Consideration of the carbon sequestration potential of seagrass to inform recovery and restoration projects within the Essex Estuaries Special Area of Conservation (SAC), United Kingdom

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Abstract
A review of available research into the blue carbon potential of seagrass was undertaken. This was then used to inform an analysis of the potential current, historic, and future value of carbon sequestered and stored in the Essex Estuaries Special Area of Conservation (SAC). The assessed status of Zostera in the SAC highlights the extent of historic loss and continued degradation of this designated sub-feature, and current water quality is incompatible with recovery or restoration. Seagrass blue carbon currently stored within the SAC equates to $\sim 18,350$ t C at a sequestration rate of $117.15$ t C yr$^{-1}$, with a lost/potential of $534,700$ t C storage capacity. The calculated financial value of current stocks (£4.6 m) is dwarfed by the lost/potential monetary value of carbon storage, £135 m, and the forfeited sequestration of £860,000-worth of carbon annually from degraded habitat. The use of carbon offset credits could help fund the huge potential for restoration that exists within the SAC.

Keywords Blue Carbon · Zostera marina · Zostera noltei · Seagrass · Essex Estuaries

Introduction
The contribution of marine and coastal habitats to the global budget for anthropogenic carbon dioxide (CO$_2$), and their value in mitigating climate change, has gained increasing recognition and is now reported by international bodies, including the Intergovernmental Panel on Climate Change (IPCC) (Luisetti et al. 2019). Vegetated habitats take up and store CO$_2$ and are capable of mitigating greenhouse gas (GHG) emissions; terrestrial ecosystems are already included in internationally-agreed economic incentive schemes to facilitate their conservation (Luisetti et al. 2019). The removal of CO$_2$ from the atmosphere by marine and coastal habitats, through photosynthesis fixing carbon in plant biomass short-term and by the trapping and burial of material from marine and terrestrial sources in sediments over longer timescales, is termed ‘blue carbon’ (Nellemann et al. 2009).

Interest in the role of coastal ecosystems in carbon sequestration and storage is strengthened by the realisation of the disproportionate capacity for delivering carbon services provided by these habitats, in comparison to their terrestrial counterparts (Irving et al. 2011; Röhr et al. 2016). Blue carbon estimates for saltmarshes and mangroves are confidently predicted, however, studies on seagrass contain large variations in values, challenging their use in delivering carbon budgets at local levels (Dahl et al. 2016; Fourqurean et al. 2012; Lavery et al. 2013; Röhr et al. 2018).

In addition to fixing and sequestering carbon, seagrass habitats provide important economic, physical, social, and biodiversity services, but are seriously impacted by environmental change resulting in significant losses, which have increased in intensity and pace damaging the integrity and condition of the ecosystems (Luisetti et al. 2019). The designation of Marine Protected Areas (MPA) was hoped to ensure conservation of marine ecosystem services in the EU and to bring these sites into ‘favourable condition’, through the implementation of conservation measures corresponding to Habitats Directive Annex I and Annex II site specific

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restoration. for financial strategies to fund conservation, recovery, and study area. It concludes with a consideration of the potential proceeds to an analysis of the blue carbon aspects of the carbon sequestration and storage in seagrass habitats and a review of the environmental constraints associated with seagrass within the Essex Estuaries SAC. It begins with

on six continents, however, only 26% of global seagrass valuable habitat covering over 300,000 km² in 159 countries, (2021).

Seagrass is a biologically rich, productive, and highly valuable habitat covering over 300,000 km² in 159 countries, on six continents, however, only 26% of global seagrass meadows currently occur within MPAs (UNEP 2020). In the northern hemisphere Zostera spp. are the most widespread with increasing biomass at higher latitudes and species able to tolerate cooler temperatures, variable depths (intertidal to 30 m), substrates (coarse sediment to fine silts), and a range of salinities (Dahl et al. 2016).

While seagrass is considered a bioindicator for ecosystem health, UK research into these habitats is lagging (Green et al. 2018). Globally there has been a paucity of management and governance for seagrass ecosystems in comparison to other coastal habitats, and a strategic approach is advocated when they are contained within MPAs (Unsworth et al. 2018). In the UK 96% of seagrass meadows occur in MPAs and these designations could be increasingly important in conserving, managing, and enhancing the blue carbon stock (Swaile et al. 2022). Since Brexit, the approach to the conservation and management of MPAs has been under review and the effectiveness of a feature-based approach has been questioned (Pelletier 2020). An ecosystem-based integrated approach for the wider seascape has been proposed to facilitate sustainability in marine management through cost-effective monitoring and evaluation across a whole site (Rees et al. 2020; Solandt et al. 2020).

A conservative average rate for the natural recovery of seagrass globally is estimated at 1.08% per decade, however this is dwarfed by continued losses and limited by unfavourable physical and water quality conditions (Irving et al. 2011). To re-establish the ecosystem services provided by seagrass, including carbon sequestration, restoration is proposed. However, these projects have proved expensive and vary in success, with consideration of environmental variables essential (Oreska et al. 2021). Extrapolating the seagrass restoration effort of the USA (up to 2011), Irving et al. (2011) calculated it would increase seagrass extent by 0.1% per decade globally and a 100-fold scaling-up of effort would be required to facilitate a 9.3% increase in carbon storage by 2260.

Restoration of natural ecosystems offers a low-risk and sustainable solution to carbon capture and storage, in comparison to artificial strategies encumbered by uncertain costs, risks, and environmental impacts, and would support other ecosystem co-benefits (Irving et al. 2011). Research suggests that it takes over nine years for full restoration of Zostera meadows, but after four years they provide 300% more carbon sequestration and storage than adjacent unvegetated areas (McGlathery et al. 2012).

**Methods**

To facilitate discussion of the potential for blue carbon services within the Essex Estuaries SAC a review of current available literature has been made to provide an overview of:
- The historic, current, and potential extent of seagrass within the SAC.
- The best accessible values for carbon storage and sequestration applicable to the species and region.
- The factors influencing carbon sequestration/storage and seagrass condition.

- The potential for restoration including evaluating carbon credit incentives.

In this systematic review of relevant evidence and data, we note: limitations created by data gaps; geographic
variations in research, evidence, and results; and low confidence figures.

**Seagrass extent estimation**

To estimate current extent, the data layers presented on MAGIC maps (in February 2022) (https://magic.defra.gov.uk/) were measured for seagrass extent within the SAC and the size of the meadows determined and combined with the observations by Gardiner et al. (2022). These results and historic information on meadow locations are presented in tables within the Results and Discussion section.

The potential seagrass habitat areas defined and mapped by the Environment Agency (2021) have been used to estimate future carbon storage and sequestration values, and to highlight the predicted loss of seagrass in the Essex Estuaries SAC. This dataset used habitat preferences, identified by the Marine Life Information Network (MarLIN), for *Zostera marina* and *Zostera noltei* including wave energy, current, turbidity, and salinity (Environment Agency 2021).

**Carbon storage and sequestration capacity calculations**

Based on work by Gregg et al. (2021) the seagrass extent, a, has been multiplied by a carbon standing stock storage rate of 39 tonnes of carbon per hectare (t C ha⁻¹), estimated for stocks up to 30 cm depths, 130 t C ha⁻¹ for stocks up to 1 m, and 0.3 t C ha⁻¹ for approximation of short-term above and below ground biomass stocks, to calculate the carbon storage capacity for seagrass within the SAC, x, (e.g. a * 39 = x t C ha⁻¹).

The sequestration rate is based on work by Burrows et al. (2014) and an annual accumulation rate calculated using an estimate of 0.83 t C ha⁻¹ yr⁻¹.

**Monetary value of blue carbon**

Following Armstrong et al. (2020), the carbon dioxide equivalent (CO₂e) of carbon was calculated based on the per tonne carbon equivalent of 3.67 t CO₂e (EPA 2022).

Using the 2020 non-traded central value price for CO₂e of £69 (from the Department for Business, Energy and Industrial Strategy (DBEIS (applicable in February 2022))) and the estimated 2050 price of £221, the current and potential monetary value for seagrass blue carbon was calculated by multiplying the per unit area of carbon stored (to 1 m depth) by the current and potential extent of seagrass within the SAC and multiplying the total by 3.67. The same method was applied for sequestration rates.

**Dissolved Inorganic Nitrogen averages**

Using opensource data downloaded from the Environment Agency (2022) and filtered to expose only Dissolved Inorganic Nitrogen (DIN) results for samples taken within the Essex Estuaries SAC, during the winter period (November to February) from 2015 to 2021, a mean for each winter season was calculated and a simple graph created to establish trends. The DIN upper threshold, identified on Natural England’s designated sites portal, was added to assess compliance with the SAC targets (Natural England 2022).

**Maps**

Maps were created using MAGIC maps (https://magic.defra.gov.uk) with layers selected to highlight the Essex Estuaries SAC and relevant features, together with opensource data imported from the Environment Agency (2021) website. These mapping and data sources provided adequate metadata to allow confidence in their use and to facilitate the construction of images that supported this study, but that are replicable for other areas and projects. Other mapping techniques were explored but required importing data from multiple sources and broader GIS skills and knowledge.

**Results and Discussion**

**Seagrass**

The diverse habitats created by seagrass meadows provide shelter and support for a rich and varied number of species (Chesman et al. 2006), and important ecosystem services benefiting coastal communities (Mazarrasa et al. 2018). Despite the relatively understudied nature of seagrass, research has found they provide 25 ecosystem services globally from ingredients for pharmaceuticals to carbon sequestration and storage, water purification, coastal protection, and nursery habitat for fisheries, with *Zostera* providing around 17 co-benefits depending on species and regional and environmentally driven physiological variations (Nordlund et al. 2016). *Zostera marina* and *Zostera noltei* are the most abundant seagrass species in the UK (Luisetti et al. 2019), but there are regional variations in distribution, with *Z. marina* the most frequently studied and predominantly found in the Southwest (Swale et al. 2022; Green et al. 2018), while *Z. noltei* is common on the intertidal mudflats around Essex (Gardiner et al. 2022).

Seagrass beds influence their chemical and physical environments by reducing current speeds within the developing meadow to encourage sediment deposition and stabilisation (Chesman et al. 2006). Root mats fasten sediments and
enable carbon to be locked in over millennia (Fourqurean et al. 2012). Zostera meadows have been shown to reduce wave height by 45 to 70% and near-bottom mean velocities by 70 to 90%, in comparison to surrounding bare sediments (Hansen and Reidenbach 2012). When species are senesced in winter their rhizomes and roots remain resistant to decomposition and continue protecting the sediments from erosion (Novak et al. 2020). However, Zostera are not a physically robust biotope and are easily degraded by trampling and other physical disturbance, particularly if growing in softer substrates (Chesman et al. 2006). One boat anchorage chain can remove 122 m² of seagrass from a meadow area (Green et al. 2018).

Extent

Despite the ecosystem benefits provided, seagrass meadows are in serious global decline, with estimates suggesting a 51,000 km² loss to 2006, and the rate of decline has increased since 1980 from <1% yr⁻¹ before 1940 to >5%, (Waycott et al. 2009) resulting in a 50% loss of extent at a rate of ~7% per annum since the 1990s (in comparison to a 25% loss of saltmarsh at a rate of 1–2% yr⁻¹ and a 20% loss of mangroves at a rate of 0.7 to 3% yr⁻¹) (Mcleod et al. 2011). An assessment of changes in European seagrass extent found that 33% of seagrass losses could be attributed to disease, coastal modification, and declining water quality, with a peak in losses in the 1970s and 80 s when Z. marina experienced an above average rate of decline, resulting in a 29% loss (35,684 ha) (de los Santos et al. 2019).

However, research found evidence of a deceleration in rates of Z. noltei decline, ascribed to improvements in water quality from specific management actions and policies adopted in the 1990s (e.g., the Water Framework Directive (WFD)) and, potentially, habitat protection resulting from designation of MPAs (de los Santos et al. 2019). Modelling of suitable coastal areas around the UK has revealed long-term seagrass losses of ~92%, equating to an area of 82,000 ha, with only approximately 8,493 ha remaining (Unsworth et al. 2021), and Luisetti et al. (2019) calculated a 1% per annum continued decline under a business-as-usual scenario.

Burton (1961), mapped seagrass meadows in Essex and noted 320 ha at Foulness, 25–30 ha at Osea Island, and considered the patches at Dengie and Tollesbury an indication of a recolonisation, with disappearances linked to the introduction of Spartina townsendii, in Hamford Water, and to the colonisation by Ulva intestinalis. In 1977, 241 ha of Z. noltei were recorded at Foulness (Wyer et al. 1977) and in 1998 Essex Estuaries SAC was described as having a nationally important population of seagrass, with specific reference to Maplin Sands (adjacent to Foulness), Dengie Flats and the north shore of Osea Island (Blackwater Estuary) (Worley and Simpson 1998). Studies by Green et al. (2021) only found data supporting 40 ha remaining at Foulness in 2005 (12.5% of the 1961 extent).

Research in 2006 stated that Zostera were only present at limited sites within the Essex Estuaries SAC, which included Maplin Sands (58 ha) and two locations on the Blackwater (a small patch at St Lawrence Bay and a strip on the northern shore of Osea Island) (Chesman et al. 2006). An Anglian region Environment Agency habitat mapping programme cited the intermixing of algal species and low densities of Z. noltei as a challenge to mapping, recording only 339.56 ha in Essex in 2016, a reduction of 16.53 ha on the area mapped in 2011 (Hambridge et al. 2018) (Fig. 2).

Jackson et al. (2016) noted increased fragmentation of meadows particularly at Foulness/Shoebury with evidence of impacts from recreational activity. Their assessment estimated a reduction in seagrass distribution within the SAC in comparison to previous surveys with thick mats of Ulvaceae noted at Osea Island and Goldhanger indicating nutrient enrichment (Table 1).

In summer 2021 shore-walking and drone surveys were undertaken by Gardiner et al. (2022) to map seagrass extent in the Stour and Orwell Estuaries (outside of the SAC) and included areas where seagrass had previously been recorded in the Blackwater Estuary. The results for the Stour and Orwell Estuaries noted many beds were fragmented or small (<10 m²), with a total coverage estimated at 10.1 ha, representing a 97.1% decline from the 345 ha present in 1973.

In the Blackwater Estuary Zostera was only recorded as a meadow in St Lawrence Bay, and Ruppia maritima (Widgeonweed) observed at Goldhanger (no surveys were undertaken at Osea Island). The total coverage recorded was 0.1 ha in comparison to 37 ha present in 1976 (Gardiner et al. 2022). Despite some ambiguity in the data, declines based on the 1930s-60 s baseline show losses in the Blackwater continued at the same magnitude between 1970 and 2014 as they did between 1930 to the 1990s and the MPA is in unfavourable condition (Gardiner et al. 2022), with seagrass currently representing only 0.3% of the SAC’s total area. The baseline of approximate extent, calculated at 141.14 ha (Table 2), has been used to review the current potential, and value, of seagrass blue carbon within the Essex Estuary SAC.

Blue carbon

The main blue carbon habitats in the UK are saltmarsh and seagrass (Legge et al. 2020), with research into ‘donor’ habitats investigating the value of macroalgae beds to inform their inclusion in blue carbon strategies (Trevathan-Tackett et al. 2015). Currently unvegetated marine sediments are not included under the blue carbon habitat term, but they do store large volumes of carbon (from marine and terrestrial
sources) (Legge et al. 2020) and are increasingly being recognised and evaluated as part of national coastal and marine carbon estimates (Parker et al. 2021).

It is increasingly acknowledged that vegetated coastal ecosystems have a vital role in carbon sequestration and storage (Röhr et al. 2018) and, despite the small area of the world’s oceans occupied by seagrass habitats (less than 0.2%), they are estimated to bury 10% of the total organic carbon sequestered by the ocean (Fourqurean et al. 2012), with seagrass accounting for 20% of the planet’s blue carbon capacity (Röhr et al. 2016) and absorbing 33% of the CO₂ generated from anthropogenic sources (Röhr et al. 2018). Coastal seagrass habitats are conservatively calculated to hold between 4.2 and 8.4 Pg carbon within the sediments and living biomass (Fourqurean et al. 2012). The average salt-marsh carbon sequestration rates in the UK are estimated at

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**Fig. 2** Current mapped seagrass locations within the Essex Estuary Special Area of Conservation created using the Water Framework Directive Habitats: Higher Sensitivity layer for intertidal seagrass beds (A2.61) (England) and insert including the Estuaries (H1130): Mudflats and sandflats not covered by seawater at low tide (H1140) polygons layer, to include seagrass beds mapped around Osea Island. (Source: https://magic.defra.gov.uk)
4.4 to 5.5 t CO$_2$ ha$^{-1}$ yr$^{-1}$ compared to $\sim$5.06 t CO$_2$ ha$^{-1}$ yr$^{-1}$ for temperate seagrass species, and 0.06 t CO$_2$ ha$^{-1}$ yr$^{-1}$ for continental shelf sediments (Stafford et al. 2021).

Rapid seagrass declines are gauged to result in a loss of sequestration equating to between 6 and 24 Mt C yr$^{-1}$, the remineralization of 1.5% yr$^{-1}$ of meadows resulting in 11.3 to 22.7 Mt C yr$^{-1}$ released back into the atmosphere–ocean system from biomass decomposition, and 63 to 297 Mt C yr$^{-1}$ from oxidization of the top 1 m of eroded/disturbed sediments (Fourqurean et al. 2012). These emissions from degraded and lost seagrass habitats are estimated to contribute 33% of the blue carbon CO$_2$ that is remineralized (Pendleton et al. 2012), together with a reduction in future carbon sequestration (Legge et al. 2020), and historic UK losses equating to 11.5 Mt C (equivalent to CO$_2$ emissions from 7.7 million cars per annum) (Unsworth et al. 2021), leaving the current UK capacity estimated at 0.9 Mt C (Green et al. 2021).

To assess the blue carbon potential of seagrass, the factors that impact on sequestration and storage of carbon within the biomass and sediments of beds needs to be investigated, including routes to remineralization and subsequent CO$_2$ emissions resulting from meadow disturbance and degradation (Mazzarasa et al. 2018). Studies by Kindeberg et al. (2018) found large spatial variations for the storage of carbon in estuarine areas, with inner estuaries storing significantly more blue carbon than outer estuaries. Research has also highlighted that organic carbon volumes vary with depth (Novak et al. 2020). However, analysis by Green et al. (2018) validated the extrapolation of data from shallower (30 cm) samples, using deeper 1 m cores to ground truth results.

The source of carbon in seagrass sediments has been estimated as 50% from autochthonous origins (Kennedy et al. 2010) and research on Zostera spp. indicates a $>$ 40% contribution from allochthonous sources (Röhr et al. 2018). It is noted that most studies focus on the organic carbon content of sediments and research into the inorganic content is limited. However, calcium carbonate (CaCO$_3$) could be important particularly in areas where sediments have an active bivalve community (Macreadie et al. 2017; Prentice et al. 2020).

### Table 1
Changes in location of *Zostera spp.* observed at sites within Essex Estuaries SAC 1961 to 2014 – where blue indicates presence, brown absence and white areas not surveyed—(based on Table 1 from Jackson et al. (2016))

| Site                          | 1961 | 1977 | 1996 | 1998 | 1999 | 2006 | 2011 | 2013 | 2014 |
|-------------------------------|------|------|------|------|------|------|------|------|------|
| Crouch (Havengore Head)       |      |      |      |      |      |      |      |      |      |
| Colne (Point Clear)           |      |      |      |      |      |      |      |      |      |
| Blackwater (Tollesbury/Goldhanger) |      |      |      |      |      |      |      |      |      |
| Blackwater (Osea Island)      |      |      |      |      |      |      |      |      |      |
| Blackwater (St Lawrence)      |      |      |      |      |      |      |      |      |      |
| Dengie Flats                  |      |      |      |      |      |      |      |      |      |
| Foulness (Maplin sands/Shoebury) |      |      |      |      |      |      |      |      |      |

### Table 2
Current extent of intertidal seagrass meadows measured at present and known historic sites within the Essex estuaries Special Area of Conservation as of February 2022 (based on MAGIC (2022); Gardiner et al. (2022))

| Essex Estuaries SAC seagrass | Hectare | Substrate                  | Notes                                           |
|------------------------------|---------|----------------------------|------------------------------------------------|
| Crouch (Havengore Head)      | 8.6     | mud and sand               | Adjacent to saltmarsh                           |
| Colne (Point Clear)          | 2.3     | intertidal soft sediments  |                                                |
| Blackwater (Tollesbury/Goldhanger) | 2.3     | sand                       |                                                |
| Blackwater (Osea Island)     | 8.6     | intertidal soft sediments  | On the Marine SAC complex feature layer but not the WFD habitats layer |
| Blackwater (St Lawrence)     | 0.64    | mud and sand               |                                                |
| Dengie Flats                 | 121     | mud and sand               | Northern section adjacent to salt-marsh         |
| Total                        | 141.14  |                            |                                                |
Duarte and Cebrián (1996) gauged that 25% of carbon associated with seagrass primary production is exported to other adjacent ecosystems, introducing a potential bias in globally defined blue carbon budgets. Whilst meadows may average 20% more stored carbon than unvegetated areas, it is important to assess adjacent unvegetated areas to facilitate the calculation of the net value of the seagrass habitat for blue carbon (Potouroglou et al. 2021).

Estimates for carbon stored in seagrass sediments in England still rely largely on global and European studies, along with limited research from the West and Southwest, and caution needs to be applied when making comparisons across areas with different species, abundance, climate, and environmental conditions (Gregg et al. 2021). Globally, there are limited evaluations of carbon stocks within Z. noltei meadows (Potouroglou et al. 2021).

Studies of carbon storage capacity for Z. noltei and Z. marina meadows in ten estuaries across Scotland produced varying values for the top 50 cm, ranging from 14.94 Mg C ha\(^{-1}\) to 105.72 Mg C ha\(^{-1}\), and Potouroglou et al. (2021) reported that their results were higher than the average calculated for temperate North Atlantic regions, approximated at 48.7 Mg C ha\(^{-1}\), comparable with worldwide studies for Z. marina (108.9 Mg C ha\(^{-1}\)), but below global averages (194.2 Mg C ha\(^{-1}\)). Variations in units of measure and depths applied across regional studies, however, hamper direct comparison. IPCC guidance recommends results are reported to 1 m sediment depth (IPCC 2013), but much of the data reviewed presents figures for depths from 25 to 50 cm (Table 3).

Storage values in English studies undertaken by Green et al. (2018) (~3,372.47 g C m\(^{-2}\)) are within the upper range for other European estimates for Z. marina (averaging between ~500 g C m\(^{-2}\) to circa 4,324.5 g C m\(^{-2}\) in the top 25 cm) (Table 4). Organic C stocks and sequestration rates noted were highly variable, but broadly in line with averages for other temperate or Zostera seagrass studies, however, figures were significantly lower than those for global seagrass averages (Prentice et al. 2020).

Estimates of the carbon content in Zostera biomass vary, both in value and in units of measurement, and sequestration rates for the UK have been based on studies undertaken in the Mediterranean and North-East Atlantic (Fourqurean et al. 2012), with Luisetti et al. (2019) estimating accumulation of 2,500 t C y\(^{-1}\) and Armstrong et al. (2020) presenting a sequestration rate of 0.027 kg m\(^{-2}\) yr\(^{-1}\). Green et al. (2021) concluded there were no reliable datasets on which to found accumulation estimates for the UK.

After evaluation of the studies reviewed calculations for the Essex Estuaries SAC were based on the data considered most appropriate from a regional and species perspective. The current extent of seagrass in the SAC was multiplied with the values for carbon stored, in sediment (39 t C ha\(^{-1}\)) and biomass (0.3 t C ha\(^{-1}\)), formulated by Gregg et al. (2021) and rates sequestered per hectare (0.83 t C ha\(^{-1}\) yr\(^{-1}\)) determined by Burrows et al. (2014). This suggests an approximate volume of carbon stored and sequestered by seagrass within the SAC of 18,348.20 t C (to 1 m depth), a biomass short-term stock of 42.34 t C, and an annual sequestration rate of 117.15 t C yr\(^{-1}\), a fraction of the possible historic levels.

When evaluating the EA ‘potential’ seagrass habitat present within the Essex Estuaries SAC, it was noted that the

| Geographic location | Sediment organic carbon content in grammes of organic carbon per meter squared (gOC m\(^{-2}\)) | Depth cm | Study reference |
|---------------------|-----------------------------------------------|----------|----------------|
| Pacific Northwest   | 600–5,125                                     | 25       | Prentice et al. 2020 |
| Pacific Northwest average | 1,752                                        | 25       | Prentice et al., 2020 |
| Northern Europe     | 500–3500                                      | 35       | Dahl et al. 2016 |
| Denmark             | 627–6,500                                     | 25       | Röhr et al. 2016 |
| England (Secretary of State responsibility) | 5,900–38,000                                 | 30       | Parker et al. 2021 |
| England (Secretary of State responsibility)—average | 1,370                                         | 30       | Parker et al. 2021 |
| Average across all Temperate regions | 2,721                                         | 25       | Röhr et al. 2018 |
| Australia           | 262–4,833                                     | 25       | Lavery et al. 2013 |
| Australia average   | 1,262                                         | 25       | Lavery et al. 2013 |
| Pacific Northwest – average extrapolated to 1 m | 6,526                                         | 100      | Prentice et al. 2020 |
| 36 degrees latitude | 218–26,523                                    | 25       | Röhr et al. 2018 |
| 36 degrees latitude—average | 2,721                                         | 25       | Röhr et al. 2018 |
| 36 degrees latitude – extrapolated to 1 m | 23.1–351.7 Mg C ha\(^{-1}\)                     | 100      | Röhr et al. 2018 |
| Global (all seagrass stocks) | 13,970                                        | 100      | Fourqurean et al. 2012 |
zones in Fig. 3 do not match all the areas where seagrass occurs today, with an overlap noted only at Osea Island. Therefore, the current and potential future hectarage of seagrass meadows have been combined, less the 8.6 ha mapped at Osea Island, to estimate future storage and sequestration values (Table 5a–c) and to discuss the loss of blue carbon capacity.

The available literature highlights large variations in sedimentary carbon stocks globally, regionally, and locally, however, these differences are predominately attributed to variations in environmental conditions and particularly sediment characteristics (Legge et al. 2020). It is important to understand the interaction between the physical, chemical, and biological factors influencing carbon storage in seagrass sediments (Samper-Villarreal et al. 2016; Serrano et al. 2016), and studies assessing environmental factors governing blue carbon sequestration and stocks have increased. However, spatial scales, characteristics, regions, and species studied tend to be limited, and there is a paucity of data to inform local conservation and restoration initiatives (Mazzarrasa et al. 2018) but understanding the factors that drive variations in sediment stocks is vital to help develop effective management strategies (Lima et al. 2020).

### Environmental constraints

Researchers have studied different environmental variables individually, or in limited combinations, to assess impacts on carbon storage and accumulation in seagrass including, plant structure; water depth, quality, and clarity; wave energy;
Fig. 3 Environment Agency mapped potential habitat areas and current mapped seagrass locations within the Essex Estuary Special Area of Conservation using the Water Framework Directive Habitats: Higher Sensitivity layer for intertidal seagrass beds (A2.61) (England) only. (Source: https://magic.defra.gov.uk)
Consideration of the carbon sequestration potential of seagrass to inform recovery and…

exposure; substrate; and anthropogenic pressures (Kennedy et al. 2010; Duarte et al. 2013; Oreska et al. 2017b; Röhr et al. 2018; Dahl et al. 2020). Differences in carbon stocks between species are up to 18-fold in magnitude (Lavery et al. 2013) and Prentice et al. (2019) found water motion and seagrass structural complexity two of the main factors influencing sediment carbon. Zostera spp. exhibit a high ratio of refractory organic compounds in their rhizomes and the lignin rich biomass decomposes slowly offering a decay-resistant carbon source (Dahl et al. 2016; Novak et al. 2020). Canopy and rhizome system complexity facilitate the trapping of particles and generate an autochthonous organic carbon source (Dahl et al. 2016; Samper-Villarreal et al. 2016). Remineralization and resuspension rates are also increased by limits in canopy (Röhr et al. 2018). Conversely, Röhr et al. (2018) found biomass variations of less importance in determining carbon storage capacity than the % mud content, degree of sorting, and dry density of sediments; the salinity and depth of water; and volume of allochthonous sources.

Hydrodynamics appear to exert an influence on a regional rather than a meadow scale and areas with lower water motion sequester and store more carbon (Prentice et al. 2019). Substrate characteristics are interconnected with exposure, as hydrodynamics influence sediment density, mud content, and degree of sorting (Kennedy et al. 2010). In high-energy environments the top 10 cm of sediment within meadows can be eroded and deposition of finer particles reduced, influencing the capacity of the seagrass bed to sequester and store carbon (Arias-Ortiz et al. 2018). Climate change induced increases in hydrodynamic forces also negatively impact the ability of seagrass to act as a natural carbon sink (Dahl et al. 2018). Research has shown that seagrass growth is modified by tidal currents, with changes in the magnitude of current strength and ebb or flood tide dominance influencing water column constituents and plant productivity (Passeri et al. 2015).

Increased water depth and salinity are also shown to influence productivity, lowering the rate of sediment carbon stock formation, whilst high temperatures can increase the mineralization rates of carbon (Röhr et al. 2018). The impact of water depth on carbon storage capacity has been linked to irradiance, limiting plant productivity and seagrass density, a factor also associated with eutrophication induced changes in water clarity (Serrano et al. 2014). Whilst, Mazarrasa et al. (2017) did not find a correlation between carbon burial and water depth, seagrass meadows at shallower depths do show higher rates of blue carbon (Samper-Villarreal et al. 2016).

There is a trade-off between meadow structure and current velocities when estimating blue carbon (de los Santos et al. 2019), with small or fragmented seagrass patches storing around three times less carbon than an equal area of intact meadow (Oreska et al. 2017a; Röhr et al. 2018).
Studies have shown that sediment carbon stocks within meadows are influenced by an ‘edge effect’, and concentrations increase with distance from the perimeter by around 20% (Ricart et al. 2015). However, carbon from macroalgal sources remains consistent, indicating material advection and in-situ macroalgal growth within meadows (Oreska et al. 2017b). Oreska et al. (2017a) advocate the inclusion of adjustments for the heterogeneity caused by meadow size and edge effect in assessments of blue carbon stocks, but most calculations are based on a uniform per unit area value for the whole seagrass bed, resulting in a potential to overestimate carbon value (Ricart et al. 2015).

Researchers have repeatedly noted a positive correlation between sediment grain size and carbon content (Oreska et al. 2017a; Röhr et al. 2018; Prentice et al. 2020), with fine grained sediments having a greater available surface area, higher porosity (Dahl et al. 2016), and more effectively binding organic carbon (Novak et al. 2020). These sediment characteristics are reasoned as the core consideration when estimating the blue carbon capacity for Zostera meadows and evaluating protection and restoration areas (Dahl et al. 2016). In studies of Z. marina in the Solent (UK), Lima et al. (2020) found the sediment characteristics were important determinants, supporting this conclusion, and research by Röhr et al. (2016) compared Z. marina meadows and found that >40% of variations could be attributed to sediment characteristics with only 10% related to the contribution of plant detritus.

**Water quality**

Although the causes of seagrass loss are complex and multifaceted, the two main factors attributed to seagrass decline are physical impacts, linked to coastal development and dredging, and water quality (Waycott et al. 2009; Govers et al. 2014), associated with agricultural run-off and sewage discharges resulting in eutrophication (Unsworth et al. 2021). The level of nitrogen found in plant tissue has been used as a bioindicator to assess the state of UK seagrass in comparison to global means, and results averaged 75% higher, with evidence of weakened seagrass resilience contributing to an overall evaluation of poor condition (Jones and Unsworth 2016).

In comparison to macroalgae species, seagrass have a relatively low nutrient requirement and can flourish in nutrient-poor environments, with eutrophication a limitation on growth and survival due to smothering from epiphyte mass and restrictions in light (Wang et al. 2017), which impact the positive carbon balance within seagrass leaves (Jones and Unsworth 2016). Seagrass species themselves display variations in responses to enrichment with Z. marina able to tolerate pulsed nitrate enrichment episodes through its ability to maintain function during dark periods, a response that can weaken them structurally (Burkholder et al. 2007). However, any resilience has a tipping point when loss becomes rapid (Bertelli et al. 2021).

*Zostera marina* and Z. noltei stop producing anti-microbial compounds when stressed by eutrophication, increasing their susceptibility to pathogenic attack from *Labyrinthula zosteriodes*, responsible for wasting disease (Burkholder et al. 2007; Bertelli et al. 2021). This disease has resulted in an estimated 90% die-back of *Z. marina* populations in the north Atlantic region since the 1930s (Waycott et al. 2009). A period concurrent with increased usage of herbicides, and recent studies by Hughes et al. (2018) have connected the presence of Diuron (from agricultural run-off and anti-fouling paints) with higher incidence of *L. zosteriodes* infection. Susceptibility to wasting disease is also increased by biochemical reactions to light inhibition from turbidity (Hughes et al. 2018) and sea temperature increases (Bull et al. 2011), and the fungus is thought to persist at low levels in the Essex estuaries (Chesman et al. 2006).

Seed driven meadow recovery may be constrained initially more by nutrient availability than limitation in light, with seedling development only supported in the early stages by the nutrients in their hypocotyls (Wang et al. 2017). Recolonization of meadows from the available seedbank could be rapid after a water quality crisis if nutrient levels are high (Burkholder et al. 2007). However, seagrass seeds are only viable for one year (Novak et al. 2020) and repeated or persistent eutrophic conditions would successively reduce the seedbank and hinder long-term recovery (Burkholder et al. 2007).

Eutrophication also increases ammonium levels and the production of sulphide in sediments, which can rise to toxic levels in porewater causing large-scale seagrass mortality (Govers et al. 2014). A correlation between continued declines in seagrass after improvements in water quality have been linked to sulphide intrusion in sediments (Fraser and Kendrick 2017). There are variations in species response to ammonium levels with *Z. marina* less resistant to high porewater levels of sulphide than *Z. noltei* but more tolerant of eutrophication in the water column (Govers et al. 2014).

Water quality conditions also influence the trophic web associated with seagrass, with the loss of higher trophic level species resulting in overgrazing of seagrass plants and/or elevation in bioturbation increasing turbidity (Macreadie et al. 2017). Conversely, eutrophication causes conditions that stress and reduce mesograzers, a key manager of algae within meadows that effectively reducing its ability to smother seagrass (Burkholder et al. 2007). Eutrophication has also been linked to infraunal changes, with an increase in dominance shifting from bivalves to polychaete biotopes (for example *Nereis diversicolor*, a robust bioindicator of pollution) (Ramalhosa 2005). This polychaete is ubiquitous
in the estuaries of the Southern North Sea (Gardiner et al. 2022) and its presence is connected to the failure of previous trials to re-establish Z. noltei in Essex estuaries (Chesman et al. 2006).

Climate change

The impacts of climate change on seagrass productivity and blue carbon potential also need to be considered. The consequences of climate change are likely to impact on the environmental constraints of seagrass habitats with increasing temperature, sea levels, and ocean acidification, together with pressures on coastal areas from escalating human activity (Mazarrasa et al. 2018).

The structure and function of temperate seagrass meadows are already impacted by sea temperature rises and further increases will influence distribution and growth patterns as temperature becomes a limiting factor (Kim et al. 2020). Models indicate climate change and sea level rise will result in a poleward shift in range for Z. noltei as habitat suitability decreases, particularly where landward migration of intertidal areas is squeezed by anthropogenic structures (Valle et al. 2014).

Essex Estuaries SAC environmental conditions and water quality

The hydrodynamics along the SAC’s outer coast are complex, and the abundance of Zostera on the Maplin Sands mudflats is attributed to the isostatic balance between erosion and accretion rates and low amplitudes of surface oscillation (< ± 7 cm y⁻¹) at the site, with any trend towards increased submergence or erosion considered detrimental to meadow survival (Frazer 1978). Tidal velocities in the Blackwater and Crouch estuaries are influenced by flood defences, and the Blackwater is currently an ebb dominant system, with net losses of material at the estuary mouth, whilst the Crouch is flood dominant and material is accreting (Essex and South Suffolk SMP 2010). The inshore areas of the Essex Estuaries SAC are predominantly characterised by finer sediment size, predicted to support seagrass beds with high carbon accumulation and storage rates, with offshore (> 1 km) sediments coarser (Worley and Simpson 1998) (Fig. 4).

The majority of seagrass found in UK MPA’s are within waterbodies experiencing eutrophication pressures and exhibiting a high coverage of macroalgae, with the Essex Estuaries SAC classified as having moderate status (i.e., below the threshold for good ecological status) for dissolved inorganic nitrogen (DIN) and, therefore, in unfavourable condition for nutrients (Swale et al. 2022). Sediment and water quality issues in the SAC have been linked to nutrient enrichment leading to macroalgal blooms and hyper-nutritification, along with an unquantified threat to seagrass beds from micropollutants including pesticides and herbicides (Chesman et al. 2006).

To assess nutrient status within the SAC Dissolved Inorganic Nitrogen (DIN) levels are used and the DIN winter level target of below 1.38 mg/l is the requirement for favourable status (Natural England 2022). Examination of the available Environment Agency (2022) data shows that the Essex Estuaries SAC does not consistently meet the threshold set and is currently on a declining trend (Fig. 5).

Blue carbon valuation

The study highlights that when estimating the economic value of ecosystem benefits provided by seagrass, caution is required as bioregional, species, and environmental conditions influence the number and extent of services, and transferring values calculated for one locality could result in an unreliable base for another (Nordlund et al. 2016). The processes that influence carbon sequestration and storage are multiplex and area specific, and analysis of hydrodynamic regimes, sediment structure, seagrass species, and levels of eutrophication are required to evaluate blue carbon storage potential and financial value (Macreadie et al. 2015; Dahl et al. 2016; Samper-Villarreal et al. 2016; Röhr et al. 2016, 2018; Oreska et al. 2017b; Mazarrasa et al. 2018; Kindeberg et al. 2018; Novak et al. 2020).

Nevertheless, using the blue carbon value of seagrass to access financing for restoration and conservation strategies could provide opportunities to offset the cost of these management plans and contribute to national strategies to reduce greenhouse gas (GHG) emissions (Mazarrasa et al. 2018). The influence of environmental variations suggests a precautionary approach to global estimates of seagrass blue carbon potential and emphasise the need to refine calculations to a local scale (Röhr et al. 2018).

Carbon offset credits

Financial initiatives to conserve and restore coastal carbon and other ecosystem services are possible, with carbon trading initiatives and economic incentives now available (Röhr et al. 2018). Following inclusion of seagrass within the IPCC definition of a wetland, and their incorporation within guidance for GHG accounting, nations have begun to included seagrass in National Determined Contributions for climate change adaptation, and carbon markets reflect the changes in developing methodologies (Lovelock and Duarte 2019). However, quantitative assessments of blue carbon fluxes and validation of stores are required to facilitate further entry into financial offsetting mechanisms (Crooks et al. 2011).

The Verified Carbon Standard (VCS) approved carbon credit methodology has specific criteria that need to be met...
to prove they are truly offsetting GHG emission. Needelman et al. (2018) demonstrated that seagrass restoration projects would exceed the 5% additionality score and, therefore, be eligible for carbon credits under the scheme within the USA. Research has found ‘credible’ evidence that seagrass ecosystems could provide GHG offset benefit, quantifying atmospheric GHG removal (including GHG flux) and defining a benchmark for financing seagrass restoration through offset-crediting (Oreska et al. 2020). The IPPC have set a default rate for seagrass carbon sequestration (0.43 t C ha$^{-1}$ yr$^{-1}$) and research by Oreska et al. (2020) supported this value for initial restoration up to 10 years, but further research is considered necessary to confirm that the rate is applicable globally.

There is concern that the volume of organic material exported from seagrass beds, which contributes to the carbon sequestered in sediments outside of meadows, could result in an underestimate of seagrass blue carbon value (Duarte and Krause-Jensen 2017). However, Hejnowicz et al. (2015), highlight the risk of double counting in carbon offset credit calculations where exchanges in organic matter through exportation, burial, and movement up trophic levels, could result in duplications of estimates for carbon sequestration sink and source ecosystems.

Market prices for carbon offset-credits currently offer a limited incentive for seagrass restoration schemes, despite a seagrass off-set accounting framework being included in the VCS Programme since 2015, and voluntary uptake is
limited due to the uncertainty surrounding their GHG offset benefits and the need to increase voluntary offset prices to fully finance projects (Oreska et al. 2020). However, the requirement for businesses to evidence mitigation of their carbon emissions and corporate social responsibility (CSR) could increase buyer interest in blue carbon credits, particularly if the economic value of other ecosystem benefits are traded alongside GHG offsetting, enhancing the financial and environmental return and motivation to invest (Kuwae et al. 2022).

Critiques emphasis that any carbon credits sold or purchased must be for a project that could not have been funded from any other source and care needs to be taken that ‘poor quality’ offset credits do not result in corporations and nations continuing to emit greenhouse gases, rather than truly investing in sustainable practices, or facilitating emission of increased GHG contributions (Toadvine 2021). Therefore, the verification and validation of the offset credit market needs to be independent to maintain credibility, increase confidence in the system, and to facilitate increased uptake, with best practice guidance from demonstration projects expanded (Kuwae et al. 2022).

**Essex Estuary SAC calculations**

Studies for England have concentrated on the South and West and on *Z. marina*, or, have relied on best available data from non-UK sources (Parker et al. 2021). Green et al. (2018) estimated UK carbon standing stock values as between £2.6 and £5.3 million and one of the largest reported in Europe. Welsh national estimates for the monetary value of blue carbon used the 2020 non-traded (central) price for CO₂e of £69 per tonne and concluded that annual sequestration benefits from marine habitats are worth approximately £6.6 million, based on an accumulation of 0.03 Mt C yr⁻¹, with the non-traded carbon prices set to treble by 2050 (to £221) further increasing the value of blue carbon (Armstrong et al. 2020).

Using the methodology outlined in Armstrong et al. (2020) an estimate of the current financial value of seagrass blue carbon in the Essex Estuaries SAC has been made (Table 6). The evaluation of the potential lost capacity for carbon storage in the SAC is approximately £419 m with forfeited sequestration rates of £2.6 m per annum, far exceeding the current estimates of £4.6 m stored in mapped seagrass meadows (based on 2050 prices).

Valuing carbon sequestered and stored is complex and meeting the requirements for the calculation of carbon offset values for seagrass recovery and regeneration projects within
the SAC will require quantification of current storage and sequestration rates, sources, and sinks. Data and evidence are needed to demonstrate any project provides additional GHG benefits that would not occur without the funding (Needelman et al. 2018).

**The future**

**Seagrass restoration**

Historic losses of seagrass in the UK are estimated at 92%, equating to a significant potential for the restoration of thousands of hectares of habitat, which could contribute to hundreds of thousands of tonnes of future carbon sequestration and storage, but only if the costs (estimated at £70,000 ha⁻¹) can be reduced, or mitigated through improved methodologies (Stafford et al. 2021) and financing strategies. Including the value of carbon stored within seagrass, and other vegetated coastal ecosystem sediments, could provide a mechanism for prioritising conservation and management based on their carbon value (Lovelock et al. 2017). However, habitat restoration and creation projects are not instant solutions due to the lag time between creation and full carbon sequestration and storage potential being realised (Gregg et al. 2021).

Attempts to restore coastal vegetated habitats, including seagrass, are considered by Irving et al. (2011) to have limited potential for substantially increasing carbon capture and storage, unless larger scale restoration projects are implemented, and ask if investment of finite financial resources would not be better aimed towards natural recovery and limitation of further losses. Conservation of saltmarsh and seagrass is considered more sustainable, effective, and economic than restoration and provide valuable co-benefits from other ecosystem services (Luissetti et al. 2019).

Currently the blue carbon value of habitats in MPA’s is not a management consideration but integrating the carbon storage and sequestration benefits into management targets could enhance protection of valuable blue carbon ecosystems and facilitate restoration or conservation projects regarded as Nature-based Solutions for climate change (Burrows et al. 2021). Considering the increased recognition and understanding of the value of marine habitats for blue carbon services, the limited availability of carbon offsetting financial mechanisms, to support restoration and habitat re-creation projects, is disappointing. However, it is acknowledged that accurately calculating carbon storage and sequestration values to allow VCS to be met is difficult (Armstrong et al. 2020).

It is also recognised that loss and degradation of blue carbon habitats releases stored CO₂ back into the atmosphere and management and protection within the MPA networks should be a priority, with mechanisms for sourcing funding from blue carbon finance an option that requires investigation (Howard et al. 2017). Seagrass restoration projects have a poor record of success and, whilst trends are improving, the reversal of the current rate of decline and loss is vital, with inclusion in carbon budgeting and mitigation programmes a potential for future protection (Röhr et al. 2018). To pay the ecological debt from historical seagrass losses and increase the meaningful potential for climate change mitigation in the short-term, restoration projects would need to be significantly up-scaled and efficiency in delivery improved (Crooks et al. 2011). The inclusion of biodiversity premiums for habitat and species conservation may add additional scope to funding management strategies (Crooks et al. 2011), particularly where links to improving nursery habitat for commercial fisheries can be made.

To maximise the potential for seagrass restoration, its presence and value need to be reconnected to public awareness and understanding of the commercial, physical, environmental, and social ecosystem services it provides increased (Orth et al. 2006). Water quality improvements are also of primary importance for the recovery of seagrass and other blue carbon habitats in coastal areas (Unsworth et al. 2018). Whilst the extent of decline in the UK offers significant potential for restoration and recovery (Jones and Unsworth 2016; Green et al. 2021). There are many limitations associated with this piece of research not least the lack of baseline data on the extent of seagrass currently standing within the SAC. It is acknowledged that mapping seagrass, specifically subtidal meadows,
has been difficult, particularly in areas with predominately turbid waters, like the Essex Estuaries SAC, and assessing the below ground biomass a further challenge. However, remote sensing offers opportunities to address the difficulties, with protocols in place allowing delineation of seagrass meadow extent even in areas with heterogeneous bathymetry, turbidity, and biophysical characteristics, provided the results are subject to rigorous ground truthing (Chanda et al. 2022).

Water quality improvement

Although research has shown that catchment scale water quality, including coastal waters, is fundamentally associated with land management (Webber et al. 2021) and impacted by an estuary’s physical characteristics, with those dominated by large tidal ranges and open areas of mudflats, as seen within the Essex Estuaries SAC, more susceptible to eutrophication (Maier et al. 2009) policy drivers, regulations, and legislation do exist to facilitate long term improvements in water quality. The Government’s 25 Year Environment Plan (25 YEP) aims to create Nature Recovery Networks that will bring freshwater, transitional water habitats, and protected sites, into favourable condition through the establishment of ‘Environmental Land Management Schemes’ (ELMS) (Gov.UK 2022a). These and other catchment-based approaches, including Catchment Sensitive Farming (CSF) and Countryside Stewardship, aim to reduce diffuse pollution, suspended solid inputs, and develop nature-based flood management solutions through habitat creation, however, they are subject to economic constraints that influence uptake by land managers and without sufficient financial incentives positive management strategies may not be implemented (Webber et al. 2021).

Sustained investment by water companies in natural environmental programmes, that encompass estuaries, to address nutrient pollution should be expanded, with enforcement of robust legislation implemented to ensure that investment in infrastructure is more financially attractive than the payment of fines (Environment Audit Committee 2018). Natural England (NE) are also advocating ‘nutrient neutrality’, by advising local planning authorities to ensure new plans and projects can demonstrate they will not increase the nutrient loads in protected freshwater and estuary environments (Gov.UK 2022b). Along with the Water Framework Directive (WFD), it is evident that policies and drivers do already exist to improve water quality.

However, although the European Union (EU) review of the Water Framework Directive (WFD) has assessed that the legislation is predominately fit for purpose, funding and implementation are not considered sufficient to achieve its goals (European Commission 2019). A view supported by the UK 2018 Parliamentary Review, which found that deficiencies in Environment Agency resources and fragmentation of regimes could undermine the 25 YEP (Environment Audit Committee 2018). Carvalho et al. (2019) highlight the need for the WFD to integrate with other water policies including those that cover flooding, wastewater treatment, energy, and climate, and to enhance the design of monitoring networks and diagnostic tools to facilitate a more evidence-based approach to river basin management. This alignment of regulations, engaging multi-sector stakeholders and encompassing a more holistic, longer term, and robustly monitored approach to pollutants, their sources and impacts, is advocated to tackle causes and sources of water pollution and create sustainable improvement in water quality (Environment Audit Committee 2018; Webber et al. 2021).

Conclusions

The seagrass extent of the Essex Estuaries SAC is small, despite the national significance attached to the beds on Maplin Sands. Currently meadows are being fragmented and the habitat is still in decline, with water quality factors incompatible with recovery, putting the remaining areas of seagrass under further pressure. However, the potential for restoration within the SAC is extensive and the sediment characteristics and hydrodynamic conditions could support good levels of carbon sequestration and storage, but the poor water quality would impact attempts to re-establish seagrass within the estuaries.

The status of the water quality and the continuing decline in seagrass extent mean the Essex Estuaries SAC is unlikely to meet the criteria to be deemed in favourable condition. It is recommended that habitat conditions, including improving water quality, are optimised before expensive restoration projects are instigated, even if funding from carbon credits or other sources could be secured.

Reviews of current regulations have determined that the necessary legislation, policies, and drivers are in place and fit for purpose (WFD, CSF, etc.), however, these do need to be aligned and integrated with adequate resources provided to improve monitoring and assessment, deliver financial incentives, and where necessary enforcement, giving the relevant agencies the teeth they require. Improving multi-stakeholder engagement, with longer-term strategic aims, beyond the current 6-year cycles, and catchment to coast sustainable and nature-based management approaches are recommended for achieving sustainable water quality improvements.

Seagrass could have a role in the UK’s battle against climate change through the conservation and restoration of this valuable habitat and use of carbon offset credits. However, the poor baseline information available locally
and nationally, and the lack of regional research, limit the potential of such initiatives that would be expensive and need to be implemented at scale.

Improvements in the WFD monitoring network and assessment systems to incorporate a wider suite of surveying, and to include earth observations and remote-sensing technologies, would increase the scope and availability of baseline data. This data needs to be open sourced, analysed and formulated into accessible reports and mapping tools. In the meantime, Project Seagrass and ReMEDIES have encouraged collaboration between agencies (EA and NE), the Essex Wildlife Trust, and the University of Essex, which has led to opportunities for citizen science and academic projects to broaden the scope of baseline information that will be available for much of the Essex Estuaries SAC area in the future. This will support nascent efforts to conserve and restore seagrass and its blue carbon potential, whilst working to fulfil the designated site’s management aims and bring the MPA into favourable condition.

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