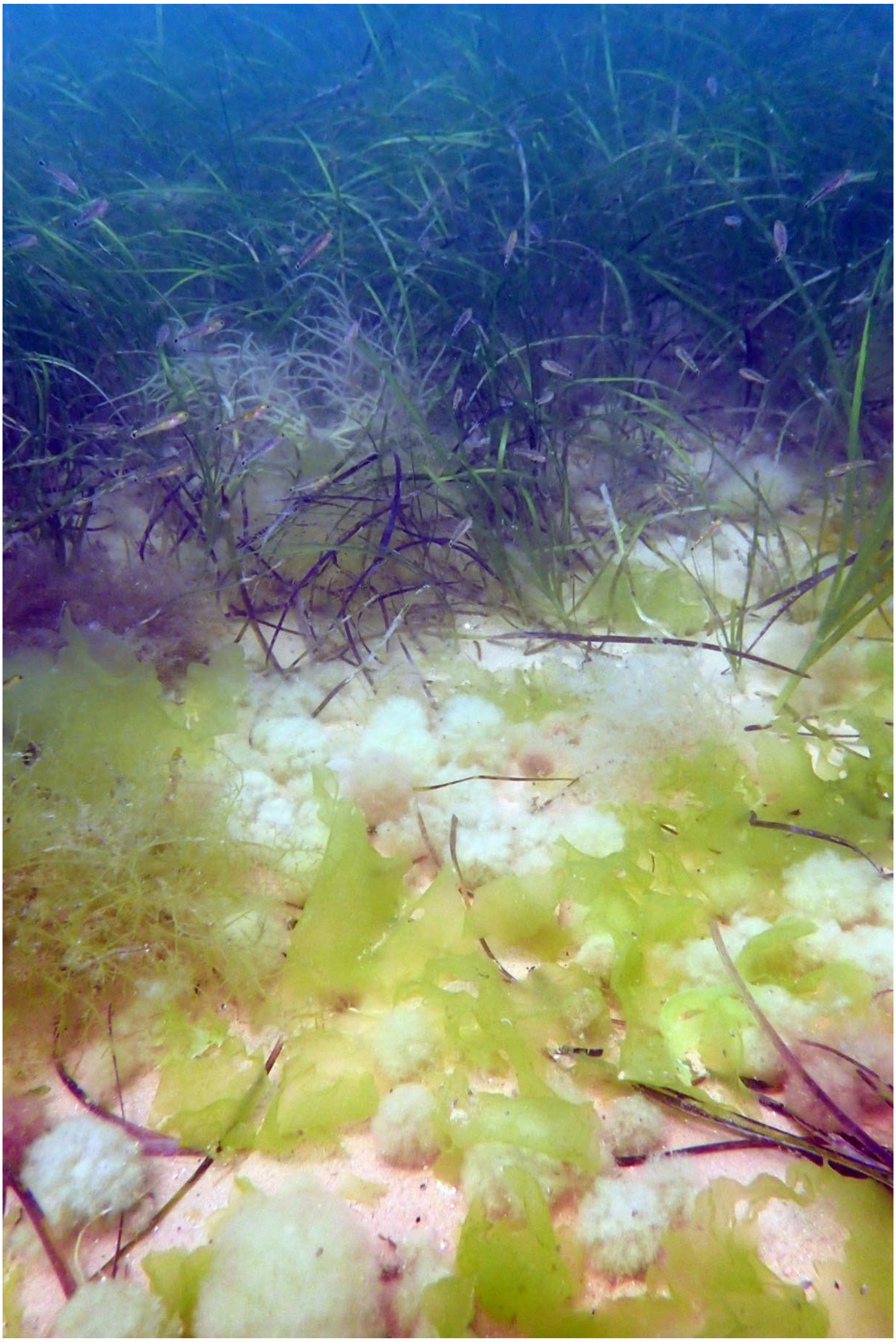

Figure 46: Very patchy site, drift *Ulva* sp., *Gracilaria* sp. and *Asparagopsis armata* at sand/*Zostera* interface at site 7, July 2021.

Site 8

- The site is the most north-easterly of those sampled in the bay and had the highest mean seagrass density of all the sites with *Zostera marina* providing 45-100% cover

over sand and muddy sand (easily stirred up) and appearing very uniform overall. 2.9-3.8m below chart datum.

- Average length *Zostera* leaf blades compared with other sites in the bay (mean = 39.3 cm, max = 80 cm).
- Many flowering *Zostera* plants observed as well as lots of detrital *Zostera* leaves within the bed.
- Reasonable quantity of algae with a mean of 22% cover either caught or growing in understorey of the *Zostera* plants.
- Occasional snakelocks anemones *Anemonia viridis* observed attached to the *Zostera* throughout the bed but fewer than sites 1 and 2.
- In patches the gastropods *Rissoa* sp. were superabundant on the seagrass leaves.
- Non-native species observed were slipper limpet, *Crepidula fornicata* and harpoon weed, *Asparagopsis armata*.
- Many juvenile pollack, *Pollachius pollachius* observed.
- Couch's goby, *Gobius couchi* was observed here. The species is protected under the Wildlife and Countryside Act 1981, Schedule 5 Section 9. Identification confirmed by Lin Baldock.

Figure 47: *Crepidula fornicata* amongst seagrass at site 8, July 2021.

Figure 48: Superabundant *Rissoa* sp. at site 8, July 2021.

Figure 49: *Asparagopsis armata* 'pompoms' amongst living and dead seagrass at site 8, July 2021.

Figure 50: *Gobius couchi* amongst seagrass at site 8, July 2021.

Site 10

- The site is situated more or less in the centre of the most extensive area of seagrass in the south of the bay in an area with a high level of mooring and anchoring activity and active recreational jet skis, paddleboards and kayaks.
- The site had the lowest overall density and lowest percentage cover of any of the sites surveyed with *Zostera marina* providing 0-45% cover over sand and muddy sand. 1.6-2.0m below chart datum.
- Short mean *Zostera* blade length compared with other sites in the bay (mean = 28.5 cm, max = 88 cm).
- Low algal cover with a mean of 9% cover caught/growing in understorey of the *Zostera* plants.
- Areas of *Zostera* beds were interspersed with open sediments areas. Muddy fine sand with lugworm *Arenicola marina* mounds.
- The large green alga *Ulva* sp. was noticeable in the beds, amongst the *Zostera* whilst the bushy bryozoan *Scrupocellaria* sp. (probably *S. reptans*) was also present amongst the *Zostera*.
- Small snakelocks anemones *Anemonia viridis* present throughout attached to *Zostera* blades.
- Some plants had flowering stems present which were extremely silted.
- There were noticeably more dead *Zostera* leaves at the base of plants as well as detrital build-up of leaves in the sediment furrows and areas of bare sediment.
- No recent evident of anchor scars seen.
- No non-native species were observed.

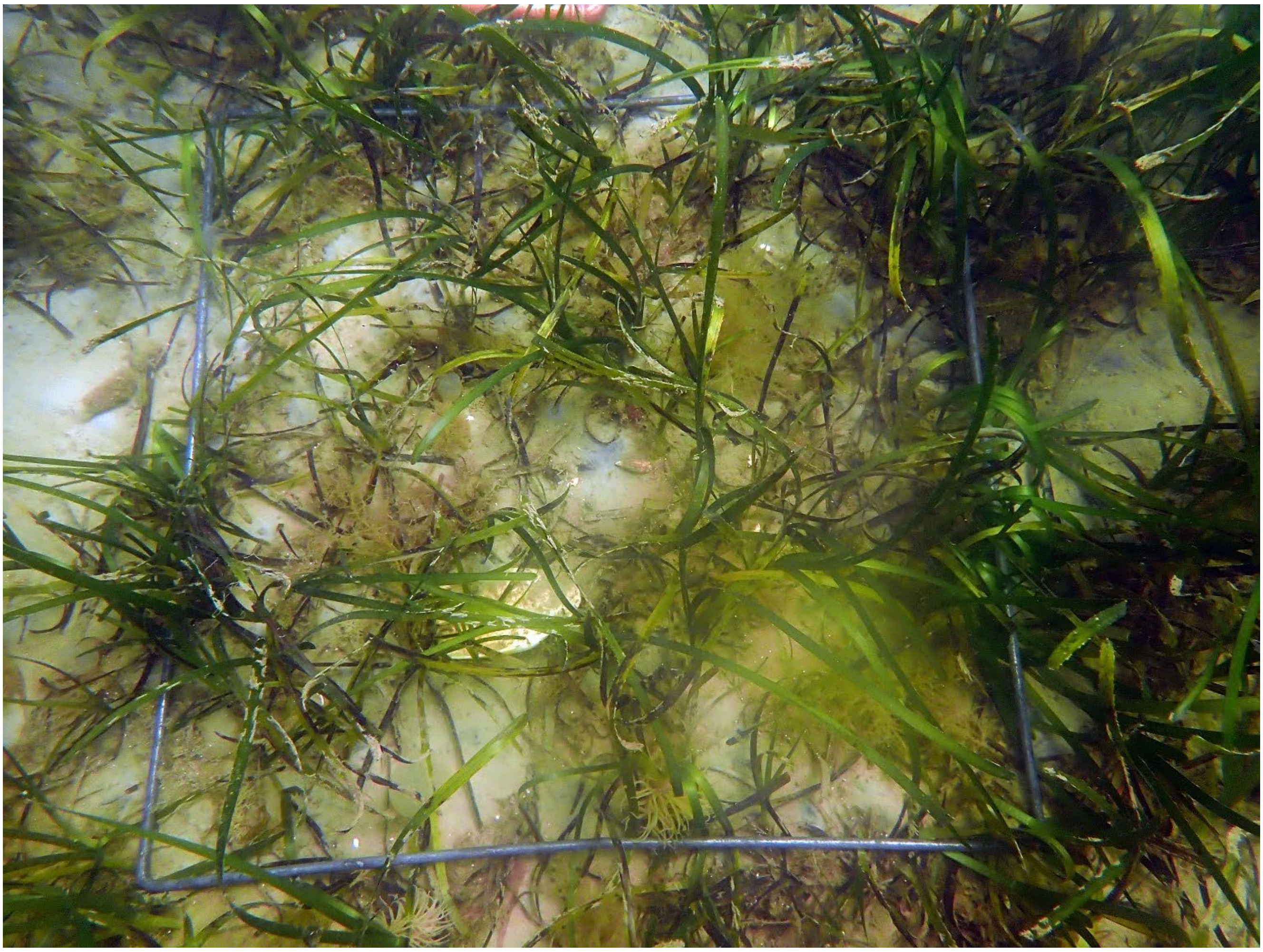

Figure 51: Low density, short seagrass at site 10, July 2021.

Figure 52: Lugworm *Arenicola marina* cast and snakelocks anemones *Anemonia viridis* at site 10, July 2021.

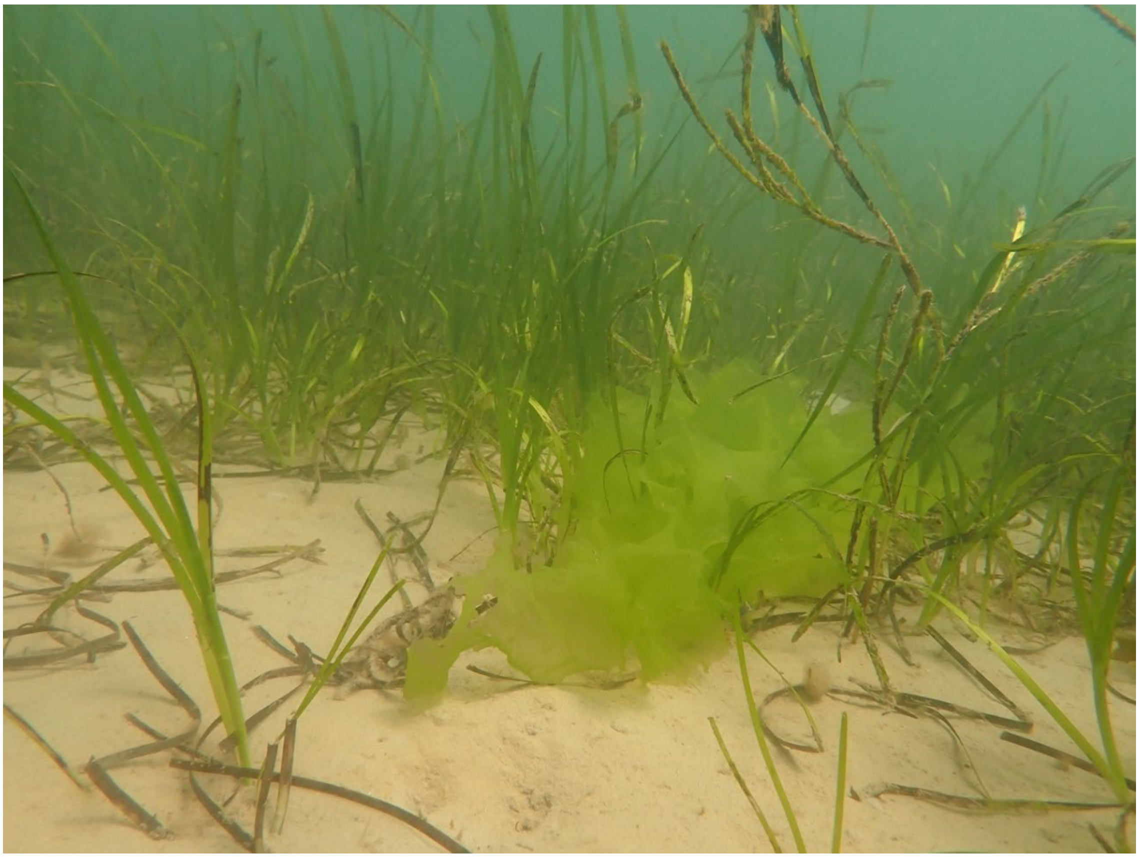

Figure 53: ***Ulva*** **sp. algae in short, sparse seagrass at site 10, July 2021.**

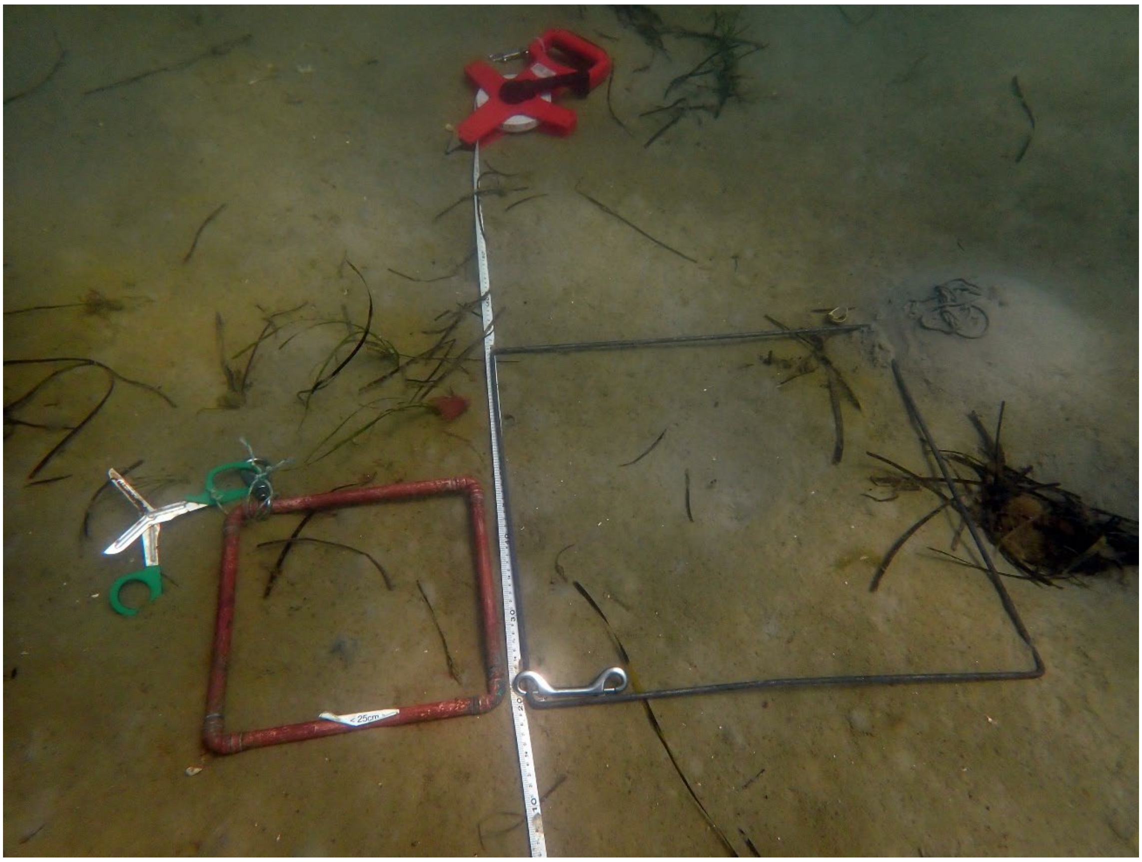

Figure 54: Sampling quadrats on bare sand at site 10, July 2021.

Figure 55: Low density seagrass in a quadrat at site 10, July 2021.

Photography

Stills images and video from the 2021 survey have been supplied to Natural England, organised into folders for each monitoring site with some key species labelled. This resource can be built upon year-on-year.

Appendix B – Project personnel

(All Natural England staff unless otherwise stated)

Project led by Natural England personnel

Survey contract managers
Lucy May and Maxine Chavner

Survey methodology
Lucy May and Ian Saunders

Field survey leaders
Lucy May and Ian Saunders

Survey team 2021
Gavin Black (supervisor)
Lucy May
Ian Saunders
Jenny Murray
Gina Wright
Caroline Waddell
Matt Doggett (Marine EcoSol)
Kate Northen (Marine EcoSol)
Nick Owen (Marine EcoSol)

Data entry and sample processing team
Nick Owen & Lin Baldock (Marine EcoSol)

Skipper of Mary Jo
Andy and Brian

Data analysis and reporting
Matt Doggett and Kate Northen

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