Seagrass and Maerl Natural Capital Literature Review

Accessible Version

May 2022

Natural England Commissioned Report NECR416

www.gov.uk/natural-england

About Natural England

Natural England is here to secure a healthy natural environment for people to enjoy, where wildlife is protected and England's traditional landscapes are safeguarded for future generations.

Further Information

This report can be downloaded from the Natural England Access to Evidence Catalogue. For information on Natural England publications contact the Natural England Enquiry Service on 0300 060 3900 or email enquiries@naturalengland.org.uk.

Copyright

This publication is published by Natural England under the Open Government Licence v3.0 for public sector information. You are encouraged to use, and reuse, information subject to certain conditions.

Natural England photographs are only available for non-commercial purposes. If any other photographs or information such as maps or data cannot be used commercially this will be made clear within the report.

For information regarding the use of maps or data see our guidance on How to access Natural England's maps and data.

© Natural England 2022

ISBN 978-1-78354-908-5

Catalogue code: NECR416

Report details

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.

Author(s)

Emily Howard-Williams

Contractor

Westcountry River Ltd.

Natural England Project Manager

Joanne Bayes, Senior Specialist, Natural England

joanne.bayes@naturalengland.org.uk

Keywords

Marine, natural capital, seagrass, maerl

Acknowledgements

The authors would like to thank all who contributed to the creation of this report, including, Fiona Crouch, Maija Marsh, Zeenat Qadir, Carolyn Waddell, Hazel Selley, Jessica Taylor, Angela Gall, Kate Sugar, Gina Wright, Professor Jason Hall-Spencer, Dr Ken Collins, Dr Matt Ashley, Dr Sian Rees, Tom Hooper, Phil Horton, Cat Palmer, Victoria Spooner, Ian Tolchard, Mark Parry, Martin Bidmead, Gary Cairns, the River Yealm Harbour Authority and Kate Ansell.

LIFE Recreation ReMEDIES (LIFE18 NAT/UK/000039) is financially supported by LIFE, a financial instrument of the European Commission. ReMEDIES is led by Natural England in partnership with The Royal Yachting Association, Marine Conservation Society, Ocean Conservation Trust and Plymouth City Council/Tamar Estuaries Consultative Forum.

Working in partnership with:

Citation

HOWARD-WILLIAMS, E. 2022. Seagrass and Maerl Natural Capital Literature Review NECR416. Natural England.

Executive summary

England's varied marine environment, its ecosystems, geodiversity and seascapes, provides people with a wide range of benefits, upon which human wellbeing depends. These benefits include thriving wildlife, cultural and spiritual enrichment, food, clean water and air and reduced risks from environmental hazards, such as flooding. Seagrass and maerl are unique ecosystems which provide a suite of benefits from carbon sequestration, enhancing water quality, to the provision of nursery habitat for commercial fish species.

This literature review and the supporting place-based mapping reports use Natural England's natural capital indicators to review and map the state of the seagrass and maerl and the ecosystem services they provide within five Special Areas of Conservation. Habitat suitability data illustrates the potential area of seagrass distribution if pressures were to be removed/reduced. Data from previous seagrass studies illustrates the potential for increased ecosystem services within these areas.

By applying a natural capital approach to better understand the links between healthy seagrass and maerl habitats and the ecosystem services they provide, we hope to increase public awareness of the importance of these habitats and the wider environmental, societal and economic benefits they provide.

Contents

List of figures

List of tables

1 Seagrass and maerl natural capital assessment

Natural England are the lead partner of the LIFE (financial instrument of the European Commission) funded project: LIFE Recreation ReMEDIES: Reducing and Mitigating Erosion and Disturbance Impacts affEcting the Seabed (LIFE 18 NAT/UK/000039). This project is running from July 2019 until October 2023 and will improve the condition of five Special Areas of Conservation (SACs) between Essex and the Isles of Scilly. This will be achieved by habitat restoration and reducing recreational pressures. Promoting awareness, communications and inspiring better care of sensitive seabed habitats will be key. Alongside Natural England the project partners are the Marine Conservation Society, Ocean Conservation Trust, Plymouth City Council/TECF and the Royal Yachting Association.

An element of this project is to improve the public knowledge of these habitats by applying the natural capital approach to describing the ecosystem services and wider benefits of healthy seagrass and maerl beds. This report provides an overview of current literature relating to the natural capital provided by seagrass and maerl, particularly in relation to the five SACs covered by the ReMEDIES project, and will be used to inform a natural capital mapping exercise for the sites.

Natural England uses the Natural Capital Committee's (2017) pp.9 definition of natural capital:

"the elements of nature that directly or indirectly produce value to people, including ecosystems, species, freshwater, land, minerals, the air and oceans, as well as natural processes and functions".

1.1 Seagrass

Two species of seagrass are found in England, *Zostera marina* (*Z. marina*) and *Zostera noltii* (*Z. noltii*). A third *Zostera angustiflolia* was thought to be a separate species, but is now considered a sub-species of *Z. marina* (Guiry and Guiry, 2020). *Ruppia maritima* is included under the 'Seagrass' category of Features of Conservation Interest (marine features that are particularly threatened, rare, or declining species and habitats) (Marine Life Information Network, 2022) but, although it is often found with seagrasses, it is not a true seagrass (Tyler-Walters and d'Avack, 2015). This report will focus on *Z. marina* and *Z. noltii.*

Seagrasses are marine flowering plants found in sheltered subtidal and intertidal zones at flow velocities up to 1.5m/s (Borum *et al.*, 2004), down to depths of 10m dependent on water clarity and species (Jackson *et al.*, 2013). Seagrasses have variable growth rates, and dispersal and range expansion can occur sexually through seed dispersal or through the spread of rhizomes. In *Z. marina* and *Z. noltii* the dispersal of rhizomes can only occur over a gentle topological gradient, therefore, disturbances that create deep scarring in surrounding sediment can result in restricted rhizomic expansion (Jackson *et al.*, 2013; D'Avack *et al.*, 2014).

Fragmented and patchy seagrass beds, with percentage cover below 60% are more vulnerable to losses during storms than more dense, uniform beds, which is likely to be related to dense patches having self-protective properties which make them more stable (Borum *et al.*, 2004). Globally, seagrasses occupy less than 0.2% of the sea bed (Fourqurean *et al.*, 2012), but they are estimated to sequester 10% of the yearly ocean organic carbon (Duarte *et al.*, 2005) and have similar soil carbon storage potential as temperate forests (Fourqurean *et al.*, 2012). They provide physical structure on a somewhat structureless sediment which enhances biodiversity as well as primary and secondary production (Duffy, 2006), provide vital habitat for protected species such as seahorses, particularly the long-snouted seahorse *Hippocampus guttulatus* (Garrick-Maidment *et al.*, 2010; Jackson *et al.*, 2013), and provide vital nursery habitats for commercial fish species (Unsworth *et al.*, 2018). In the United Kingdom (UK) this includes species such as pollack *Pollachius pollachius*, sole spp., mullet spp., plaice *Pleuronectes platessa*, skates spp., rays spp., (Ashley *et al.*, 2020). The European Nature Information System (EUNIS) classifications for seagrass habitats are detailed in Table 1.

Table 1 – Seagrass EUNIS habitat classifications and codes.

EUNIS Name	Habitat code
Seagrass beds on littoral sediments	A2.61
Zostera noltii beds in littoral muddy sand	A2.6111
Sublittoral seagrass beds	A5.53
Zostera beds in full salinity infralittoral sediments	A5.533
Zostera marina/*angustifolia* beds on lower shore or infralittoral clean or muddy sand	A5.5331
Angiosperm communities in reduced salinity	A5.54
Zostera beds in reduced salinity infralittoral sediments	A5.545

1.2 Maerl

Maerl has been defined by Hall-Spencer *et al.* (2008) pp.3 as:

"'Maerl' is a collective term for various species of non-jointed coralline red algae (Corallinaceae) that live unattached. These species can form extensive beds, mostly in coarse clean sediments of gravels and clean sands or muddy mixed sediments, which occur either on the open coast, in tide-swept channels or in sheltered areas of marine inlets with weak current. As maerl requires light to photosynthesize, the depth of live beds is determined by water turbidity, being found from the lower shore to 40 m or more. Maerl beds may be composed of living maerl, dead maerl or varying proportions of both".

Maerl is extremely slow growing (0.1 – 1.0 mm y^{-1}) (Bosence and Wilson, 2003 in Newton, 2011) and forms thick beds of dead skeletal matter over centuries and even millennia. Maerl can support disproportionately high diversity and abundance of associated species compared to surrounding habitats (Hall-Spencer *et al.*, 2010). Some species are only found in the maerl or are rarely seen elsewhere (Hall-Spencer *et al.*, 2008), making them highly valuable and productive habitats, particularly as a nursery ground for commercially important species such as queen scallops *Aequipecten opercularis* (Kamenos *et al.*, 2004a), cod *Gadus morhua* and edible crabs *Cancer pagurus* (Hall-Spencer *et al.*, 2008).

When disturbed through the use of towed fishing gear, anchoring or mooring, recovery is very slow as re-growth can take 10-25 years and as maerl beds have formed over millennia, the habitat may never fully recover after a disturbance event (Hall-Spencer *et al.*, 2008).

The European Nature Information System (EUNIS) classifications for maerl are detailed in Table 2.

Table 2 – Maerl EUNIS habitat classifications and codes

Habitat type	Habitat code
Maerl beds	A5.51
Phymatolithon calcareum maerl beds in infralittoral clean gravel or coarse sand	A5.511
Phymatolithon calcareum maerl beds with red seaweeds in shallow infralittoral clean gravel or coarse sand	A5.5111
Phymatolithon calcareum maerl beds with *Neopentadactyla mixta* and other echinoderms in deeper infralittoral clean gravel or coarse sand	A5.5112
Lithothamnion corallioides maerl beds on infralittoral muddy gravel	A5.513

1.2.1 Importance of live maerl beds

Pristine live maerl beds provide nursery areas for commercial populations of queen scallops *Aequipecten opercularis* and other invertebrates, such as the soft clam *Mya arenaria*, the sea urchins *Psammechinus miliaris* and *Echinus esculentus*, and the starfish *Asterias rubens*, more effectively than impacted dead maerl and other common substrata (Kamenos *et al.*, 2004a).

Laboratory mesocosm studies have shown that for queen scallops (*Aequipecten opercularis*) they selected pristine live maerl as a preferred substrate compared to dead maerl, gravel or sand, and in the presence of predators (starfish or crabs) selected more complex ‘matrix’ live maerl and sheltered between and under maerl nodules (Kamenos *et al*., 2004b). Further study showed increased survival in complex live maerl compared to other habitats showing that impacts that alter live maerl to lead to dead maerl substrate may reduce the refuge potential and alter growth–predation relationships for commercial scallops (Kamenos *et al*., 2004c).

Further research has shown that this habitat preference is also found for other molluscs and a trigger for settlement from plankton, linked to a likely chemical signature from live calcareous algae, rather than the presence of a biofilm, demonstrating the value of live maerl for attracting and aiding populations of bivalve molluscs (Roberts *et al*., 2010).

1.3 Study areas and scope of the review

Natural England commissioned a report to assess the natural capital benefits of seagrass beds found within five SACs across southern England (Figure 1) (Fal and Helford, the Isles of Scilly Complex, Plymouth Sound and Estuaries, Solent Maritime and Essex Estuaries) and maerl beds located within the Fal and Helford SAC.

Comprehensive literature and data reviews were undertaken in relation to the seagrass and maerl beds located within the specified SACs, including quality of the habitat, provision of ecosystem service benefits, and the potential increased ecosystem service benefits upon removal of pressures relating to recreational activities (ie, mooring, anchoring and trampling). As part of the ReMEDIES project updated seagrass and recreational surveys were planned in summer 2020 to inform the study, however due to restrictions related to the Covid-19 pandemic these surveys were postponed. Survey data from 2021 for the Solent seagrass beds have been included, but otherwise this study is informed by desk-based assessments of pre-existing surveys and data only.

Figure 1– Locations of target Special Areas of Conservation (SACs) across southern England. Map shows (from east to west) the location of the Essex Estuaries, Solent Maritime, Plymouth Sound and Estuaries, Fal and Helford and Isles of Scilly Complex SACs.

2 Methods

2.1 Literature review method

The Quick Scoping Review (QSR) method outlined by DEFRA and the Joint Water Evidence Group was utilised to guide the literature review process (Collins *et al.*, 2015). A detailed description can be found in Appendix 1. The limited time frame for delivery of this work package meant that the search terms selected were highly specific to each site and only literature which was clearly relevant to each site were read in full.

2.2 Seagrass and maerl natural capital indicators

Natural England show the links between ecosystem assets, services, benefits and value to people through the use of logic chains (Figure 2). These show how the state of an asset, its quantity and location affect services and benefits provided (Wigley *et al.*, 2020). During the literature review and when speaking with experts, indicators of quality, location and measures of the ecosystem services provisioned by seagrass and maerl were collated in order to review the natural capital benefits provided.

To describe the extent and location of the seagrass and maerl beds, Natural England have provided the most up-to-date spatial data, which is not presented in this report but will be used directly in the subsequent natural capital mapping exercise. In order to assess quality, a range of indicators were investigated through the literature review, including both direct indicators such as plant measurements from surveys, and indirect data that may be indicative of quality such as water quality. Direct indicators are outlined for each SAC in section 3.1 and indirect indicators for all SACs are discussed in section 3.2. Indicators of ecosystem services and the service flows are outlined in section 4 and are based on the list of marine natural capital indicators and the associated ecosystem services produced by Natural England (Lusardi *et al.*, 2018). Lastly the potential for increased ecosystem service benefits based on the findings of the literature review are discussed in section 5.

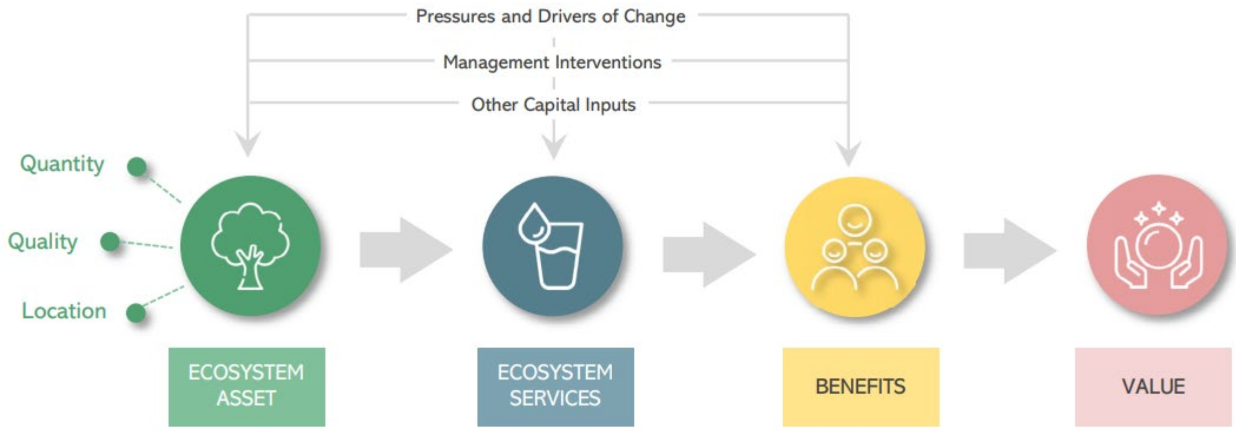

Figure 2 – Generalised natural capital logic chain taken from Wigley *et al.* (2020). The logic chain shows that how much, how good and where natural assets are, affect the ecosystem services, benefits and value people get from them. It shows that management interventions, as well as pressures and drivers of change, influence this chain. Other capital inputs are also often needed for people to obtain the benefits from ecosystem services (a simple example is the processing of trees to produce wood products).

3 Indicators of seagrass and maerl quality

3.1 Seagrass direct quality indicators

The direct quality indicators of seagrass were taken from SAC condition assessments, or equivalent where these were not available (the Isles of Scilly Complex and Essex Estuaries) (Table 3). Alternative published reports and peer reviewed literature were reviewed and relevant data extracted; particularly measurement of plant descriptions as these provided an indicator of health (Wood and Lavery, 2001; Ruiz and Romero, 2003) (ie, shoot density, leaf length, % cover, presence of wasting disease). Bull and Kenyon (2019) noted that measurements should not be extrapolated to allow comparison with other reports as this implies knowledge of spatial heterogeneity across spatial scales, therefore, plant measurements are reported in units in which they were recorded in the respective reports. Epiphyte cover was not included as this was measured differently between studies or was not included at all. Some reports include flowering incidents, however Jackson *et al.* (2016) suggest that where a survey is carried out over a short time frame, the use of flowering as an indicator of quality is not appropriate, therefore this has not been included in the quality assessments below.

Ratios of leaf nutrient concentrations provide indicators of the abiotic conditions which influence seagrass quality:

- Declining Carbon:Nitrogen (C:N) ratio provides an early indicator of restricted light availability (McMahon *et al.*, 2013) (high light=≥20, reduced light=14-20, low light=≤14 (Jones and Unsworth, 2016))
- Carbon:Phosphorus (C:P) ratio indicates environmental P availability, <400 indicates over-enrichment of P which can impact seagrass quality (Mckenzie *et al.*, 2012; Jones and Unsworth, 2016)
- Nitrogen:Phosphorus (N:P) ratio provides an indicator of the balance of environmental N and P (Mckenzie *et al.*, 2012) (between 0-20 considered to be balanced (Jones and Unsworth, 2016)).

The quality indicators of maerl are not as clearly defined, Allen *et al.* (2014) suggested a method for assessing favourable condition based on survey results, but did not provide a final assessment on condition as there was no clear method for measuring change. The indicators presented here include Natural England assessment of condition, expert opinion, extent and number of species recorded.

3.1.1 SAC condition assessments (seagrass as a sub-feature)

Condition assessments for SACs are undertaken every six years, these reports exist for the Fal and Helford, Plymouth Sound and Solent Maritime SACs. The overall condition of the seagrass as a complex sub feature (subtidal and intertidal) have been extracted and included in Table 3.

Table 3 – SAC condition indicators for each location. **()Reference relating to reported metric.**

SAC Name	SAC condition assessment	Species	Nutrient status	Light availability	Mean shoot density - per/m^2 unless otherwise specified (Range)	Mean Leaf length - cm (range)	Wasting disease - %	Mean cover - % (range)	References
Isles of Scilly Complex	Declining[(1)]	*Z. marina*	Good	Good	9.3-15 per 0.0625m^2 [(1)] [approx. equivalent to 148-240 per m^2] 4±1.4 per 0.25m^2 [(2)]	78.8±4.9[(2)]	No data	91.3±2.5[(2)]	[(1)]Bull and Kenyon, (2016) [(2)]Jones and Unsworth, (2016)
Plymouth Sound and Estuaries	Unfavourable Unknown[(3)]	*Z. marina*	No data	No data	64-119 (0-240)[(4)]	52-80 (07-144)[(4)]	29-53 [(4)]	11-69 [(4)]	[(3)]Gall, (2018) [(4)] Bunker and Green, (2018)
Fal and Helford	Unfavourable[(5)]	*Z. marina*	Anecdotal evidence of nutrient loading	No data	14-85 (0-256)[(6)]	33-43 (8-62)[(6)]	12-20[(6)]	18-69 (0-88)[(6)]	[(5)]JNCC, (2011) [(6)]Curtis, (2015)
Essex Estuaries	Unfavourable[(7)]	*Z. marina*	Evidence of nutrient loading	Evidence of restriction (leaf length)	63 (8-230)	18 (13-28)	No data	48 (10-100)	[(7)]Jackson *et al.* (2016)

SAC Name	SAC condition assessment	Species	Nutrient status	Light availability	Mean shoot density - per/m^2 unless otherwise specified (Range)	Mean Leaf length - cm (range)	Wasting disease - %	Mean cover - % (range)	References
Essex Estuaries	Unfavourable[7]	*Z. noltii*	Evidence of nutrient loading		54 and 88 (1-210)	8 and 15 (2-25)	No data	38 and 42 (5-100)	[7]Jackson *et al.* (2016)
Solent Maritime	Unfavourable Unknown[8]	*Zostera* spp.	Evidence of nutrient loading[8]	No data	max.1150[9] 209 (0-600)[10] 45.38 ± 30.79[11]	34.3 ±15.8 stdev (5-100)[10] 52.17 ± 16.0[11]	Infection scores for presence of *Labyrinthula zosterae* were low (<1.2 which is = <25%) for all the sites and ranged from 0 – 3.4 (0-100%)[10]	48.5 (0-100)[10] 45.66 ± 28.72[11]	[8]Natural England, (2018c) [9]Marsden and Scott, (2015) [10]Doggett and Northern (2022) [This study excluded beds at Totland and Ryde] [11]Furness and Unsworth (2022)

3.1.2 The Isles of Scilly Complex

Using leaf nutrient ratios, the seagrasses of the British Isles have been described to be in a perilous state (Jones and Unsworth, 2016). The study sites for this research included four beds in the Isles of Scilly Complex SAC. Compared to the other seagrass beds surveyed for this study, the seagrasses in the Isles of Scilly had the highest observed shoot biomass and seagrass cover (91.3±2.5%), lowest shoot density (4±1.4 per 0.25m^2), longest (78.8±4.9cm) and widest leaves (10.7±0.5mm). The %N values at all of the study sites across the British Isles were above the global average of 2.04 (Jones and Unsworth, 2016). The seagrasses in the Isles of Scilly Complex were at the lower end of recorded N values with below 3%N, yet nutrient ratio N:P was found to be highly imbalanced with a ratio of over 40 with a P% under the global average and high C:P ratios (over 500) (Jones and Unsworth, 2016). This study concluded that plants in the Isles of Scilly Complex were growing in a limited P pool, with high light availability (>20 C:N) and high N%, but were in good ecological state compared with seagrasses in other locations despite being over-enriched with N. This was attributed to the distance from large human populations. This finding is in contrast to the assessment made by Project Seagrass, who described the seagrasses of the Isles of Scilly as declining based on long-term extent data (Bull and Kenyon, 2019). Bull and Kenyon (2019) reported shoot density of 9.3-15 per 0.0625m^2 (this is roughly equivalent to shoot density of 148-240 per m^2, however as discussed earlier in the report there are difficulties with extrapolating the data across spatial scales). The prevalence of wasting disease was variable throughout the SAC; in some locations a declining trend was observed, while the opposite was recorded in others. The results presented in Bull and Kenyon's (2019) study shows that the seagrass beds in the SAC are declining, however, the results of Jones and Unsworth's (2016) study suggest that this decline is not driven by restricted light availability or the nutrient status, therefore it is likely that another factor is causing this decline.

3.1.3 Plymouth Sound and Estuaries

The most recent study of the seagrass beds in the Plymouth Sound and Estuaries SAC, carried out in 2018, indicates that five of the six beds are declining compared to the results of a study undertaken in 2012 (Curtis, 2012). An increase of 56% in seagrass cover was observed at the Cawsand bed, however the confidence in this is low due to poor sea conditions and equipment failures experienced in Curtis' (2012) survey.

The average shoot density and leaf length (cm)[1] per bed across the Plymouth Sound and Estuaries SAC is 64-119 per m^2 and 52-80cm respectively (Bunker and Green, 2018). Wasting disease was present at all sites with bed averages of between 29-53% of leaves

[1] Described as mm in report, however, this should be cm.

showing signs of the disease. Cover by epiphytes was not considered to be detrimental to the seagrasses.

3.1.4 Fal and Helford

The Fal and Helford SAC is the only one of the ReMEDIES SACs which is looking at both seagrass and maerl. The direct quality indicators for each habitat are outlined separately in the sections below.

3.1.4.1 Seagrass

The subtidal seagrass beds in the Fal and Helford SAC are in unfavourable condition; there is a small area of intertidal seagrass which is considered to be in favourable condition, but the confidence in this assessment is low (Natural England, 2019a). The most recent survey of the seagrass in the Fal and Helford SAC indicate that two small areas of seagrass were lost in the lower Percuil and in the Fal north of Trefusis Point (Curtis, 2015). In the remaining 14 seagrass beds apparent extension from previous surveys were recorded, however Curtis (2015) notes that the confidence is low due to the possibility that the full extent of the beds were not mapped entirely in past surveys. Using drop-down/towed video surveys (DD/TV) percentage cover was highest in the Flushing bed (69%) and lowest at Gyllngvase (18%), the range for the whole SAC was between 0-88%. The average shoot density recorded for individual seagrass beds was between 14-85 (0-256) per m^2, with the highest shoot density recorded in the Amsterdam Point and Carricknath Point bed and in the lowest at Penarrow Point to Trefusis Point. Wasting disease was observed infecting between 12-20% of leaves sampled.

While no direct data was located which referred to seagrass and the nutrient status or light availability, personal communication with Carolyn Waddell (Natural England Lead Marine Adviser) indicated that nutrient enrichment is an issue in the SAC.

3.1.4.2 Maerl

Few data are available that describe the quality/condition of maerl in the Fal and Helford SAC. They are considered to be in unfavourable or unfavourable declining condition across the SAC (Natural England, 2019a). Significant declines were reported in the quality of the maerl within the Fal and Helford SAC due to extraction. This is a particular problem as extraction not only removes live maerl, it also deposits sediment on plants that avoided extraction, inhibiting their recovery (Hall-Spencer, 2005 in Hall-Spencer *et al.*, 2008). Extraction of the maerl is no longer allowed, although there are no data to assess the recovery after this intervention (Hall-Spencer *et al.*, 2008).

In 2014 Natural England published a report on the maerl beds in the Fal and Helford SAC (Allen *et al.*, 2014). This report specifically focused on the extent, distribution and species composition of the maerl beds in the SAC and concludes that there was no evidence to indicate that there had been a change in the extent of the maerl beds between 2002 and 2013 (Allen *et al.*, 2014). Live maerl beds are located at St. Mawes, Castle Point and in the Helford Estuary, however the bed in the Helford Estuary was not surveyed due to access issues (Allen *et al.*, 2014). South of Pendennis Point and north of the Fal harbour, small amounts of live maerl were found among the dead maerl gravel. Two species can be found in the SAC, *Phymatolithon calcareum* and *Lithothamnion corallioides* (Allen *et al.*, 2014). During a telephone conversation with Professor Hall-Spencer (01/09/2020) he described the maerl bed at St Mawes as the best example in England.

Seasearch surveys in 2012 (Gall, 2012) recorded the density of live maerl within the Helford estuary and suggested the bed is composed primarily of the maerl species *Lithothamnion corallioides* based on visual identification. It is in shallow waters and approximately 4m below chart datum. The percentage of live maerl within the bed was high, averaging about 80% across the bed. It supports a high diversity of other seaweed species including the Priority species *Cruoria cruoriaeformis* (a red seaweed) (F.Bunker, pers comm.), *Ostrea edulis* (native oyster) and *Edwardsia timida* (timid burrowing anemone).

Seasearch surveys in 2021 (Selley, 2021) within the Carrick Roads area of the Fal estuary recorded maerl density in two locations as 75-90% maerl coverage, with live maerl having 50% cover at one location and large nodules 20-30cm in size, and 80% live maerl west of St. Mawes. At both locations two distinct nodule forms were present suggesting both *Lithothamnion corallioides* and *Phymatolithon calcareum* were present in live form; the Priority species *Cruoria cruoriaeformis* was also present (H. Selley, pers comm).

Genetic uniqueness in the Fal

Phymatolithon calcareum maerl beds across the north-east Atlantic have low dispersal potential and limited connectivity between regions. Jenkins *et al.* (2021) found that *P. calcareum* from the Fal Estuary is genetically distinct from all other *P. calcareum* sampled in the north-east Atlantic, even from The Manacles Marine Conservation Zone located only 13 km away. Analysis revealed that this is not the result of hybridisation of the closely related species, *Phymatolithon purpureum* or *Lithothamnion corallioides*, but was likely shaped over time by the geographical isolation of the Fal Estuary maerl bed and a lack of gene flow with other *P. calcareum* populations.

3.1.5 Essex Estuaries

Jackson *et al.* (2016) surveyed seagrass beds within the Essex Estuaries SAC. Of the seven sites surveyed, seagrass was only found in two (Shoeburyness and St Lawrence) and they were found to be patchier than in previous surveys undertaken in 2012 (Jackson *et al.,* 2016). Two species of seagrass were recorded during the surveys, *Z. marina* (Shoeburyness) and *Z. noltii* (Shoeburyness and St Lawrence). Jackson *et al.* (2016) concluded that all the seagrasses in the Essex Estuaries SAC are in unfavourable condition, with a decline in all the previous sites. Four of the sites, which have historically supported seagrass, were not surveyed as they are within the MOD danger area, therefore the quality and condition of the seagrass within these areas is unknown. The danger area provides protection from trampling, mooring, and anchoring as this area is strictly “No Access”.

The mean shoot density for *Z. noltii* was 54 (Shoeburyness) and 88 (St Lawrence) per m^2, and 63 per m^2 for *Z. marina* (Jackson *et al.,* 2016). Leaf length and % cover was higher for *Z. marina* (18cm and 48% respectively) compared to *Z. noltii* (8-15cm and 40% respectively). The observed leaf lengths of *Z. marina* were lower than reported by other studies and may suggest constricted growth due to reduced light availability (Jackson *et al.*, 2016). Jones and Unsworth (2016) found there were indicators of light limitation at seagrass beds at Southend-on-Sea (11km from Shoeburyness) which would support this theory.

3.1.6 Solent Maritime

The Hampshire and Isle of Wight Wildlife Trust's report on the seagrass in the Solent Maritime SAC provides excellent details on the extent of seagrass beds within the SAC and the wider area. However data on plant measurements which would indicate quality could not be easily extracted to provide summaries for this SAC. The following maximum plant densities per m^2 were extracted from the data: Langstone Harbour 350, Cowes 1150 and Osbourn Bay 750.

Surveys of the Solent seagrass beds took place in 2021, providing information on the condition of the beds including shoot density, leaf length and cover. These figures have been included in Table 3.

Dr Ken Collins has confirmed in a telephone conversation (28/08/2020) that there are few published data on the seagrasses in the Solent Maritime SAC, but his expert opinion is that the beds are generally in good condition. Natural England's SAC condition assessment of the extent of subtidal seagrass beds in 2018 provides some evidence to support this, showing that the seagrass extent has not declined over the last 10 years (to 2018), however there was only medium confidence in the assessment (Natural England, 2018a; Natural England, 2018b) and overall the subtidal seagrass beds were considered to be in unfavourable condition due to water pollutants and disturbance. Similar results exist for the intertidal seagrass beds, where either no changes or increases were recorded for a number of beds in the short-term, however analysis of long-term data shows a decline in some areas (Natural England, 2019), and the intertidal seagrass is also considered to be in unfavourable condition. Watson *et al.* (2020) mentioned that despite the area being extensively studied, the data on seagrass within the Solent Maritime SAC are limited, which subsequently resulted in low confidence scores when mapping the natural capital stocks. The Southern Inshore Fisheries Conservation Association (SIFCA) have byelaws which prevent commercial harvesting, hand gathering and bait digging, however this is difficult to police and these activities still occur (personal communication with Jessica Taylor (Natural England Marine Lead Adviser) 21/08/2020).

3.2 Seagrass and maerl indirect quality indicators for all SACs

3.2.1 Water quality and clarity

Declining water quality and clarity are the main threats to the health of seagrass habitats, with nutrient loading and increased turbidity through activities such as eutrophication, aquaculture, coastal development, dredging and spoil disposal being of particular concern for seagrass and maerl as they can negatively affect health and productivity (Jones *et al.*, 2000; Ruiz and Romero, 2003). van Katwijk *et al.* (2016) found that in areas where seagrass restoration was attempted, 54% of loses prior to restoration were attributed to water quality deterioration.

Nutrient loading indirectly affects seagrass and maerl by reducing light reaching the habitat. The increased availability of nutrients causes a shift in the dominant vegetation to faster growing species, eg opportunistic macroalgae and epiphytes, ultimately reducing the light availability (Jones and Unsworth, 2016). Jones *et al.* (2000) noted that increased turbidity, detritus from fish farming, and algal blooms from excessive nutrients and dredging all decrease the penetration of light through the water column and inhibits photosynthesis. Reduction in light levels lowers the ability of seagrass and maerl to photosynthesize, in turn affecting growth and reproduction. A two-week reduction in light penetration is tolerable for *Z. noltii* (Peralta *et al.*, 2002), if this period is extended then the plant experiences losses in biomass, leaf density and growth rate, these factors would impact recovery rates after periods of damage (Philippart, 1995).

Turbidity can also reduce the oxygen availability for seagrass respiration and may result in hypoxic conditions (Mateo *et al.*, 2006). Decreased water quality linked to sewage discharge, and shellfish and finfish farm waste are likely to be particularly damaging to maerl due to the increased oxygen demand required to break down the waste (Wilson *et al.*, 2004).

It is proposed that modelled light attenuation and nutrient data (N and P) (Butenschön *et al.*, 2016) are utilised as indicators of seagrass and maerl quality within the target areas. As sewerage can reduce oxygen availability for maerl species (Wilson *et al.*, 2004), it would also be recommended that data on sewerage entering the Fal and Helford SAC is used as an indirect indicator of maerl quality.

3.2.2 Rare, protected or indicator species

The presence of indicator species within the seagrass beds could provide a further indication of quality. For example the seahorse species *H. guttulatus* and brent geese *Branta bernicla* have particularly strong associations with seagrass. *H. guttulatus* utilise complex vegetative habitats, such as seagrass for feeding, breeding and protection; 43% of *H. guttulatus* records are in *Zostera* spp. (Garrick-Maidment *et al.,* 2010). *B. bernicla* prefer to feed on seagrass due to the high nutritional value (Ganter, 2000). Distribution data for these species (or other relevant species) within the target SACs could be mapped and would give a further indirect indicator of seagrass quality.

3.2.3 Recreational boating

Due to their close proximity to the shore and intertidal coastal zones, seagrass beds are easily accessible to humans and exposed to both terrestrial and marine based threats (Cullen-Unsworth *et al.*, 2013). Boating can cause various types of disturbance to seagrass beds, including through propeller damage and the impacts of mooring and anchoring (D'Avack *et al.*, 2014). Mooring and anchoring will be addressed here, as per the ReMEDIES project aims. Reducing the impact of these activities are priorities for the Solent Maritime, Isles of Scilly Complex, Plymouth Sounds and Estuaries and Fal and Helford SACs.

All the target SACs were ranked in the top 26 of 173 sites exposed to mooring and anchoring as shown in Table 4 (Griffiths *et al.*, 2017). Although Griffiths *et al.* (2017) ranked the Essex Estuaries as 7th most impacted by mooring and anchoring, this is a result of the number of moorings rather than anchoring pressure within the SAC (IEG, 2020).

Table 4 – Ranking for each SAC according to moorings and anchoring pressure. Table adapted from Griffiths *et al.* (2017).

SAC	Rank (out of 178)
Solent Maritime	4
Essex Estuaries	7
Plymouth Sound and Estuaries	9
Fal and Helford	18
Isles of Scilly Complex	26

The average size of an individual mooring and anchoring scar is 122m^2 (Unsworth *et al.*, 2017) and 1-4m^2 (Collins *et al.,* 2010), respectively. These data suggest that the impact of an individual anchoring event is less than a swing mooring. In Australia, organic carbon (C_{org}) was found to be five times lower in mooring scars (1.6 Kg C_{org} m^{-2}) than in the surrounding undisturbed seagrass bed (6.4 Kg C_{org} m^{-2}) (Serrano *et al.*, 2016), which demonstrates the potential of each mooring to reduce the carbon sequestration rates of seagrass habitats. The impact an anchor has on seagrass can depend on the type and size of anchor. The number of shoots uprooted during the complete anchoring process ranged from 1.8 (Hall and Donforth) to 5.5 (Folding Grapnel) (Milazzo *et al.*, 2004).

Moorings are generally a permanent feature with chronic impact (Griffiths *et al.*, 2017), making the impact on seagrass and maerl easier to quantify. Anchoring on the other hand, can occur any number of times in seagrass or maerl beds and is highly variable spatially and temporally. Anchoring also tends to be free and unregulated. This variability makes the impact of anchoring difficult to measure and quantify and is therefore more of an unknown threat. Maerl is the most sensitive habitat to anchoring and mooring due to its slow growth and recovery rates (Hall-Spencer *et al.*, 2008; Griffiths *et al.*, 2017). Although no specific data have been located on the impact of individual anchoring events on maerl, a 25m scour was observed in maerl beds within the Fal and Helford SAC (Ashley, 2017).

Data on the extent and amount of boating activity in the SACs could provide an indirect indicator of habitat quality; higher boating activity could result in greater exposure to mooring and anchoring, resulting in lower quality, the data and evidence of boating pressures are summarised in Table 5. The Essex Estuaries is not a priority for reducing the impact of boating, the main pressure here is trampling which is discussed in the following section. The Royal Yachting Association (RYA) have agreed to share their data on recreational boat use for use within the natural capital mapping exercise.

Table 5 – Summary of the data and evidence of the impact of recreational boating. AMS=Advanced Mooring System.

SAC name	Evidence of boating impact	Number of moorings over seagrass beds	Average scar radius (m)	No. advanced mooring systems	No anchor zone	Other interventions	References
Isles of Scilly Complex	Mooring – scars St Marys[(1)]	142 St Mary's[(1)]	6.75[(1)]	No data	No data	No data	[(1)]Unsworth *et al*. (2017)
Isles of Scilly Complex	Anchoring – Porthcressa[(2)]	No data	No data	No data	No data	No data	[(2)]Island's Partnership (n.d.)
Isles of Scilly Complex	Potential anchoring and mooring – Higher Town Bay and Tean [(3)]	No data	No data	No data	No data	No data	[(3)]Kate Sugar (Natural England Marine Lead Adviser)
Plymouth Sound and Estuaries	Reduction of *Z. marina* cover and average canopy height in a 4 m to 6 m radius around mooring blocks [(4)]	1 Yealm[(5)] 13 Cawsands (5 AMS; 8 Swing mooring)[(4)]	No data	5[(6)]	No data	No data	[(4)]Bunker and Green (2018) [(5)]Yealm Assistant Harbour Master [(6)]Mark Perry (Ocean Conservation Trust) [(7)]Langmead *et al.* (2017)
Plymouth Sound and Estuaries	Mooring, anchoring and slipways (Cawsands, Yealm and Drakes Island) [(7)]	As above	No data	As above	No data	No data	As above

SAC name	Evidence of boating impact	Number of moorings over seagrass beds	Average scar radius (m)	No. advanced mooring systems	No anchor zone	Other interventions	References
Fal and Helford	Mooring – scars St Mawes and Helford[1]	Approx. 18 St Mawes[9]	3.6[1]	No data	Helford[8]	Mooring and anchoring discouraged around fisheries and water sports areas [9,10]	[1]Unsworth *et al.* (2017) [8]Ian Tolchard (Helford Mooring Officer) [9]Gary Carnes (St Mawes Harbour Authority) [10]Carolyn Waddell (Natural England Marine Lead Adviser) [11] Falmouth Harbour Commissioners [12] Curtis, 2015
Fal and Helford		36 Helford[1]	4.7[1]	No data	As above	As above	As above
Fal and Helford		14 Fal[11]	No data	No data	As above	As above	As above
Fal and Helford		3 Parbean Cove[12]	No data	No data	As above	As above	As above
Solent Maritime	Anchoring Osbourn Bay[13]	No data	No data	No data	No data	No data	[13]Jessica Taylor (Natural England Marine Lead Adviser)

3.2.4 Trampling

There are limited data on the impacts of trampling on seagrass globally, and as far as can be determined, there are no UK based studies. Trampling of intertidal seagrass beds is an issue in the Solent Maritime and Essex Estuaries SACs. Eckrich and Holmquist (2000) investigated the effects of trampling of seagrass in Puerto Rico, they observed an overall decrease in biomass with increased trampling pressure and time. Specifically a reduction in seagrass *Thalassia testudinum* rhizome biomass, leaf area index, short-shoot density, canopy height and standing crop. In Willapa bay, Washington, trampling has a greater impact in soft substrata (Major *et al.*, 2004) and shallower water, as there is less buoyancy (Eckrich and Holmquist, 2000). Garmendia *et al.* (2017) found that in Spain, under heavy trampling conditions *Z. noltii* shoot density reduce by 23%, whereas under light trampling there was no difference from the control (no trampling pressure).

Travaille, Salinas-de-León and Bell (2015) observed a significant reduction in blade length (mm) and shoot count (per 100cm^2) between highly impacted areas (trampled) compared to the control. These results correspond with Eckrich and Holmquist (2000), where percent cover reduced by an estimated 22%[2] when compared to the control group. An increase in bare sand was also observed and would suggest there was a corresponding reduction in the associated ecosystem services. Under the no trampling scenario the seagrass cover increased by an estimated 5%[2] over 4 months. The impacted seagrass was found to only moderately recover from trampling after seven months if the pressure has ceased entirely. Estimated average increases included leaf area index ↑ 0.5[2] (m^2/m^2), short shoot density ↑ 125[2] (m^{-2}) and canopy height ↑ 6[2] mm (Eckrich and Holmquist 2000). Suggesting that even low levels of trampling have potentially long-lasting impacts on seagrass beds (Travaille, Salinas-de-León and Bell, 2015). Data on trampling within the target SACs were not identified, although Jackson *et al.* (2016) observed trampling by walkers, horse riders, dogs, kite surfers and boats launching in the Essex Estuaries SAC. As a result of limited data specifically related to the impact of reducing trampling, estimations of the potential for the provision of ecosystem services were not attempted.

4 Ecosystem service flow indicators

In order to assess the natural capital of seagrass and maerl beds within the target SACs, a series of ecosystems service flow indicators have been identified based on a combination of the ecosystem services, service flows, and benefits provided by Natural England and the findings of the literature review. Natural England has produced a list of marine natural capital indicators and the associated ecosystem services (Lusardi *et al.*, 2018). Table 6

---

[2] based on estimates taken from graphical representation of the results not the actual figures.

breaks these down by habitat type (seagrass and maerl) and the resulting ecosystem service flows and benefits are outlined in Table 7.

Table 6 – Natural capital indicators and ecosystem services (Lusardi *et al.*, 2018).

Natural capital indicators	Ecosystem service: Wild animals, plants, algae and outputs	Ecosystem service: Water quality	Ecosystem service: Maintenance of nursey populations and habitats	Ecosystem service: Climate regulation
Seagrass	✓	✓	✓	✓
Maerl	✓		✓	

Table 7 – Natural England's breakdown of ecosystem services, service flows and benefits which relate to seagrass and maerl.

Ecosystem service	Ecosystem service flow	Benefits
Wild animals, plants, algae & outputs	Fish, shellfish, seaweed and other products (tonnes)	Products from the sea eg fish, shellfish & seaweed for food, fertiliser, angling bait, medicines. Quality of fish & shellfish (age/length profile; % affected)
Water quality	Water quality (chemical & biological, including viral & bacterial	Clean water, also underpinning eg sustainable ecosystems, cultural services, health benefits
Maintenance of nursery populations & habitats	Maintenance of sustainable ecosystems/life cycle stages	Biodiversity, in of itself, and underpinning all other services such as recreation (including wildlife watching), tourism, research and education, food from wild populations & aquaculture
Climate regulation	Carbon sequestered (tonnes CO_2, per m^2 or m^3) and greenhouse gases fixed	Equitable climate eg reduced risk of drought, flood & extreme weather events, lower summer temperatures, reduced health & safety risks, reduced flood risk, protection of infrastructure/lack of transport disruption

4.1 Maintenance of nursery populations & habitats ecosystem service indicators

A measure (presence/abundance) of species that utilise the seagrass beds within the SACs for spawning and during juvenile life stages would prove a useful indicator for this service. Biodiversity metrics could also be used as an indicator of this ecosystem service. Jackson *et al.* (2001) noted that the nursery value of a specific seagrass bed is dependent on a number of processes including food availability, which in turn has a fundamental influence on survival rates. Therefore, it is proposed that biodiversity is used as an indicator of food availability within the seagrass beds. Higher diversity in terms of richness and evenness within the seagrass should logically translate into increased availability of food resources for a wider range of functional taxonomic groups.

A. opercularis are a commercially important shellfish and while dredging is not permitted over the maerl beds directly, they do act as nursery grounds for juveniles and sources for permitted areas (Kamenos *et al.*, 2004a). *A. opercularis* can be attracted by chemical stimulants which are released by the live maerl, and juveniles were found to attach in higher numbers to pristine live maerl compared to impacted dead maerl, gravel or sand (Kamenos *et al.*, 2004a). Therefore *A. opercularis* numbers over different age classes within the maerl beds would provide a good indicator for the provision of this ecosystem service.

The ecosystem services identified through Natural England's Indicators report (Lusardi *et al.*, 2018) is limited to the key services for each indicator. In the case of seagrass this does not include mass stabilisation or flood protection, however the ecosystem service flows and benefits presented in Table 7 suggest that it would be appropriate to consider during this literature review. The presence of seagrass beds can provide a degree of coastal protection through the attenuation of wave transmission onshore (Duarte *et al.,* 2013). The degree at which wave attenuation occurs depends on leaf length and the density of seagrass (Fonseca and Cabalan, 1992; Chen *et al.*, 2007; Hansen and Reidenbach, 2012). The effectiveness can vary spatially and temporally, for example Fonseca and Cabalan, (1992) found that when seagrass leaf length was equal to the water depth 40% reduction in wave energy per m^2 was achieved. Similarly, Chen *et al.* (2007) found that peak wave attenuation was observed during the flowering season when shoot density was over $1000m^{-2}$ and reduced with shoot density and canopy height. Gambi (1990) tested the flow dynamics in different shoot densities (400-1200 per m^2) they found that current was reduced within the canopy at all densities. Hansen and Reidenbach (2012) observed mean currents 2-3 times lower in seagrass compared to bare sand. The lowest shoot density recorded in the aforementioned paper was 150±80 shoots per m^2, which also provided the lowest within-canopy flow reduction. When compared to the shoot densities recorded at the target SACs, densities equivalent to this were recorded at all of the SACs, however, the average shoot densities were all lower (Table 3). There is a possibility that seagrass could provide wave attenuation during extreme weather events, particularly in seagrass beds which are in leaf throughout the year. When the condition of the seagrass beds within the target areas for this project are considered, it would suggest that coastal protection provision of seagrass may be highly variable between site, season and dependant on the condition.

4.1.1 Suggested indicator datasets for “Maintenance of nursery populations & habitats” ecosystem service by seagrass and maerl

Nursery and spawning ground data are available to download from CEFAS, and would provide details on the presence of species utilising seagrass and maerl beds.

It would be challenging to standardise biodiversity metrics over all of the target SACs, a species count from a national survey would be the best option for representing the diversity of the habitats. APEM (2019) used sediment biota data to assess the natural capital value of selected marine protected areas in England, these data are available from the Marine Biological Association of the UK and could be obtained for use here.

Kamenos *et al.* (2004a) estimated that in the west coast of Scotland 18.2 juvenile (<45mm shell height) queen scallops per 100m^2 were attached to pristine live maerl (which may increase to 38.75 per 100m^2 in December), while impacted dead maerl attracted 0.21 per 100m^2. These figures could be combined with the assessment of the maerl beds undertaken by Allen *et al.* (2014) to estimate the local provision of this ecosystem service.

As mentioned above, the provision of coastal protection will be dependent on seagrass shoot density and leaf lengths. The figures obtained for these measurements in section 3.1 and the estimation of a 2-3 times reduction in the mean current could be used as an indicator of this service. However this would be dependent on locating current data across the SACs and would take further investigation.

4.2 Wild animals, plants, algae & outputs – Fish, shellfish, seaweed and other products (tonnes)

Ashley *et al.* (2020) reported that on the Isles of Scilly, the species that are most reliant on seagrass are pollack, sole, mullet, plaice, skates and rays, all of which would be negatively affected if seagrass were degraded. Seagrass is reported to provide valuable nursery habitat for 21.5% of the top 25 landed species (Unsworth *et al.*, 2018). Jackson *et al.* (2015) found that seagrass associated species contributed an estimated 30-40% of the value of commercial fisheries landings in the Mediterranean, which included shellfish species. In some countries seagrasses are harvested for use in thatching, packing material and even for consumption, however in England there are no direct uses (Jackson *et al.*, 2013). Fishing intensity data were collected for the Isles of Scilly Complex and contributed to the likely relative condition (Ashley *et al.*, 2020), therefore the same measure could be used to assess the provision of this service by seagrass and maerl.

4.2.1 Suggested indicator datasets for "Wild animals, plants, algae & outputs" ecosystem service by seagrass

Data which link directly to seagrass outputs are unlikely to exist, however, there are data on the landed value of fish including shellfish per port (Monthly Sea Fisheries Statistics Dec 2019). Species associated with seagrass can be identified. It should be noted that the associations presented in the previous section are not all based on data collected in the UK and include a variety of seagrass species, therefore the confidence in the estimations in the UK context is low.

4.3 Water quality – chemical, biological including bacteria and viruses (seagrass)

There are a number of measures of water quality which could be utilised to indicate the provision of this service within the SACs. As discussed previously the nutrient content and clarity of the water both have an impact on water quality. Seagrasses can improve the quality of water by removing detrimental anthropogenic inputs, through nutrient uptake and by depositing suspended particles within the water column (Short and Short, 1984). An estimation of the N and P burial rates and sedimentation rates over the area of the seagrass would provide an indicator of the seagrasses provision of this ecosystem service.

The sediment accumulation rates (SAR) of seagrass have not been studied long-term (Röhr *et al.*, 2016). Many of the estimates are linked to carbon sequestration rates (eg, Duarte *et al.*, 2013; Serrano *et al.*, 2014 and Miyajima *et al.*, 2015). Short and Short (1984) experimentally tested the sediment removal potential of two seagrass species *Halodule wrightii* and *Syringodium filiforme* in Florida. They found that both species were more effective at depositing suspended particles than the control, but estimates of SAR are not made in this study. Miyajima *et al.* (2015) observed SAR of between 0.37 and 1.34 mm yr^{-1} for *Z. marina*. Therefore, the SAR could be used as an indicator of the provision of this service.

The bacterial filtration ability of seagrass was assessed by Lamb *et al.* (2017) on the midshelf of the Spermonde Archipelago, Indonesia. They observed a 50% reduction in the relative abundance of harmful bacteria when seagrass beds were present compared to when they were not. Data on the presence and abundance of bacteria within the SACs could provide an indication of seagrasses' contribution to the localised water quality. No data could be located on seagrasses' potential to filter viruses.

Seagrasses can act as a detoxifier for Tributyltin (TBT) (Francois and Weber, 1988 in Jackson *et al.*, 2013), they can absorb heavy metals and where the burial of dead seagrass tissue is high, they could act as heavy metal sinks (Jackson *et al.*, 2013) and store heavy metals within the tissues and sediment. Records of TBT and heavy metals within the sites could offer an indication of the seagrass detoxification potential.

4.3.1 Suggested indicator datasets for "Water Quality" ecosystem service by seagrass

Short and Short's (1984) estimated the N (NH_4) uptake potential of seagrasses, however, to use this estimation for the SACs, details of N concentrations and seagrass biomass would be needed. Furthermore, this estimation is over 30 years old and more up to date information is available. Watson *et al.* (2020) provided a comprehensive summary of N and P burial rates as well as estimation of denitrification taken from a number of existing papers. These figures present the most up to date estimations and could be used to estimate this service across the SACs.

Table 8 – Seagrass nitrogen, phosphorus and carbon removal rates – table adapted from Watson *et al.* (2020). Sources of figures varies. N = Nitrogen (g N m^{-2} yr^{-1}); P = Phosphorous (g P m^{-2} yr^{-1}); C = Carbon (g C m^{-2} yr^{-1})

Ecosystem process	N mean	N med	N min	N max	P mean	P med	P min	P max	C mean	C med	C min	C max
Burial	4.9	3.9	3.7	8	-2.2*	-4.3	-12.8	12.5	83	110	19	191
Denitrification	15.1	14.3	14.1	16.1	No data	No data	No data	No data	No data	No data	No data	No data

*Note: There are limited studies available to provide accurate figures for P change, the average figure shown in this table is based on one study which actually found a seasonal net release of P from a particular seagrass bed. Future studies would be useful to confirm whether this is a common scenario for other seagrass beds.

Röhr *et al.* (2016) summarises the available minimum (Miyajima *et al.*, 2015), average (Duarte *et al.*, 2013) and maximum (Serrano *et al.*, 2014) SAR from three papers (0.32, 2.02 and 4.20 mm yr^{-1}, respectively). However, these studies do not easily allow estimates of SAR per areal unit, therefore, it is suggested Gacia and Duarte's (2001) estimate of 2 mm m^{-2} y^{-1} is used to extrapolate sedimentation rates as a proxy for the provision of this service as this includes area. This estimate is based on data collected in Spain on the seagrass species *Posidonia oceanica* and therefore may not be entirely accurate for *Zostera* spp..

Data collected by the Food Standards Agency on the presence and quantity of *E. coli* within shellfish populations for human consumption are available from CEFAS. Areas are classified by the amount of *E. coli* within the flesh of Shellfish, see Appendix 2 for classification ranges. These data provide an indicator of the presence of bacteria within the water at a local level.

Since 1997 surveys of imposex in dog whelks have been carried out, this is thought to be the best measure of TBT levels. Therefore, the imposex survey undertaken in England would indicate the degree of TBT contamination within the SACs and subsequent water quality.

4.4 Climate regulation seagrass and maerl

Due to the limited distribution of maerl in English waters it is not considered to be a key indicator for climate regulation. However, Professor Hall-Spencer (01/09/2020) mentioned that calcifying taxa release carbon during the process of calcification. While this would be true of maerl, sediments containing carbon are stored within the maerl beds and it could be considered an indicator for climate regulation. This would be supported by Burrows *et al.* (2017) who suggest that the inorganic carbon produced is lower than inorganic carbon sequestered, and that in a Scottish MPA, maerl was the most significant habitat for storage of inorganic carbon. Therefore, there is evidence suggesting that maerl should be added as an indicator of climate regulation.

The ability of seagrasses to stabilise and accumulate sediments results in the storage of organic and inorganic carbon, and the sediment is an important repository for carbon produced within the seagrass beds and elsewhere. The sediments within seagrass beds are largely anaerobic (Duarte *et al.*, 2011), meaning that material is broken down slowly and carbon can be stored indefinitely. The estimation of sequestration rates ranges from 19 to 191 g C m^{-2} yr^{-1} (Watson *et al.*, 2020).

Carbon stocks (C_{stocks})[3] are highly variable locally, nationally and internationally with differences reported between sites on the same latitude and in close proximity to one another, as discussed by Green *et al.* (2018). The global average of C_{stocks} in seagrass soils is estimated to be 194.2 ± 20.2 Mg C ha (megagram (Mg) is the same unit as tonne (t)) which is comparable to boreal and temperate forests as well as tropical uplands (Fourqurean *et al.,* 2012). The aforementioned paper estimated that 19.9 Pg C (petagram (Pg) is a billion megagrams/tonnes) was stored in the top 1m of seagrass sediments, which equates to more than the combined global estimates of carbon emissions from fuels used for international aviation and maritime transport, fossil fuel (combustion and oxidation) and cement production in 2018 (9.98 Pg C) (Green *et al.*, 2018; Friedlingstein *et al.*, 2019). Fourqurean *et al.* (2012) reported high variability in C_{stocks} between geographic regions, from 23.6 ±8.3 Mg C ha in the Indio-Pacific to 372.4 ±74.5 Mg C ha in the Mediterranean. Green *et al.* (2018) measured the organic carbon in sediments in UK seagrass beds. The average for the southwest was 140.98 ±73.32 Mg C ha taken from 13 samples at a depth of 100cm, with a range of 98.01–380.07 Mg C ha. Green *et al.* (2018) presented site specific data for the Plymouth Sound and Estuaries SAC, the lowest and highest values for the study were both recorded within this SAC, emphasising the variability of carbon storage between sites within close proximity.

Green *et al.* (2018) used the UK government estimated traded central value of CO_2 (£24/Mg or t) to estimate the value of the C_{stocks} in UK seagrass beds at between £2.6

[3] C_{stocks} here refers to the total stock of organic carbon within the sediments of seagrass beds of a known size (Green *et al.*, 2018).

million and £5.3 million. Although it should be noted that this value (£24/Mg or t) is based on CO_2 emissions and as Friedlingstein *et al.* (2019) noted that 1Mt carbon is equal to 3.664 Mt CO_2, which suggests Green *et al.*'s (2018) estimation may be undervalued. Ganguly *et al.* (2017) estimated the global Social Carbon Cost (SCC) (value of avoided damages as a result of a unit reduction of carbon dioxide or its equivalent emissions) of seagrass as $114.43/Mg or t C and $31.21/Mg or t CO_2.

4.4.1 Suggested indicator datasets for "Climate Regulation" ecosystem service by seagrass and maerl

Burrows *et al.* (2017) suggest that the average inorganic carbon (IC) sequestration rates of maerl was 74g IC m^2 yr^{-1}. But there was no reference to quality or condition of the maerl and it is likely that this estimation would change with quality. Watson *et al.* (2020) provide a clear summary of the carbon burial estimations for seagrass (Table 8), and as this paper is very recently published, it is proposed that these estimations are used to assess the carbon sequestration potential for all of the seagrass sites.

It is proposed that C_{stocks} is used as an indicator of carbon storage in each of the SACs. Where seagrass beds can be identified from Green *et al.*'s (2018) study, the true values should be used to estimate the C_{stock} of each site and each seagrass bed. For all other locations, the southwest average should be used.

5 Potential for increased benefit

The Environment Agency has undertaken spatial modelling which considered bathymetry, wave and salinity to map the restoration potential of seagrass around the coast of England (Environment Agency, 2020). During the natural capital mapping exercise the current distribution of seagrass beds per SAC will be compared to their potential distribution. This will allow the potential for increased benefit to be estimated based on the ecosystem service indicators selected to quantify current ecosystem services.

The University of Exeter has carried out high resolution habitat suitability mapping to identify sites for restoration using a number of environmental variables and available data in the Solent Maritime and Plymouth Sound and Estuaries SACs. The results of this study are currently pending, however once available the findings will be used to select the most appropriate locations for seagrass replanting.

5.1 Seagrass restoration

As with any population reintroduction/reinforcement, the threats and causes of decline should be removed before reintroduction (or replanting) takes place (IUCN/SSC, 2013; van Katwijk *et al.*, 2016). van Katwijk *et al.* (2016) carried out a meta-analysis of seagrass restoration programs and found restoration needs to take place over large spatial scales to compensate for stochastic events and environmental stress. Replanting using weighted rhizome fragments was consistently more effective than using unanchored rhizomes or seeds, however collecting rhizomes can cause damage to the donor seagrass bed (Unsworth *et al.*, 2019). van Katwijk *et al.* (2016) concluded that close proximity to donor site and scale of restoration (>10,000 planted shoots/seeds) had positive effects on recovery.

Unsworth *et al.* (2019) assessed the effectiveness of using a "Bags of Seagrass Seeds Line" (BoSSLine) to establish *Z. marina* beds. After 10 months a 3.6% seed success was observed, with 94% of the bags developing mature shoots when deployed into a suitable environment. Selection of appropriate sites was essential; one site completely failed due to the location being too close to the intertidal zone and subject to very mobile sandy substrate. Unsworth *et al.* (2019) recommended that high resolution is used within any habitat suitability model.

5.2 Reducing traditional mooring impacts in Solent, Isles of Scilly, Plymouth and Fal and Helford (removal of up to 70 traditional moorings and replacement with Advanced Mooring Systems)

The location of moorings will be collected through a desk study by Westcountry Rivers Limited and the local Natural England site leads. Where possible locations of moorings can be mapped, the area of mooring scars per SAC can be estimated based on the findings of Unsworth *et al.* (2017) ($122m^2$), giving the possible area gain if moorings were removed. This calculation will allow the quantification of habitat increase, sediment accumulation, N and P burial as well as carbon sequestration potential of the mooring scars after the installation of advanced mooring systems through the wider ReMEDIES project.

Luff *et al.* (2019) assessed the impact of a simple and cost-effective mooring modification designed to reduce the impact of the mooring chain on the seagrass. They found shoot density was over twice as high compared to a standard mooring and blade length also exceeded that of the standard mooring. Mark Parry of the National Marine Aquarium has confirmed that there are five advanced mooring systems installed in the Plymouth Sound and Estuaries SAC.

The ReMEDIES project aims to improve the knowledge of seagrass and maerl habitats by undertaking a Behaviour Change Project. The overall aims of this project are to develop a clearer understanding of the behaviours of recreational boaters in relation to anchoring and mooring in seagrass, to facilitate the design and development of interventions to address any issues uncovered, and to evaluate the effectiveness of the interventions in order to achieve measurable behavioural changes that can lead, in the long term, to positive biodiversity outcomes. The purpose of the project is to develop evidence-based interventions to change the behaviour of recreational boaters in order to reduce disturbance and damage to seagrass caused by anchoring and mooring. The project will identify drivers for damaging practices and explore the reasons why boaters engage in these behaviours: these reasons may be rooted in individual attitudes, beliefs and practices, be associated with social factors such as the social norms and meanings which influence the way people think and act, or in the material environment which conditions their practices. The project will draw on existing literature along with information from ReMEDIES project partners, wider stakeholders and boaters, as well as direct observation, focusing on two of the SACs (Plymouth Sound and Estuaries and Solent Maritime) to define the behaviours of recreational boaters that affect seagrass and barriers to changing damaging behaviours.

5.3 Reducing anchoring impacts in Solent, Isles of Scilly, Plymouth and Fal and Helford (areas of voluntary no anchor zones)

On the Helford River (Fal and Helford SAC) there is a voluntary “No Anchor Zone” in the seagrass bed off Grebe Beach. As far as can be determined, this was implemented in 2000 and compliance is not formally monitored. There are plans for volunteers to monitor the “No Anchor Zone” next year. The Helford Moorings Officer provides ad hoc monitoring and advice when possible. The RYA is due to publish a guide to anchoring with care, which is aimed to support the Behaviour Change element of the ReMEDIES project mentioned in the previous section. Making assessments of the potential ecosystem services if anchoring pressures were to be removed is more challenging than moorings because they are harder to quantify. Moorings are permanent installations, and their impact is therefore more consistent (Griffiths *et al.*, 2017). Jessica Taylor, the Marine Lead Adviser for the Solent Maritime SAC suggested that Free AIS Ship Tracker - VesselFinder could be used to assess the numbers of boats anchoring. Although this would only show vessels with a tracker, if data could be easily extracted, it may offer a good measure of anchoring pressure, not just in the Solent Maritime SAC but across all of the SACs. Griffiths *et al.* (2017) used a similar method in their paper to assess anchoring pressure.

Where data are available an estimate of the impact of anchoring could be calculated. Surveys are being undertaken in the Plymouth Sound and Estuaries SAC to assess the anchoring pressures; these data are not readily available at this point. Anecdotal evidence suggests that in the Plymouth Sound SAC during peak season, up to 122 anchoring events per day occur at the Cawsand seagrass bed, and 50 boats per day anchor in the Yealm seagrass bed. Anchoring location will depend on wind direction. There are no anchoring data for the Isles of Scilly Complex, Fal and Helford or Solent Maritime SACs at this point.

Osborne Bay on the Isle of Wight (Solent Maritime SAC), has no moorings, but is a popular anchoring site due to its sheltered location. No data are available on the number of anchoring events on this site. It is likely that this data will be collected in the future. The Percuil (Fal and Helford SAC) is a popular area for anchoring due to its proximity to St Mawes, but there are no data on the numbers of anchoring events in this area. There is no protection for the seagrass bed in St Mawes and many people are unaware of its presence

(Personal Communications with Carolyn Waddell (Natural England Marine Lead Adviser 2020). Recent condition assessments reported losses in this area (Curtis, 2015).

It may be possible to use the RYA data as an indicator of anchoring impact. Alternatively, data could be extrapolated from the Plymouth Sound estimates, more details would be needed on popular anchoring locations to undertake this analysis.

Four free visitor moorings were installed outside the seagrass bed in North Haven (Skomer Marine Conservation Zone) to discourage boats from anchoring on the seagrass bed. The seagrass bed there saw an increase in area of 26% over 17 years (Burton *et al.*, 2015). While this increase cannot be attributed to the removal of anchoring pressure alone, this figure could provide a useful estimate when calculating the potential ecosystem service benefits of "No Anchor Zones" in the target SACs as there are limited alternative data on the effectiveness of this intervention. This calculation would allow the quantification of, habitat increase, sediment accumulation, N and P burial as well as carbon sequestration after implementation of a "No Anchor Zone".

5.4 Reducing trampling impacts from bait collection and walking in Essex and Solent

No specific data on the extent of trampling in either the Solent Maritime or Essex Estuaries SAC has been identified. Jackson *et al.* (2016) observed trampling by walkers, horse riders, dogs, kite surfers and boats launching in the Essex Estuaries SAC. Jessica Taylor (Natural England Marine Lead Adviser) advised that trampling is an issue in East Hampshire by bait diggers. This activity is illegal when carried out commercially, however differentiating personal use and commercial is very difficult.

It may be possible to assess the impact of light, medium and heavy trampling based on the findings of Eckrich and Holmquist (2000). This would allow the estimations of the increased ecosystem service potential based on different trampling reduction scenarios (heavy to medium or medium to light), although as this would be based on estimations derived from species located outside the UK, the confidence in this would be low.

5.5 Potential ecosystem services

It is possible that there are future ecosystem service benefits that seagrasses could provide. For example, as well as being a sink for heavy metals as discussed in an earlier section, seagrasses may play a vital role in storing microplastics in coastal regions (Huang *et al.*, 2020). Unsworth *et al.* (2012) found that seagrass may buffer the effect of ocean acidification for coral reefs. While this study was largely theoretical, seagrass has the potential to provide this service at a local scale and could be particularly relevant in protecting the maerl in the Fal and Helford SAC against future acidification.

6 References

Allen, C., Axelsson, M., Dewey, S., Wilson, J. (2014) *Fal and Helford SAC Maerl Drop-down Video and Dive Survey 2013*. A report to Natural England by Seastar Survey Ltd.

APEM (2019) *Testing the potential of Natural Capital Indicator Mapping in Marine Protected Areas*.

Ashley, M. (2017) *An approach to map interaction of ship anchoring with designated features within SACs and SPAs*.

Ashley, M., Rees, S.E., Mullier, T., Reed, B., Cartwright, A., Holmes, L., Shee (2020) *Isles of Scilly Natural Capital Asset and Risk Register to Inform Management of Isles of Scilly Fisheries Resources*. A report by research staff the Marine Institute at the University of Plymouth.

Borum, J., Duarte, C., Krause-Jensen, D., Greve, T. eds. (2004) *European Seagrasses: An Introduction to Monitoring and Management*. The M&MS project.

Bosence, D.W.., Wilson, J. (2003) Maerl growth, carbonate production rates and accumulation rates in the northeast Atlantic. *Aquatic Conservation Marine and Freshwater Ecosystems*. **13**(1), 21–31.

Bull, J., Kenyon, E. (2019) *Isles of Scilly eelgrass bed voluntary monitoring programme 2019 Annual Survey*. Natural England.

Bunker, F., Green, B. (2018) *Seagrass condition monitoring in Plymouth Sound and Estuaries SAC 2018*. Natural England Commissioned Report, Number294.

Burrows, M., Hughes, D., Austin, W., Smeaton, C., Hicks, N., Howe, J., Allen, C., Taylor, P., Vare, L. (2017) *Assessment of Blue Carbon Resources in Scotland ' s Inshore Marine Protected Area Network*. Scottish Natural Heritage Commissioned Report No. 957.

Burton, M., Lock, K., Clabburn, P., Griffiths, J., Newman, P. (2015) *Skomer Marine Conservation Zone. Distribution & Abundance of Zostera marina in North Haven 2014.* NRW Evidence Report No 68.

Butenschön, M., Clark, J., Aldridge, J.N., Allen, J.I., Artioli, Y., Blackford, J., Bruggeman, J., Cazenave, P., Ciavatta, S., Kay, S., Lessin, G., van Leeuwen, S., van der Molen, J., de Mora, L., Polimene, L., Sailley, S., Stephens, N., Torres, R. (2016) A generic model for marine biogeochemistry and the ecosystem dynamics of the lower trophic levels. *Geosci. Model Dev*. **9**, 1293–1339.

Chen, S., Sanford, L., Koch, E., Shi, F., North, E. (2007) A nearshore model to investigate the effects of seagrass bed geometry on wave attenuation and suspended sediment transport. *Estuaries and Coasts*. **30**(2), 296–310.

Collins, A., Miller, J., Coughlin, D., Kirk, S. (2015) The production of quick scoping reviews and rapid evidence assessments: A how to guide. . (November), pp59.

Collins, K., Suonpää, A., Mallinson, J. (2010) The impacts of anchoring and mooring in seagrass, Studland Bay, Dorset, UK. *Underwater Technology*. **29**(3), 117–123.

Cullen-unsworth, L., Mtwana, L., Paddock, J., Baker, S., Mckenzie, L., Unsworth, R. (2013) Seagrass meadows globally as a coupled social – ecological system: Implications for human wellbeing. *Marine Pollution Bulletin*. [online]. Available from: http://dx.doi.org/10.1016/j.marpolbul.2013.06.001.

Curtis, L.. (2012) *Plymouth Sound and Estuaries SAC Seagrass Condition Assessment 2012*.

Curtis, L. (2015) *Fal and Helford SAC: Subtidal seagrass condition assessment 2015*. Natural England ER15-289.

D'Avack, E., Tillin, H., Jackson, E., Tyler-Walters, H. (2014) *Assessing the sensitivity of seagrass bed biotopes to pressures associated with marine activities*. JNCC Report No:505: Peterborough, Joint Nature Conservation Committee.

Doggett, M., Northen, K.O. 2022. Condition assessment monitoring for subtidal seagrass beds, Solent Maritime SAC June 2021. A report to Natural England by Marine Ecological Solutions Ltd.

Duarte, C., Kennedy, H., Marbà, N., Hendriks, I. (2011) Assessing the capacity of seagrass meadows for carbon burial: Current limitations and future strategies. *Ocean and Coastal Management*, 1–7. [online]. Available from: http://dx.doi.org/10.1016/j.ocecoaman.2011.09.001.

Duarte, C., Losada, I., Hendriks, I., Mazarrasa, I., Marbà, N. (2013) The role of coastal plant communities for climate change mitigation and adaptation. *Nature Climate Change*. **3**(11), 961–968.

Duarte, C., Middelburg, J., Caraco, N. (2005) Major role of marine vegetation on the oceanic carbon cycle. *Biogeosciences*. **2**, 1–8.

Duffy, J.E. (2006) Biodiversity and the functioning of seagrass ecosystems. *Marine Ecology Progress Series*. **311**, 233–250.

Eckrich, C., Holmquist, J. (2000) Trampling in a seagrass assemblage: direct effects, response of associated fauna, and the role of substrate characteristics. *Marine Ecology Progress Series*. **201**, 199–209.

Environment Agency (2020) Seagrass Restoration Potential. [online]. Available from: https://data.gov.uk/dataset/5b943c08-288f-4d47-a924-a51adda6d288/seagrass-potential#licence-info.

Fonseca, M., Cabalan, A. (1992) A Preliminary Evaluation of Wave Attenuation by Four Species of Seagrass. *Estuarine, Coastal and Shelf Science*. **35**, 565–576.

Fourqurean, J., Duarte, Carlos, Kennedy, H., Marbà, N., Holmer, M., Mateo, M.A., Apostolaki, E., Kendrick, G., Krause-Jensen, D., McGlathery, K., Serrano, O. (2012) Seagrass ecosystems as a globally significant carbon stock. *Nature Geoscience*. **5**(7), 505–509.

Fourqurean, J., Duarte, Carlos M., Kennedy, H., Marbà, N., Holmer, M., Mateo, M.A., Apostolaki, E.T., Kendrick, G.A., Krause-Jensen, D., McGlathery, K.J., Serrano, O. (2012) Seagrass ecosystems as a globally significant carbon stock Supplementary Information. *Nature Geoscience*. **5**(7), 505–509.

Francois, R., Weber, J.H. (1988) Speciation of methyltin and butyltin compounds in eelgrass (Zostera marina L.) leaf tissue from the great bay estuary (NH). *Marine Chemistry*. **25**(3), 279–289.

Friedlingstein, P., Jones, M., O'Sullivan, M., Andrew, R., Hauck, J., Peters, G., Peters, W., Pongratz, J., Sitch, S., Le Quéré, C., Bakker, D., Canadell, J., Ciais, P., Jackson, R., Anthoni, P., Barbero, L., Bastos, A., Bastrikov, V., Becker, M., Bopp, L., Buitenhuis, E., Chandra, N., Chevallier, F., Chini, L., Currie, K., Feely, R., Gehlen, M., Gilfillan, D., Gkritzalis, T., Goll, D., Gruber, N., Gutekunst, S., Harris, I., Haverd, V., Houghton, R., Hurtt, G., Ilyina, T., Jain, A.K., Joetzjer, E., Kaplan, J., Kato, E., Klein Goldewijk, K., Korsbakken, J., Landschützer, P., Lauvset, S., Lefèvre, N., Lenton, A., Lienert, S., Lombardozzi, D., Marland, G., McGuire, P., Melton, J., Metzl, N., Munro, D., Nabel, J., Nakaoka, S.-I., Neill, C., Omar, A., Ono, T., Peregon, A., Pierrot, D., Poulter, B., Rehder, G., Resplandy, L., Robertson, E., Rödenbeck, C., Séférian, R., Schwinger, J., Smith, N., Tans, P., Tian, H., Tilbrook, B., Tubiello, F., van der Werf, G., Wiltshire, A., Zaehle, S. (2019) Global Carbon Budget 2019. *Earth Syst. Sci. Data*. **11**(4), 1783–1838.

Furness, E., Unsworth, R. (2022) Solent Zostera marina Health & Reproductive Surveys: June 2021. Swansea University & Project Seagrass. In collaboration with Natural England, Ocean Conservation Trust and Hampshire & Isle of Wight Wildlife Trust.

Gacia, E., Duarte, C. (2001) Sediment retention by a mediterranean Posidonia oceanica meadow: The balance between deposition and resuspension. *Estuarine, Coastal and Shelf Science*. **52**, 505–514.

Gall, A. (2012) Seasearch: Maerl in Cornwall 2012 Survey Report, available at http://helfordmarineconservation.co.uk/wp-content/uploads/Cornwall-maerl-2012-report.pdf

Gall, A. (2018) Change to the condition assessment for subtidal seagrass beds within Plymouth Sound and Estuaries SAC.

Gambi, M.C., Nowell, R., Jumars, P. (1990) Flume Observations on Flow Dynamics in Zostera marina (Eelgrass) Beds. *Marine Ecology Progress Series*. **61**, 159–169.

Ganguly, D., Singh, G., Purvaja, R., Bhatta, R., Paneer Selvam, A., Banerjee, K., Ramesh, R. (2017) Valuing the carbon sequestration regulation service by seagrass ecosystems of Palk Bay and Chilika, India. *Ocean and Coastal Management*.

Ganter, B. (2000) Seagrass (Zostera spp.) as food for brent geese (Branta bernicla): an overview. *Helgol Mar Res*. **54**, 63–70.

Garrick-Maidment, N., Trewhella, S., Hatcher, J., Collins, K., Mallinson, J. (2010) Seahorse Tagging Project, Studland Bay, Dorset, UK. *Marine Biodiversity Records*. **3**(e73), 1–4.

Green, A., Chadwick, M., Jones, P. (2018) Variability of UK seagrass sediment carbon: Implications for blue carbon estimates and marine conservation management. *PLoS ONE*. **13**(9), 1–18.

Griffiths, C., Langmead, O., Readman, J., Tillin, H. (2017) *Anchoring and Mooring Impacts in English and Welsh Marine Protected Areas: Reviewing sensitivity, activity, risk and management*. A report to Defra Impacts Evidence Group.

Hall-spencer, J., Kelly, J., Maggs, C. (2008) *Assessment of maerl beds in the OSPAR area and the development of a monitoring program*. Ireland: Department of the Environment, Heritage & Local Government (DEHLG), Ireland.

Hall-Spencer, J., Kelly, J., Maggs, C. (2010) *Background document for maerl beds*. OSPAR Commisson.

Hansen, J., Reidenbach, M. (2012) Wave and tidally driven flows in eelgrass beds and their effect on sediment suspension. *Marine Ecology Progress Series*. **448**, 271–287.

Harzing, A.. (2007) Publish or Perish. [online]. Available from: https://harzing.com/resources/publish-or-perish [Accessed August 1, 2020].

Huang, Y., Xiao, X., Xu, C., Devi Perianen, Y., Hu, J. (2020) Seagrass beds acting as a trap of microplastics - Emerging hotspot in the coastal region? Seagrass beds acting as a trap of microplastics - Emerging hotspot in the coastal region? *Environmental Pollution*. **257**(113450).

IEG (2020) *Recreational Anchoring and Mooring in Marine Protected Areas and the Potential of Eco-Moorings for Mitigation of Impacts (MMO1165)*.

Island's Partnership Yachts. [online]. Available from: https://www.visitislesofscilly.com/travel/yachts [Accessed November 2, 2020].

IUCN/SSC (2013) *Guidelines for Reintroductions and Other Conservation Translocations. Version 1.0*. Gland, Switzerland: IUCN Species Survival Commission.

Jackson, E., Cousens, S., Bridger, D., Nancollas, S., Sheehan, E. (2016) Conservation inaction in action for Essex seagrass meadows? *Regional Studies in Marine Science*. **8**, 141–150.

Jackson, E., Griffiths, C., Durkin, O. (2013) *A guide to assessing and managing anthropogenic impact on marine angiosperm habitat*. Natural England Commissioned Report NECR111.

Jackson, E., Rowden, A., Attrill, M., Bossey, S., Jones, M. (2001) The Importance of Seagrass Beds as a Habitat for Fishery Species. *Oceanography and Marine Biology: An Annual Review*. **39**, 269–303.

Jackson, E.L., Rees, S.E., Wilding, C., Attrill, M.J. (2015) Use of a seagrass residency index to apportion commercial fishery landing values and recreation fisheries expenditure to seagrass habitat service. *Conservation Biology*. **29**(3), 899–909.

Jenkins, T.L., Guillemin, M.-L., Simon-Nutbrown, C., Burdett, H.L., Stevens, J.R., Peña, V. (2021) Whole genome genotyping reveals discrete genetic diversity in north-east Atlantic maerl beds. *Evol Appl.* **14**: 1558– 1571.

Jones, B., Unsworth, R. (2016) The perilous state of seagrass in the British Isles. *Royal Society Open Science*. **3**(150596).

Jones, L., Hiscock, K., Connor, D. (2000) *Marine habitat reviews. A summary of ecological requirements and sensitivity characteristics for the conservation and management of marine SACs*. Peterborough, Joint Nature Conservation Committee. UK Marine SACs Project report.

Kamenos, N., Moore, P., Hall-Spencer, J. (2004)a Nursery-area function of maerl grounds for juvenile queen scallops Aequipecten opercularis and other invertebrates. *Marine Ecology Progress Series*. **274**, 183–189.

Kamenos, N., Moore, P., Hall-Spencer, J. (2004)b Attachment of the juvenile queen scallop Aequipecten opercularis (L.)) to maerl in mesocosm conditions; juvenile habitat selection. J*ournal of Experimental Marine Biology and Ecology*. **306**: 139– 155.

Kamenos, N., Moore, P., Hall-Spencer, J. (2004)c Maerl grounds provide both refuge and high growth potential for juvenile queen scallops (Aequipecten opercularis L.) *Journal of Experimental Marine Biology and Ecology* **313**: 241– 254.

van Katwijk, M., Thorhaug, A., Marba, N., Orth, R., Duarte, C., Kendrick, G., Althuizen, I., Balestri, E., Bernard, G., Cambridge, M., Cunha, A., Durance, C., Giesen, W., Han, Q., Hosokawa, S., Hiswara, W., Komatsu, T., Lardicci, C., Kun-Seop, L., Meinesz, A., Nakaoka, M., O'Brien, K., Paling, E., Pickerell, C., Ransijn, A., Verduin, J. (2016) Global analysis of seagrass restoration: the importance of large-scale planting. *Journal of Applied Ecology*. **53**, 567–578.

Laffoley, D., Grimsditch, G. eds. (2009) *The Management of Natural Coastal Carbon Sinks*. IUCN, Gland, Switzerland.

Lamb, J., van de Water, J., Bourne, D., Altier, C., Hein, M., Fiorenza, E., Abu, N., Jompa, J., Harvell, C. (2017) Seagrass ecosystems reduce exposure to bacterial pathogens of humans, fishes and invertebrates. *Science*. **355**(6326), 731–733.

Langmead, O., Tillin, H., Griffiths, C., Bastos, E., Milburn, H., Butler, J., Arnold, M. (2017) *EMS Recreation Study Document 04. Survey of recreational use within the Plymouth Sound and Estuaries European Marine Site: Scoping report and survey results.*

Luff, A., Sheehan, E., Parry, M., Higgs, N. (2019) A simple mooring modification reduces impacts on seagrass meadows. *Scientific Reports*. **9**(20062).

Lusardi, J., Rice, P., Waters, R., Craven, J. (2018) *Natural Capital Indicators: for defining and measuring change in natural capital*. Natural England Research Report, Number 076.

Major, W., Grue, C., Grassley, J., Conquest, L. (2004) Non-target Impacts to Eelgrass from Treatments to Control Spartina in Willapa Bay, Washington. *Journal of Aquatic Plant Management*. **42**, 11–17.

Marine Life Information Network (2022) Habitat list. Available from https://www.marlin.ac.uk/habitats (accessed 04/04/2022)

Marsden, A., Scott, A. (2015) *Inventory of eelgrass beds in Hampshire and the Isle of Wight 2015*.

Mateo, M., Cebrian, J., Dunton, K., Mutchler, T. (2006) Carbon Flux in Seagrass Ecosystems. In A. Larkum, R. Orth, & C. Duarte, eds. *Seagrasses: Biology, Ecology and Conservation*. Netherlands: Springer, pp. 159–192.

Mckenzie, L., Collier, C., Waycott, M., Unsworth, R., Yoshida, R., Smith, N. (2012) Monitoring inshore seagrasses of the GBR and responses to water quality. In *12th International Coral Reef Symposium*. Cairns, pp. 9–13.

McMahon, K., Collier, C., Lavery, P.S. (2013) Identifying robust bioindicators of light stress in seagrasses: A meta-analysis. *Ecological Indicators*. **30**, 7–15.

Milazzo, M., Badalamenti, F., National, I., Ceccherelli, G., Chemello, R. (2004) Boat anchoring on Posidonia oceanica beds in a marine protected area (Italy, western Mediterranean): effect of anchor types in different anchoring stages. *Journal of Experimental Marine Biology and Ecology*. **299**, 51–62.

Miyajima, T., Hori, M., Hamaguchi, M., Shimabukuro, H., Adachi, H., Yamano, H., Nakaoka, M. (2015) Geographic variability in organic carbon stock and accumulation rate in sediments of East and Southeast Asian seagrass meadows. *Global Biogeochemical Cycles*. **29**, 397–415.

Natural Capital Committee (2017) How to do it: a natural capital workbook. [online]. Available from:

https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/608852/ncc-natural-capital-workbook.pdf.

Natural England (2019a) *Fal and Helford - Special Areas of Conservation*.

Natural England (2018a) Solent Maritime SAC - H1110 Sandbanks which are slightly covered by sea water all the time - Subtidal seagrass beds. *Designated Sites View*. [online]. Available from: https://designatedsites.naturalengland.org.uk/MarineCondition/PublicFeatureAttributes.aspx?featureGuid=fd5dbf45-a650-e411-a6ba-000d3a2004ef&SubFeatureCode=A5.53&SiteCode=UK0030059 [Accessed September 3, 2020].

Natural England (2019b) Solent Maritime SAC - H1130 Estuaries - Intertidal seagrass beds - Extent and distribution. *Designated Sites View*. [online]. Available from: https://designatedsites.naturalengland.org.uk/MarineCondition/PublicFeatureAttributes.aspx?featureGuid=fe5dbf45-a650-e411-a6ba-000d3a2004ef&SubFeatureCode=A2.61&SiteCode=UK0030059 [Accessed September 3, 2020].

Natural England (2018b) Solent Maritime SAC - H1130 Estuaries - Subtidal seagrass beds - Extent and distribution. *Designated Sites View*. [online]. Available from: https://designatedsites.naturalengland.org.uk/MarineCondition/PublicFeatureAttributes.aspx?featureGuid=fe5dbf45-a650-e411-a6ba-000d3a2004ef&SubFeatureCode=A5.53&SiteCode=UK0030059 [Accessed September 3, 2020].

Natural England (2018c) *Solent Maritime Special Area of Conservation (SAC)*.

Newton, J. (2011) *The Impacts of commercial anchoring on maerl beds in Falmouth Bay*.

Peralta, G., Perez-Llorens, J., Hernaendez, I., Vergara, J. (2002) Effects of light availability on growth, architecture and nutrient content of the seagrass Zostera noltii Hornem. *The Journal of Experimental Marine Biology and Ecology*. **269**, 9–26.

Philippart, C. (1995) Effects of shading on growth, biomass and population maintenance of the intertidal seagrass Zostera noltii Hornem. in the Dutch Wadden Sea. *Journal or Experimental Marine Biology and Ecology*. **188**, 199–213.

Roberts, R., Barker, M., Mladenov, P. (2010). Is Settlement of Haliotis iris Larvae on Coralline Algae Triggered by the Alga or Its Surface Biofilm?. *Journal of Shellfish Research*. **29**. 671-678.

Röhr, M.E., Boström, C., Canal-vergés, P., Holmer, M. (2016) Blue carbon stocks in Baltic Sea eelgrass (Zostera marina) meadows. *Biogeosciences*. **13**(22), 6139–6153.

Romero, J., Pdrez, M., Mateo, M.A., Sala, E. (1994) The belowground organs of the Mediterranean seagrass Posidonia oceanica as a biogeochemical sink. *Aquatic Botanyc*. **47**, 13–19.

Ruiz, J., Romero, J. (2003) Effects of disturbances caused by coastal constructions on spatial structure, growth dynamics and photosynthesis of the seagrass Posidonia oceanica. *Marine Pollution Bulletin*. **46**, 1523–1533.

Selley, H. (2021) Seasearch Surveyor survey forms, Carrick Roads, Fal & Helford SAC, 16-17th Sept 2021.

Serrano, O., Lavery, P., Kendrick, G.A., Hickey, S., Masqué, P., Arias-Ortiz, A., Steven, A., Duarte, C.M. (2016) Impact of mooring activities on carbon stocks in seagrass meadows. *Scientific Reports*. **6**(23193).

Serrano, O., Lavery, P.S., Rozaimi, M., Mateo, M.Á. (2014) Influence of water depth on the carbon sequestration capacity of seagrasses. *Global Biogeochemical Cycles*. **28**, 950–961.

Short, F., Short, C. (1984) The Seagrass Filter: Purification of Estuarine and Coastal Waters. In V. Kennedy, ed. *The Estuary as a filter*. Orlando: Academic Press, pp. 395–143.

Touchette, B., Burkholder, J. (2000) Review of nitrogen and phosphorus metabolism in seagrasses. *Journal of Experimental Marine Biology and Ecology*. **250**, 133–167.

Tyler-Walters, H. & d'Avack, E.A.S., 2015. Ruppia maritima in reduced salinity infralittoral muddy sand. In Tyler-Walters H. and Hiscock K. (eds) Marine Life Information Network: Biology and Sensitivity Key Information Reviews, [on-line]. Plymouth: Marine Biological Association of the United Kingdom. Available from: https://www.marlin.ac.uk/habitat/detail/266 (accessed 04/04/2022)

Unsworth, R., Bertelli, C., Cullen-unsworth, L., Esteban, N., Jones, B., Lilley, R., Lowe, C., Nuuttila, H., Rees, S. (2019) Sowing the Seeds of Seagrass Recovery Using Hessian Bags. *Front. Ecol. Evol.*. **7**(311).

Unsworth, R., Collier, C., Henderson, G., McKenzie, L. (2012) Tropical seagrass meadows modify seawater carbon chemistry: implications for coral reefs impacted by ocean acidification. *Environmental Reseach Letter*. **7**.

Unsworth, R., Mtwana Nordlund, L., Cullen-unsworth, L. (2018) Seagrass meadows support global fisheries. *Conservation Letters*. **e12566**.

Unsworth, R., Williams, B., Jones, B., Cullen-Unsworth, L. (2017) Rocking the boat: Damage to eelgrass by swinging boat moorings. *Frontiers in Plant Science*. **8**(1309).

Watson, S., Preston, J., Beaumont, N., Watson, G. (2020) Assessing the natural capital value of water quality and climate regulation in temperate marine systems using a EUNIS biotope classification approach. *Science of the Total Environment*. **744**(140688).

Wigley, S., Paling, N., Rice, P., Lord, A., Lusardi, J. (2020) *National Natural Capital Atlas: Mapping Indicators*. Natural England Commissioned Report Number 285.

Wilson, S., Blake, C., Berges, J., Maggs, C. (2004) Environmental tolerances of free-living coralline algae (maerl): implications for European marine conservation. *Biological Conservation*. **120**, 283–293.

Wood, N., Lavery, P. (2001) Monitoring seagrass ecosystem health — the role of perception in defining health and indicators. *Ecosystem Health*. **6**(2), 134–148.

7 Appendices

7.1 Appendix 1 – QSR extended method

Harzing's Publish or Peril (V7.24.2867.7511) (Harzing, 2007) software was used to extract complete lists of search results from searches undertaken using Google Scholar and Scopus. Results were stored within a Microsoft Excel spreadsheet, where duplicates were removed. The total number of results were 550, 135 duplicates were removed, after the first pass 111 were identified as relevant, the second pass reduced this number to 67. Additional papers were added during the reviewing process. Table 9 outlines the search terms (n=10), dates and the location of search.

Table 9 - Complete list of searches undertaken in Google Scholar and Scopus

Search Term	Database	Number of Results
seagrass OR maerl OR Zostera OR Eelgrass AND "Special area of Conservation" AND UK OR United Kingdom	Scopus	2
Seagrass OR Maerl OR Zostera OR Eelgrass AND "Special area of Conservation" AND "Fal and Helford" OR "Solent Maritime" OR Scilly OR "Essex Estuaries" OR "Plymouth Sound"	Google Scholar	87
seagrass OR Maerl OR Zostera OR Eelgrass AND trampling OR mooring OR anchoring AND UK OR United Kingdom	Scopus	4
Seagrass OR Maerl OR Zostera OR Eelgrass trampling OR mooring OR anchoring "Fal and Helford" OR "Solent Maritime" OR Scilly OR "Essex Estuaries" OR "Plymouth Sound"	Google Scholar	223
seagrass OR Maerl OR Zostera OR Eelgrass "ecosystem service benefits" UK or United Kingdom	Scopus	10
seagrass OR Maerl OR Zostera OR Eelgrass "ecosystem service benefits" Fal OR Solent OR Scilly OR Essex OR "Plymouth Sound"	Google Scholar	24
Seagrass OR Maerl OR Zostera OR Eelgrass "Blue Carbon" "Fal and Helford" OR Solent OR Scilly OR Essex OR "Plymouth Sound"	Google Scholar	83
Seagrass OR Maerl OR Eelgrass OR Zostera AND "Blue Carbon" United Kingdom OR UK	Scopus	4
Seagrass OR Maerl OR Zostera OR Eelgrass "Carbon Sequestration" "Fal and Helford" OR "Solent Maritime" OR Scilly OR "Essex Estuaries" OR "Plymouth Sound"	Google Scholar	51

Search Term	Database	Number of Results
Seagrass OR Maerl OR Eelgrass OR Zostera AND "Carbon Sequestration" United Kingdom OR UK	Scopus	6

7.2 Appendix 2 – Shellfish classification

Class A (80% of samples ≤ 230 E. coli/100g; all samples must be less than 700 E. coli/100g) - molluscs can be harvested for direct human consumption.

Class B (90% of samples must be ≤ 4600 E. coli/100g; all samples must be less than 46000 E. coli/100g.) - molluscs can only be sold for human consumption:

- after purification in an approved plant, or
- after re-laying in an approved Class A re-laying area, or
- after an EU-approved heat treatment process.

Class C (≤ 46000 E. coli/100g) - molluscs can be sold for human consumption only after re-laying for at least two months in an approved re-laying area followed, where necessary, by treatment in a purification centre, or after an EU-approved heat treatment process.

There are two classification systems in England and Wales:

- The annual or "temporary" classification system
- The long-term classification (LTC) system (only applies to eligible class B areas).

The ***LIFE Recreation ReMEDIES: Reducing and Mitigating Erosion and Disturbance Impacts affecting the Seabed*** project (LIFE18 NAT/UK000039) runs from July 2019 – October 2023 and will improve the condition of five SACs between Essex and Isles of Scilly. This will be achieved by restoration, demonstration and reducing recreational pressures. Promoting awareness, communications and inspiring better care of sensitive seabed habitats will be key.

LIFE Recreation ReMEDIES is financially supported by LIFE, a financial instrument of the European Commission. ReMEDIES is led by Natural England in partnership with The Royal Yachting Association, Marine Conservation Society, Ocean Conservation Trust and Plymouth City Council/Tamar Estuaries Consultative Forum.

Working in partnership with: