Investigating the Importance of UK Seagrass Habitats and Current Conservation Action

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Abstract

Since 1936, at least 44% of UK seagrass has been lost and the deterioration of seagrass habitats is predicted to continue. Seagrass habitats are rich in species diversity and have the ability to sequester large quantities of carbon. Initially, a systematic review investigated threats to UK seagrass habitats, current policies, and conservation measures in place that cover UK seagrass habitat. Results from this review indicate that most reported threat in the primary literature to seagrass within the UK is eutrophication. However, current policies within the UK focus on physical disturbances to seagrass, highlighting a failure to address the most commonly reported threats to seagrass. Protecting seagrass habitats maximises the abundance of commercial fish and bivalve species within the area supporting local fisheries. *Lebensspuren* present within the area were measured to determine diversity within the area. *Lebensspuren* refers to life trails left by a species such as a shell or worm casts. A total of 300 photo quadrats were taken across three sites across the UK: Exmouth, Anglesey, and Grimsby. Results stated that there was a significant positive correlation between *Z. noltii* density and *lebensspuren* abundance and biodiversity within the area. This emphasises the importance of seagrass habitats as a habitat for numerous species along the UK coastline and emphasises the need to protect seagrass habitats from degradation to maximise density. Seagrass habitats account for 10% of marine carbon sequestration globally allowing to removal of atmospheric carbon dioxide, and hence reducing greenhouse gas concentrations. Literature searches returned three studies investigating organic carbon levels within UK seagrass sediments. Each of the studies reported a positive relationship between seagrass density and abundance and organic

carbon levels. However, there was a great variability in organic carbon levels reported at each site. While denser seagrass patches lead to increased carbon burial, further research is needed to determine a more accurate picture of organic carbon levels within UK sediments. The addition of appropriate protective policies designed to offset frequent threats to seagrass such as eutrophication will lead to a higher abundance and biodiversity of species along the UK coastline as well as leading to the increased burial of carbon, offsetting UK carbon emissions

Contents

Chapter 1: Introduction

1.1 Importance of seagrass habitats

Coastal vegetated habitats such as seagrass habitats may provide a pivotal role in the burial of organic carbon in the sediment (Kim et al., 2022). Seagrass habitats have been reported to be responsible for 10% of oceanic carbon burial globally as opposed to 9% buried by mangrove habitats and a further 25% by salt marsh habitats (Campbell et al., 2015, Duarte et al., 2005). Organic carbon refers to carbon found within organic molecules (Holland and Turekian, 2014). Organic carbon found within the surrounding water becomes trapped within the canopy by tightly packed, entangled seagrass shoots (Figure 1) where it settles to the benthos and becomes buried in the root matrix for long term storage (Macreadie et al., 2014, Tanaya et al., 2018). Buried carbon then remains within the sediments as long as the seagrass habitat remained undisturbed, after which time the stored carbon may be released into the water column (Macreadie et al., 2014). By protecting seagrass habitats from degradation and promoting dense seagrass habitats, the amount of buried carbon within the sediment can be maintained, and dispersal into the surrounding habitats prevented (Marba et al., 2015).

Seagrass habitats bury autochthonous carbon (carbon sourced from within the seagrass habitat) and allochthonous carbon (carbon sourced externally to the seagrass habitat) and also photosynthesize which removes CO_2 from the water column as seen in Figure 1(Ganguly et al., 2017, Kim et al., 2022, Tanaya et al., 2018). The resulting CO_2 removal then leads to an increase in pH levels within the local area (Semesi et al., 2009). The removal of CO_2 from the surrounding water allows for more CO_2 to become dissolved in the water (Benson and Cole, 2008) by increasing the concentration gradient. (Macreadie et al., 2017).

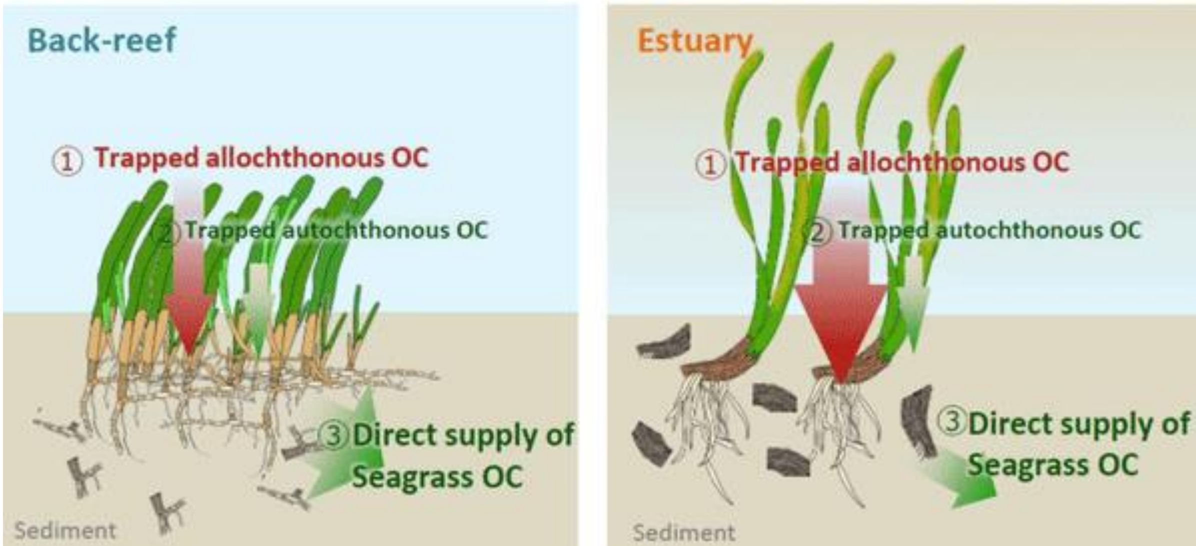

Figure 1 Diagram displaying how allochthonous and autochthonous carbon can become trapped within the seagrass canopy leading to its long-term storage. Left of the figure shows an area of Thalassia hemprichii dominated back-reef with the right of the figure showing Enhalus acoroides dominated estuarine environments (Tanaya et al., 2018).

Seagrass has the ability to stabilise sediments and reduce sediment suspension into the water column (Reidenbach and Timmerman, 2019). This allows seagrass habitats to become self-stabilizing, replacing particles that are lost during wave disturbance (Roger and Koch, 2004). When the habitats are able to maintain this level of optimized stability, they are able to reduce wave velocity leading to a reduction in coastal erosion in the area (Johnson et al., 2019). As waves approach seagrass beds, they induce hydrodynamic drag and lose energy. Such observations have been observed within *Zostera marina* meadows and *Posidonia oceanica* meadows (Luhar et al., 2017). Seagrass beds act as a barrier to redirect water direction and reduce the hydrodynamic nature of the seabed (Avdeev et al., 2006). The drag created by the redirection of water leads to a decrease in the waves velocity (Hansen and Reidenbach, 2013). Reduction of wave action has been key in the reduction of coastal erosion (Innocenti et al., 2018, Feagin et al., 2019).

1.2 Biodiversity within seagrass habitats

Seagrass habitats may act as hotspots for biodiversity of fish species throughout temperate and sub-tropical waters (Hyman et al., 2019). The tightly packed shoots and dense canopy provide juvenile commercial fin-fish species such as mullet (Mugilidae), herring (Clupeidae) and snapper (Lutjanidae) with the ideal nursey habitat (Unsworth et al., 2018b, Nordlund et al., 2018). The complex habitat provides protection and nutrition for juvenile fish as a result of the complex food web formed, reducing their mortality rate (Olson et al., 2019). It is reported that seagrass habitats also provide strong connective networks to other coastal habitats such as kelp forests and saltmarshes, which local fauna migrate towards as they mature (Espadero et al., 2021).

Dense seagrass beds provide protection for juvenile commercial fish species such as plaice (*Pleuronectes platessa*) and haddock (*Melanogrammus aeglefinus*) preventing their predation before they are able to reach maturity (Bertelli and Unsworth, 2014). Providing fish with a nursery habitat allows for more individuals to reach maturity and reproduce, regenerating the population (Lande, 1988, Lefcheck et al., 2019). Promoting healthy seagrass habitats has been reported as vital for supporting fisheries (Unsworth et al., 2018b).

Not all fish species leave seagrass habitats upon reaching maturity. The same networks that protect juvenile fish species also provide the optimal habitat for the survival and reproduction of pipefish and seahorse species (Syngnathidae) (Scapin et al., 2018). Dense seagrass habitats provide seahorses which a rich abundance of holdfasts to grip onto as well as providing offspring with protection from predation (Curtis and Vincent, 2005). Seahorse and pipefish species are often used as flagship species to draw public attention to seagrass habitats situated within UK waters (Shokri et al., 2009).

While seagrass habitats provide protection from predation to juvenile fish from larger fish, the habitats also provide a plentiful food source for many larger species (Curtis and Vincent, 2005). The canopy of tropical and subtropical seagrass meadows provides a key food source to green sea turtles (*Chelonia mydas*) (Lal et al., 2010, Sarkis et al., 2022). Another large grazer that feeds on subtropical seagrass is the manatee (Lefebvre et al., 2017). Manatees (*Trichechus manatus*) within US waters feed primarily on a diet of seagrass however the recent decline of seagrass habitat has forced manatees to search for other food sources to ensure their survival (Allen et al., 2022).

Invertebrate species such as crabs and scallops equally find benefit in seagrass habitats globally (Hovel and Fonseca, 2005, Irlandi et al., 1995). *Callinectes sapidus* (Blue crab) are found primarily within US seagrass habitats (Seitz et al., 2005). The beds provide optimum habitat for blue crabs, providing them with protection and a plentiful food source of molluscs and arthropods (Ralph et al., 2013). Echinoderms are found throughout benthic habitats, including seagrass habitats (Teoh and Woo, 2021, Susetya et al., 2019). Seagrass habitats throughout the Mediterranean provide shelter, food and nurseries for echinoderms (Coulon and Jangoux, 1993). Echinoderms feed on detritus within the habitat and larger echinoderms such as Echinoidea (sea urchins) feed on smaller, herbivorous echinoderms such as Holothuroidea (sea cucumbers) limiting grazing populations and preventing overgrazing (Lannin and Hovel, 2011; Muzaki et al., 2019).

The root-rhizome layer created by seagrass beds provides a home to polychaete assemblages (Box et al., 2010). As the canopy of seagrass meadows traps particles of detritus and incorporates them into the sediment it provides a rich and plentiful food source for polychaetes promoting their survival (Gambi et al., 1991). Decreases in seagrass abundance has led to a decrease in trapped detritus particles, minimising abundance of associated polychaeates that the habitat can support (Omena and Creed, 2004). Seagrass beds provide bivalves with protection from turbidity of localized wave activity (Reusch and Chapman, 1995). Filter feeding mussels remove waste from the seagrass habitat and increase nutrient levels within the sediment (Peterson and Heck, 2001), which then increases the health and productivity of seagrass habitats (Cardini et al., 2022).

1.3 Decline in coastal marine habitats

Habitat loss and degradation are prominent threats to coastal marine ecosystems (Teichert et al., 2018). From tropical coral reefs (Maxfield, 2018) to temperate kelp forests (Deza and Anderson, 2010), marine habitats are experiencing degradation and declines in area. This reduction in coverage has resulted in a reduction of species diversity, particularly among species which rely on a specific habitat type, for example freshwater ponds (Horváth et al., 2019). Anthropogenic factors, such as coastal development and eutrophication are commonly found to be the cause of coastal habitat decline(Luff et al., 2019; Rabalais et al., 2009). For example, increased coastal development and land reclamation has led to a decline in seagrass habitats with at least 44% of UK seagrass being lost since 1936 (Gu et al., 2018, Green et al., 2021). Kelp habitats are experiencing decline and degradation as a result of climate change (Raybaud et al., 2013, Fredriksen et al., 2020) and seagrass habitats are currently experiencing degradation from over-nutrification as a result of eutrophication (Burkholder et al., 2007). Increased eutrophication induces nutrient stress on seagrass habitats, leading to decreased rates of growth and restoration (Fuggle et al., 2023).

Increases in air and sea temperatures causes inhibition in the growth of seagrass seedlings, preventing regeneration of the habitat (Pereda-Briones et al., 2019; Román et al., 2022). The seasonal nature of seagrass species such as *Zostera marina* means that the ability of the habitat to regenerate is crucial for the ongoing success of the habitat during its regrowth in future summers (Hemminga and Duarte, 2000). Such impacts are significant as sea surface temperatures are expected to rise between 2°C and 5°C by the end of the 21st century as a result of climate change (Holopainen et al., 2016). Waters surrounding the UK have seen an increase of 0.3°C per decade over the previous 40 years with models predicting an increase of 3°C by the end of the 21st century (MCCIP, 2023)

Habitat loss and fragmentation has been found to affect many coastal marine habitats including seagrass habitats (Ding et al., 2020, Swadling et al., 2023). Anthropogenic disturbances in the form of terrestrial run-off and the building of marine infrastructure have led to the fragmentation of benthic habitats within coastal marine habitats (Swadling et al., 2023). Increases of habitat fragmentation have led to reductions in recruitment and abundance of fish assemblages associated with kelp forests (Deza and Anderson, 2010) and decreased amphipod abundance within seagrass habitats (Sweatman et al., 2017).

Increases in bottom-contact fishing has led to an increase in benthic habitat disturbance (Thrush et al., 1998, Broad et al., 2020). Use of bottom contact fishing gear such as dredges and trawls lead to the disturbance of sediment and benthic dwelling fauna (Burnett and Sarà, 2019). Burrowing organisms living within soft sediments are key to maintaining the structure and stability of an ecosystem however such species are at risk of becoming dislodged, injured or die as a result of the use of bottom contact fishing gear within benthic environments (Thrush and Dayton, 2002). Bivalves are filter feeders that are able to remove detritus from surrounding water, creating a more habitable environment for fish species (Dame, 2012). Disturbances to sediment leads to a loss of bivalve species and therefore increased levels of detritus within surrounding water (Piersma et al., 2001). The removal of bivalves from seagrass habitats has led to an increase in detritus reducing water quality which leads to reduced light penetration needed for seagrass photosynthesis (Cardini et al., 2022).

1.4 Seagrass habitat decline

Since 1990, global coastlines have experienced losses of seagrass at an estimated rate of 7% per year as a result of biotic and abiotic threats such as climate change and eutrophication (Cullen-Unsworth et al., 2018, Perry et al., 2020, van Katwijk et al., 2010). Seagrass degradation has led to 15% of seagrass species being identified as Endangered in accordance with the IUCN Red List making some seagrass habitats one of the most threatened marine habitats globally (Evans et al., 2019, Hughes et al., 2009, Waycott et al., 2009, Pazzaglia et al., 2021). Since the 1980s, 39% of seagrass has been lost from UK waters with up to 100% seagrass loss in areas such as Suffolk (Green et al., 2021).

Declines in seagrass populations have been reported to be as a result of numerous factors, both anthropogenic and as a result of natural causes such as disease (Duarte, 2002, Groner et al., 2014). Increased sea surface temperatures have been shown to limit the growth of seagrass and increase the mortality rate leaving up to 75% of Mediterranean seagrass meadows at risk of severe, irreparable degradation (Pruckner et al., 2022, Traboni et al., 2018). Increases in storms and extreme weather events have led to an increase in wave action (Chen et al., 2018). Seagrass habitats are unable to adjust to significant increases in wave action leading to the uprooting of shoots and further degradation to the habitat (Suykerbuyk et al., 2016). At lower wave intensities seagrass beds are able to reduce wave velocity (Hansen and Reidenbach, 2013), however, higher intensities can lead to the damage and dislodging of seagrass beds and their inhabitants causing them to be washed away (Samper-Villarreal et al., 2016, Reusch and Chapman, 1995). While seagrass habitats are able to restore localized pH levels to counteract ocean acidification (Semesi et al., 2009), larger decreases in pH levels weaken the structure of seagrass shoots and can inhibit photosynthesis levels leading to restrictions in growth (Christianen et al., 2011).

Humans have also had a direct impact on the degradation of seagrass habitats through the increase in physical disturbances (Broad et al., 2020). Increases in boating have led to significant physical damage to seagrass meadows (Carreno and Lloret, 2021). Propeller scars and anchoring cause damage and dislodging of seagrass shoots allowing them to be washed away (Pergent-Martini et al., 2022, Glasby and West, 2018). Increased boat traffic often brings with it increased pollutants in the form of fuel spills and littering which pollute the area and can lead to reduced water quality (Ansanelli et al., 2016). Reduced water quality leads to a decrease in light availability to an area, reducing rates or photosynthesis by seagrass (Campbell et al., 2007, Lapointe et al., 2020). Further to this, increased boating leads to increased wave action (Bilkovic et al., 2019). Increases in wave action lead to the breakdown of seagrass shoots leading to their eventual degradation and removal from the habitat (Hansen and Reidenbach, 2013).

Physical disturbance to seagrass habitat has also resulted from bottom contact fishing gear and aquaculture activities. Benthic fishing techniques such as trawling have led to the uprooting and removal of seagrass within UK seagrass beds (Collie et al., 2000, Islam et al., 2014, McMahon et al., 2022). Fish farms have also been found to cause a decrease in shoot recruitment and an increase in shoot mortality within the Mediterranean (Apostolaki et al., 2011). Introducing fish farms near seagrass habitats has been found to increase herbivory pressure and waste production from the increased abundance of fish within the area which led to increasing damage to seagrass shoots within the habitat and reducing its ability to regenerate (Ruiz et al., 2010, Kletou et al., 2018).

Habitat fragmentation has occurred throughout seagrass habitats globally as a result of habitat destruction as a result of climate change and physical disturbances from anthropogenic sources (Deza and Anderson, 2010, Hovel and Lipcius, 2001). Fragmented habitats have an increased percentage of the habitat that can be found on the edge of the habitat, experiencing edge effects (Yarnall et al., 2022). Edge effects refer to the difference in pressures that are faced by the interior of the habitat but become more prominent towards the edge of the habitat such as predation (Laurance et al., 2007). Larger predators are unable to enter densely packed habitats such as seagrass beds and so such predators feed primarily on organisms towards the exterior of the habitat (Andrén, 1995). The fragmentation of seagrass habitats has led to a smaller area in which fauna are protected from predation and therefore leading to a decrease in predated species such as *Callinectes sapidus* (Blue crab) (Mizerek et al., 2011, Laurance et al., 2007). Crustacean population densities were shown to have decreased as seagrass beds began to fragment (Shaban et al., 2016), whereas pipefish such as *Stigmatopora argus* and *S. nigra* on the other hand were found to be more abundant at the edges of fragmented habitats (Smith et al., 2008).

1.5 Declines in UK seagrass habitats

Seagrass habitats within the UK have experienced significant decline (Duarte, 2002). At least 44% of UK seagrass has been lost since 1936 as seen in Figure 2 (Green et al., 2021). Seasonal decline of seagrass is common with *Zostera marina* degrading over winter and dispersing seeds to encourage regrowth the next summer (Hemminga and Duarte, 2000). Annual seagrass regrows and regeneration is inhibited by that of ocean acidification and nutrient stress from eutrophication leading to a failure to reach densities seen within the area in years prior (Carus et al., 2021; Chefaoui et al., 2018; Garrard and Beaumont, 2014). Single shoots, patches and meadow edges lead to an increase of localized turbulence, leading to the erosion towards the edges of seagrass habitats (Maxwell et al., 2017). Due to the such factors leading to the isolation and decline of seagrass patches, remaining UK seagrass meadows have been seen to have lower genetic diversity leading to reduced ecological success (Alotaibi et al., 2019).

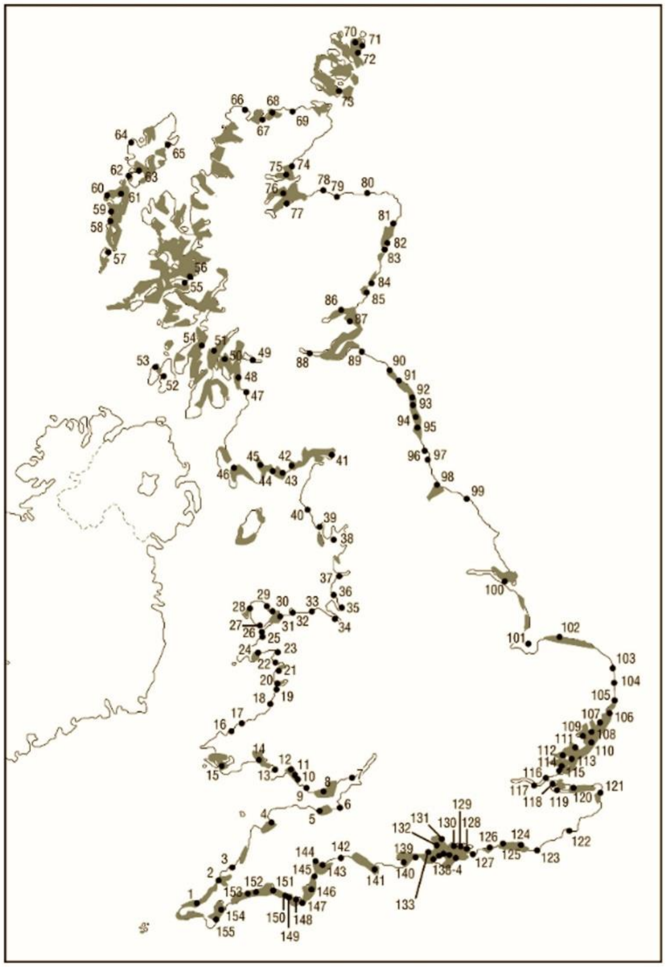

Figure 2 Map showing known seagrass beds surrounding the UK coastline during the 1930s as shaded grey and known seagrass beds during the 1990s as numbers (Green et al., 2021).

1.6 Aims of the current study

This research was divided into three sections. Chapter 2 investigated the threats currently faced by seagrass habitats and conservation and management action plans in place for seagrass habitats in the Europe. A systematic review aimed to assess threats to seagrass habitats as they were reported within the academic literature. A review of UK conservation measures for seagrass was also conducted to evaluate the conservation and management action in place.

Chapter 3 aimed to investigate the relationship between shoot density and *lebensspuren* diversity within UK *Zostera noltii* beds. *Lebensspuren* refers to the life trails left behind by organisms within the area, such as tracks, shells and worm casts (Allaby, 2020). There has been a decrease in the density of seagrass and within recent years projects have taken place to reintroduce and replant seagrass beds. This research aimed to investigate if increased shoot density led to higher abundances and diversity of *lebensspuren* present.

Finally, Chapter 4 used pre-existing literature to investigate and evaluate methods used to estimate the rates of carbon burial within UK waters. Seagrass habitats have been identified as ideal blue carbon habitats to aid in the mitigation of global carbon emissions (Greiner et al., 2013), however, there is limited standardisation across methods for measuring carbon storage. This research aimed to improve understanding of current limitations in carbon storage estimates from seagrass habitats.

Chapter 2: Investigating threats to UK seagrass habitats and their existing protective policies.

2.1 Introduction

Over the coming decades it is expected that ¾ of the world's population will live among coastal regions, increasing coastal development and putting more pressure on coastal marine habitats . Over the previous century these increased pressures have led to a decrease in seagrass habitats (Mazarrasa et al., 2017). Seagrass habitats are considered under threat with declines in both UK and global populations (Burke, 2004), and individual seagrass communities suffering habitat fragmentation (Johnson and Heck Jr., 2006). Within recent decades, seagrass habitats have also shown wider degradation through direct damage, for example, benthic fishing methods and damage from boating (Suykerbuyk et al., 2016). The use of bottom contact fishing gear such as trawls has led to damage and removal of seagrass shoots from the habitat (Meyer et al., 1999) as well as the removal of target species such as shrimp and also bycatch incorporating other crustaceans, molluscs and fish found within the benthic habitat (Stallings et al., 2014).

Seagrass habitats provide a place for fish to grow and thrive with reduced rates of predation from larger predators (Bertelli and Unsworth, 2014). Lower rates of predation ensure that a higher percentage of juvenile fish mature to their full size allowing for increased reproductive rates and therefore increased population sizes (Baillie et al., 2015).

Seagrass species are prominent parts of the global coastline, populating 160 000 km or 14% of coastal areas (Strachan et al., 2022). Seagrass habitat provides numerous ecosystem services such as carbon sequestration and the removal of localized CO_2 (Apostolaki et al., 2011). Seagrass species are primary producers which allow for the removal CO_2 through photosynthesis while the habitat uses only a small area (Duarte et al., 2013). This ability of seagrass to take in and remove this carbon from the environment has allowed the habitat to become a key part of strategies to mitigate against anthropogenic CO_2 levels (Marba et al., 2015).

This research aims to i) investigate the most reported threats faced by seagrass habitats within Europe within the peer-reviewed literature ii) review what protection measures are currently in place to protect seagrass habitats within European waters and iii) evaluate whether the current protection measures are suitable for mitigating against the threats that faced by UK seagrass habitats.

2.2 Method

2.2.1 Threats to seagrass habitats

2.1.1 Systematic review:

A systematic review was used to assess threats to seagrass that have been reported within literature. The review allowed for an objective approach to the analysis of publicly available scientific literature using repeatable methods and defined search criteria (Kendrick et al., 2019). The most frequently reported threats to seagrass were then compared to existing protective policies in the UK to establish whether existing policies mitigated the threats that local seagrass habitats are experiencing. The Web of Science database was used for this search. This was chosen due to the reliability of the sources present. The literature search was conducted in December 2022.

Step 1:

Three different primary search terms were selected and searched using Web of Science (Table 1). These terms were selected using names of species commonly found throughout Europe. Three secondary terms were selected to allow for the search to be narrowed to review literature including threats, dangers and risks to seagrass habitats. Data was taken up until December 2022.

Step 2:

To ensure each piece of research answered the initial question, the titles, and abstracts of each were read. Any that did not reference threats, dangers or risks to seagrass populations were removed as well as any results that had appeared multiple times within the search (duplicates). A total of 250 articles were gathered initially and upon the deletion of duplicate papers, 118 remained.

Step 3:

Research was to be focused on Europe and so each paper was read to identify the geographic region covered by the research. Any that referred to locations within Europe or spoke about global threats to seagrass remained, however, any research focusing on locations outside of Europe were removed. A total of 51 papers remained once papers that did not relate to the review due to geographical location or differences in subject matter were removed. Geographical locations noted within literature were collated and used to form a map illustrating the frequency of which data was collected from the countries. The year in which publications were released was also recorded in order to compare rates of publications and rates of seagrass decline.

Step 4:

Each paper was read to identify mentions of threats to seagrass. Many projects identified more than one threat and so each threat mentioned was recorded separately. Words relating to anthropogenic threats within the paper were recorded.

Table 1: Primary and secondary search terms used in the systematic review.

Primary search terms	Secondary search terms
Seagrass	Threats
Posidonia	Dangers
Zostera	Risks

2.2.2 Policies protecting seagrass habitats

A grey literature review was undertaken to assess policies protecting UK seagrass on a UK, EU and international level. Policies were obtained through searches of government websites such as NatureScot and GOV.UK as well as through reviews of scientific literature citing policies. Policies included those that protected specific species native to the UK such as *Zostera marina* and *Zostera noltii* as well as policies that protected seagrass as a habitat. The search began with international conventions which aimed to protect seagrass habitats on a global or continental level. To further this research, investigations took place to search for polices enacting these conventions on a more localized level. Each policy was recorded identifying the policy or convention, a brief description, a description of how the policy relates to seagrass habitats, the area covered by the policy and finally, the date the policy was enacted. Upon gathering this data, it was compared to that of the threats to seagrass within Europe to assess how effective these policies are and whether the threats are being suitably addressed.

2.2.3 Spatial protection and Marine Protected Areas (MPA)

To assess spatial protection, maps of UK protected areas such as RAMSAR sites, Sites of Special Scientific Interest (SSSi) and Marine Conservation Zones (MCZ) were examined. Seagrass Spotter by Project Seagrass was used to build the most accurate picture possible of known seagrass beds within the UK and Ireland. Seagrass Spotter was used due to its open access nature allowing for the inclusion of civilian science to provide the most up to date records. These were then cross referenced with JNCC maps indicating the location and types of MPAs. Both data sets were collated in QGIS in order to create a map displaying all known seagrass beds and MPAs within the UK and Ireland. Shapefiles for countries were obtained through government websites and shapefiles for MPAs were obtained through the JNCC website (JNCC, 2020). Each MPA that encompassed a seagrass bed was recorded. Along with documenting the location and size of each MPA including seagrass the protective policies enforced in each area were noted and whether or not seagrass was listed as a protected feature was noted. Forms outlining details concerning the habitat such as location and protection measures were obtained through the JNCC website to allow analysis of management plans and to what extent these plans are enforced.

2.3 Results

2.3.1 Threats to European seagrass

Data was compiled from 15 countries throughout Europe (Figure 3). Mediterranean countries dominated research discussing threats to seagrass habitats with most research being completed within Spain with a total of 10 papers. Research was seen to be lower within countries surrounding the Baltic Sea.

Figure 3: Map showing the frequency at which threats to seagrass were investigated within each country.

Seagrass habitats experienced significant declines within the 1990s and early 2000s (Chefaoui et al., 2018; Duarte, 2002). Research within the sector did not begin to escalate until 2010, with the majority of research taking place from 2018 onwards (Figure 4).

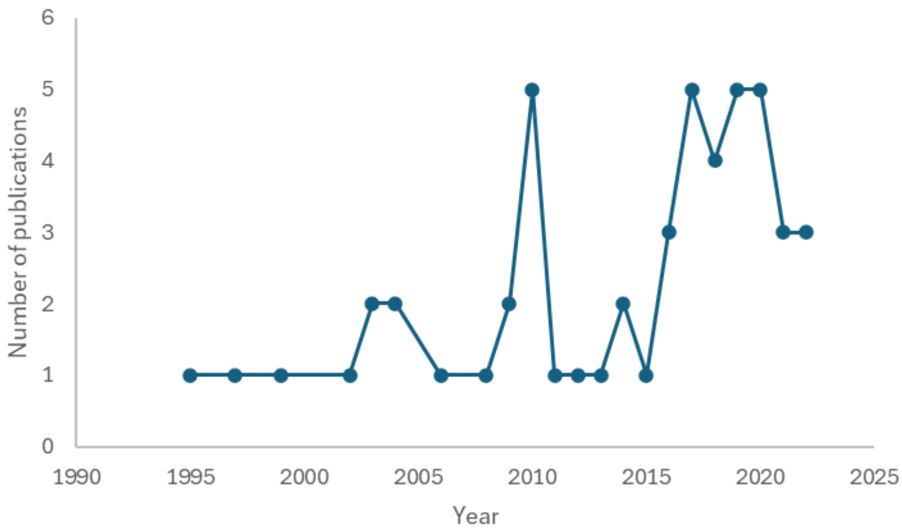

Figure 4: Graph showing the number of papers published invesitgating threats to seagrass habitats within Europe per year.

Eutrophication was the most reported threat to seagrass in the primary literature encompassing all species throughout Europe (26 out of 51 papers). Eutrophication was defined as the runoff and input from agricultural sources with particular influence on fertilizers and herbicides such as atrazine leading to over nutrification causing such factors as algal blooms and inhibition of growth within the habitat (Tedengren, 2021, van Katwijk et al., 2010). The influx of new chemicals, including that of Irgarol 1051 and atrazine (Chesworth et al., 2004; Scarlett et al., 1999), leads to a loss in water quality through impairing water clarity and altering chemical composition while also loading further chemicals into the sediment (Alvarez, 2020). This increase in nutrients to the habitat inhibits the uptake of nutrients required for the growth of the seagrass therefore reducing rates of growth among seagrass beds (Apostolaki et al., 2011). Reports stated that increased eutrophication to Mediterranean habitats led to an increase in shoot mortality and a decrease in photosynthesis levels, limiting the ability of the habitat to recover (Marbà and Duarte, 2010; Ontoria et al., 2019)

The second most prominent threat reported within literature was aquaculture (12 out of 51 papers). Negative impacts of aquaculture were noted across Europe however research focuses primarily on the negative impacts of aquaculture surrounding the Mediterranean (Blake et al., 2014; Duarte, 2002; Ruiz et al., 2010). Seagrass beds which are exposed to cages used in finfish aquaculture are more susceptible to increased nitrogen levels within shoot tissues as a result of nutrient loading (Rountos et al., 2012). Increases in nutrient loading lead to a decrease in productivity of seagrass shoots (Waycott et al., 2009). Further research is needed to categorize optimum nitrogen levels for seagrass habitats due to the variability of contributing factors such as geographic location and species however current estimates put the optimum range nor nitrogen at approximately 8ppt for *Zostera marina* habitats found within UK waters (Vieira et al., 2022). Continued exposure to fish farms leads to a lack of biodiversity and the eventual loss of the seagrass habitat (Thomsen et al., 2020).

Climate change was one of the third most reported threats to European seagrass (10 out of 51 papers). By 2050, global temperatures are expected to rise by 2◦C (Rogelj et al., 2015). Within this time, increased temperatures are expected to inhibit growth rates and regeneration of *Posidonia oceanica* leading to a loss of 70-75% of its suitable habitat globally based on current conditions (Chefaoui et al., 2018). Global increases in sea temperatures and decreases in pH are leading to the breakdown of seedlings and reductions in photosynthetic rates required to allow for the growth of seagrass habitats (Hemminga and Duarte, 2000, Pereda-Briones et al., 2019). This prevents the regrowth of seagrass after its seasonal reductions over the winter period as well as the reduction or restoration and regrowth after other anthropogenic disturbances (Duarte, 2002, Hemminga and Duarte, 2000).

Coastal development was also the third most prominent threat to seagrass habitats within literature (Figure 5). Coastal development was shown to be most at threat surrounding UK and Scandinavian waters (Burke, 2004; Eriander et al., 2017). Increasing coastal developments increases shading of coastal areas leading to reduced coverage of *Zostera marina* but up to 64% within Swedish waters (Eriander et al., 2017). Development causes the fragmentation of habitats leading to decreased ecosystem function and impaired migration and consequently, reduced genetic diversity(Blake et al., 2014; Jones and Unsworth, 2016).

One of the least reported threats was habitat fragmentation which was only reported once and lacked the detailed documentation that other factors such as eutrophication and aquaculture received.

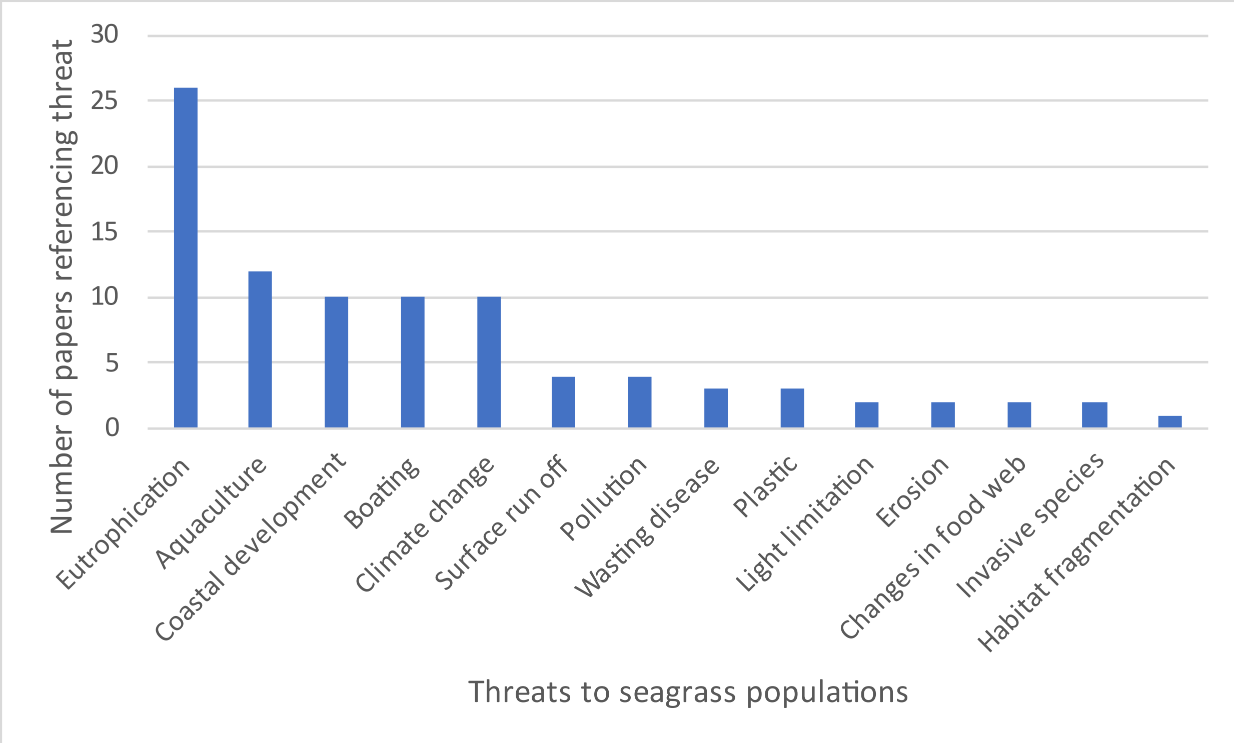

Figure 5. Frequency of which threats to seagrass are reported in literature found during November, 2022 using Web of Science.

2.3.2 Policies protecting seagrass habitats

All policies within Europe that aim to protect seagrass habitats can be seen Table 2. The IUCN Red List assesses global populations of seagrass species to determine their risk of extinction. At the time of researching, *Zostera* species were noted as 'least concern but declining, however, this is due to be reviewed and so may change upon review.

The Bern Convention (1982) is a European policy which prohibits the intentional picking, cutting, collecting, or uprooting of *Zostera marina.* The policy is often used as the basis of the creation of national policies. The Habitats Directive (1992) is another European policy which aims to protect both *Zostera noltii* and *Zostera marina.* Both species receive protection under Annex II meaning that efforts must be made to manage their ecological needs.

The UK provides protection to seagrass habitats through the Marine and Coastal Access Act (2009). Part 5 documents that MPA designation must include protected features and document how they will be protected. In addition to this, the Ramsar Convention of Wetlands (1971) highlights the importance of Ramsar sites within the UK, many of which incorporate seagrass beds. The UK Biodiversity Action Plan (1992) identifies seagrass beds as experiencing extreme threat and loss. Such identification ensures that the habitat is identified as a priority to UK ministers to ensure its protection by ensuring regular reviews of protective measures and their efficacy in terms of current seagrass density.

Countries within the UK have their own policies aiming to protect seagrass habitats. Scotland has introduced the Scotland National Marine Plan which identifies seagrass beds as a Priority Marine Feature. In doing so, it ensures that efforts are made to educate the public on the habitat and to ensure that the habitat is provided with adequate protection through the designation of MPAs. The Environment (Wales) Act (2016) also identifies seagrass as a priority habitat ensuring that Welsh ministers make efforts to protect and restore the habitat. Northern Ireland Habitat Action Plan (2003) ensures the introduction of schemes to reduce the impact of fishing and other coastal activities on seagrass habitats within Northern Irish waters.

Table 2. Policies within Europe that act to protect and restore seagrass beds and Zostera as a species.

	Convention	Sub-section	Commitments to Contracting Parties	Protection of Seagrass	Date Created
International	IUCN Red List		IUCN Red List uses global population data to document whether a species is at risk of extinction. Species are ranked on a scale from 'least concern' to 'extinct'.	At time of research, *Zostera* species were labelled 'least concern but declining'. It was also documented that this is due to be reviewed.	1964
Europe	Bern Convention	Appendix I	Bern convention aims to conserve and protect species and habitats within Europe. This legislation is also the driver of many more localised policies such as The Wildlife and Conservation act.	Under the act, the deliberate picking, cutting, collecting, or uprooting of *Zostera marina* is prevented. Further to this, the convention has influenced many acts within the UK and has led to the implementation of many Marine Protected Areas (MPA).	1982
	Habitats Directive	Annex II	Habitats directive aims to protect native animal and plant species within Europe that are rare or threatened along with 200 different habitat types.	*Z. noltii* and *Z. marina* are protected under annex II stating that efforts must be made to manage the ecological needs of the species.	1992
	Marine Strategy Framework Directive (MSFD)	Annex I	MSFD aims to protect benthic habitats, prevent deterioration and where possible restore the habitats.	In accordance to MSFD, Human influenced pollution such as eutrophication should be minimised and benthic habitats should be protected and damage minimised. It is also stated that there should be no introduction of	2008

	Convention	Sub-section	Commitments to Contracting Parties	Protection of Seagrass	Date Created
				non-indigenous species to these benthic habitats.	
	OSPAR List of Threatened and/or Declining Habitats	OSPAR Regions I, II, III, IV	OSPAR identifies habitats that are at risk, commonly from anthropogenic impacts, or are rapidly declining.	Identifies *Zostera* beds as habitats that are at risk and are declining at rapid rates	2015
	Water Framework Directive (WFD)	Annex IV, V, VIII	WFD aims to protect aquatic ecosystems and prevent degradation of the areas as well as promoting sustainable water usage and reducing water pollution.	Annex IV identifies *Zostera* beds as habitats of higher sensitivity meaning these beds must be considered when development plans are made in the area. Annex V refers to the need to protect the benthic environments in which *Zostera* beds inhabit. Annex VIII indicates the impact of eutrophication of marine species.	2000
Mediterranean	Barcelona Convention		Barcelona Convention focuses on the reduction of harm to coastal and marine environments from pollution and other anthropogenic factors whilst also ensuring a positive quality of life for individuals in the area.	An action plan was introduced to discuss how in which seagrass meadows should be protected. Threats were highlighted which matched that of the findings displayed in the threats graph. The convention also states the need for Special Protected Areas (SPA) to be introduced where in which there would be no deliberate picking or disruption of seagrass meadows	1978

	Convention	Sub-section	Commitments to Contracting Parties	Protection of Seagrass	Date Created
				whist also highlighting the need for research and education.	
UK	Marine and Coastal Access Act	Part 5	Marine and coastal access act sets out new systems of marine management, including new fishing policies and further designations of areas for protection.	Part 5 of the act sets out further designation of MPAs. It documents that orders for MPAs must include the protected features and how the area intends to protect these features. Many MPAs within the UK encompass seagrass beds with a fraction noting them as protected features, therefore in need of protection.	2009
	Ramsar Convention on Wetlands		Sites of specific interest are identified, and Marine Protected Areas (MPAs) are implemented. These ensure that habitats and species are maintained and protected from degradation.	Many seagrass beds are protected within MPAs within the UK. These take the form of Sites of Special Scientific Interest (SSSI), Special Protected Areas (SPA), Special Areas of Conservation (SAC) and Marine Conservation Zones (MCZ)	1971
	The Wildlife and Countryside Act	Part 2	Wildlife and countryside act implements Bern Convention and Birds Directive in the UK. It sets out protection for wildlife as well as public access and rights of way. The act enforces the implementation of several SSSI sites throughout the UK.	MPAs have been added to the UK in which *Zostera* beds are named as a listed feature (see UK MPA table). This ensures they are protected in the legislation and protection provided by the area.	1981

	Convention	Sub-section	Commitments to Contracting Parties	Protection of Seagrass	Date Created
	UK Biodiversity Action Plan (UK BAP)	Volume 5	UK BAP lists habitats and species in the UK that are seen to be under the most threat and experiencing significant loss. These habitats and species are then identified and should be a focus when implementing conservation projects within the UK.	Seagrass beds have been identified as a habitat experiencing extreme threat and loss. This therefore identifies than as a priority to UK ministers as in need of protection and should therefore be considered within policies and restoration attempts.	1992
England	Natural Environment and Rural Communities Act		The act sets out legislation to protect terrestrial and marine habitats. The act also introduces the importance SSSIs within England.	Seagrass habitats are noted as a Habitat of Principle Importance (HPI). This emphasises the importance of the habitats and prioritises its protection and restoration.	2006
Scotland	Scotland National Marine Plan		Scotland identified 81 species and habitats within their waters to be identified as Priority Marine Features (PMF). These features then become the focus of conservation and protection within Scottish policies. Efforts are made to educate individuals about the features and restoration projects are implemented where possible.	Seagrass beds have been named as a Priority Marine Feature (PMF) and so they have been identified by Scottish government as a habitat in need of protection. Efforts have therefore been made to educate the public on such habitats, further designation and MPAs have been introduced naming seagrass as protected features and restoration projects have been undertaken to restore these habitats.	2014

	Convention	Sub-section	Commitments to Contracting Parties	Protection of Seagrass	Date Created
Wales	Environment (Wales) Act	Section 7	Environment act sets out the need to 'sustainable management of natural resources.' Section 7 identifies species and habitats within Wales which should be identified as priority meaning that ministers must take steps to ensure their protection.	Seagrass beds are identified as a priority habitat meaning that Welsh ministers must make efforts to protect and restore these habitats.	2016
Northern Ireland	Northern Ireland Habitat Action Plan	Seagrass Beds	NIHAP ensures the conservation and management of habitats that are seen to be at risk within Northern Ireland.	The action plan further identifies threats to seagrass that are occurring within the waters of Northern Ireland. It also documents the need for MPAs in the form of Areas of Special Scientific interest (ASSI), candidate Special Areas of Conservation (cSAC), Special Protection Areas (SPA), Ramsar sites and National Nature Reserves (NNR). It also ensures the introduction of schemes to ensure fishing and similar activities do not negatively affect seagrass beds. Equally, it notes the need to record, and report damaged and degrading seagrass beds.	2003

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2.3.3 Spatial protection and Marine Protected Areas (MPAs)

2.3.1 Types of Marine Protected Areas (MPAs)

The map shown in Figure 6 displays the size, shape, and location of MPAs within the UK and Ireland and known seagrass beds. There were seven types of MPAs identified as containing seagrass habitats throughout the UK. Table 3 documents each MPA within the UK which is known to contain a seagrass bed. The table also documents the type of MPA in place, the size of the MPA, the level of protection the area provides and whether seagrass is listed as a protected feature.

The most common type of MPA protecting seagrass beds is Special Areas of Conservation (SAC) designated under the Habitats Directive. Within UK waters there were 38 SACs identified containing known seagrass beds equating to 20,807 km^2 of protection. Special Protected Areas (SPA) designated under the Birds Directive are another type of MPA protecting seagrass. A total of 25 SPAs were documented spanning 6949 km^2 of marine habitat. Marine Conservation Zones (MCZ) throughout England and Wales were less common, however, 12 protected seagrass beds totaling 1598 km^2 of protection. Many types of MPA are more regionalized such as Sites of Special Scientific Interest (SSSI) which can be found in the marine environment. There were 13 SSSIs across the UK containing seagrass beds within their geographical boundaries. The smaller nature of these areas, however, means that the total area protected is only 186 km^2. In addition to this, Scotland has designated 17 areas known as Nature Conservation Marine Protected Areas (NCMPA). Of these areas, only 1 MPA named seagrass as a listed feature.

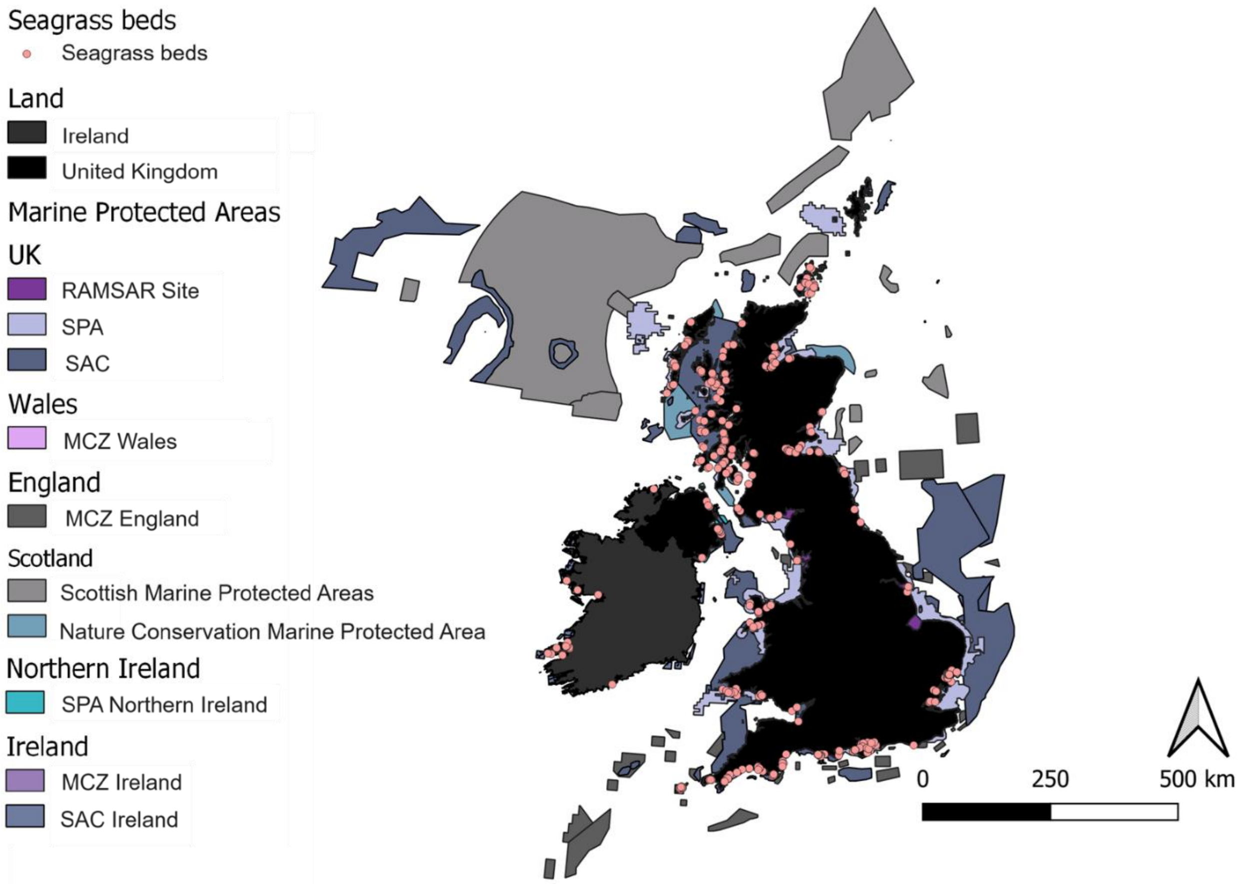

Figure 6: Marine Protected Areas (MPAs) in the UK and Ireland and known seagrass beds.

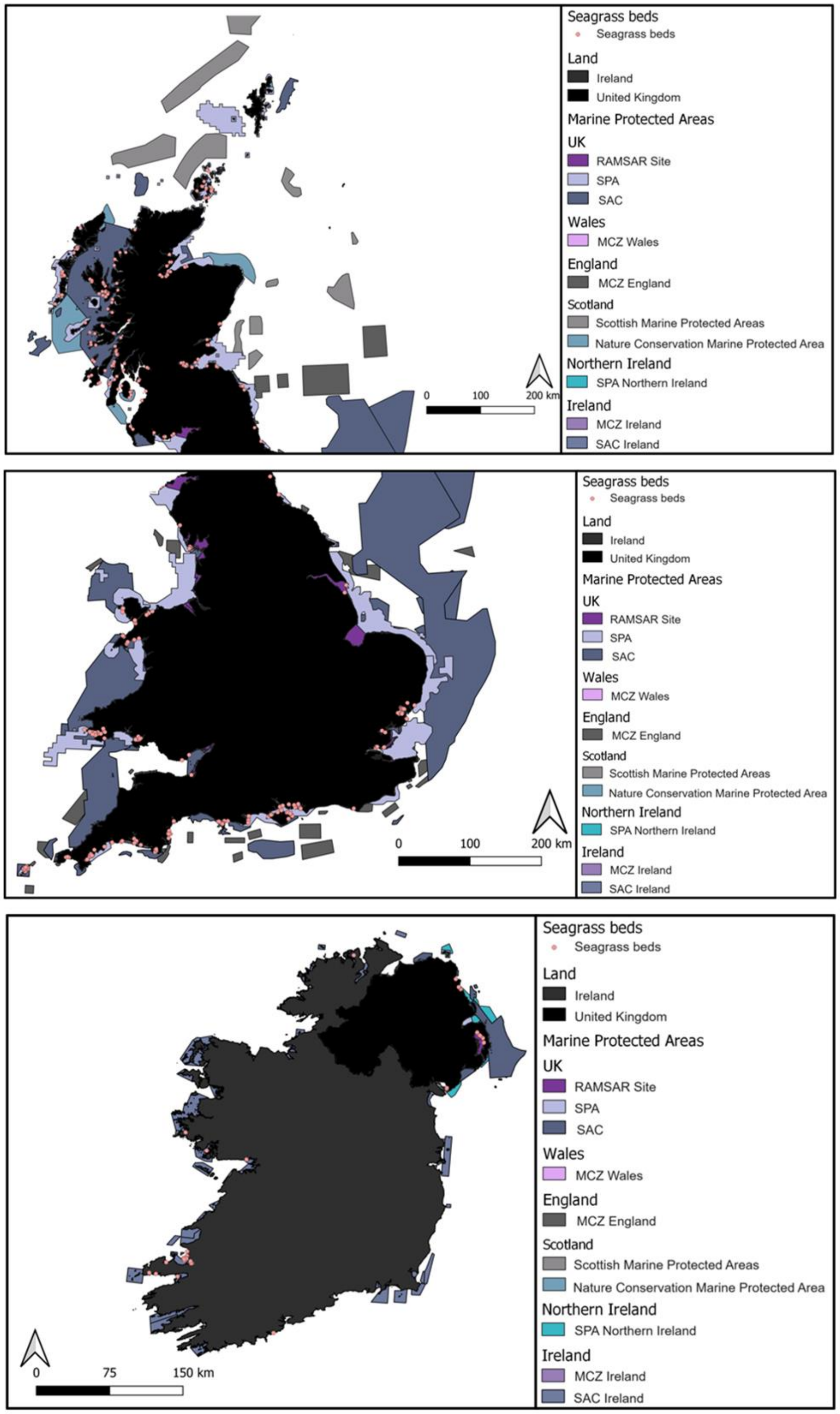

Figure 7: Maps showing seagrass beds and MPAs within the UK and Ireland. Top image shows coastal MPAs and seagrass beds within Scottish waters. Middle image shows coastal MPAs and seagrass beds within England and Wales. Bottom image shows coastal MPAs and seagrass beds within Northern Ireland and Republic of Ireland.

2.3.2 Listed Features

The type of protection an MPA provides is very much dependent on the species it aims to protect. Each MPA documents a list of species which the area aims to protect. Both SPAs and SACs will undergo a Habitat Regulation Assessment (HRA) as a part of their proposal. This assessment will set regulations which users of the area must abide by whilst also considering cultural and commercial activities that may be affected. While areas which protect other species will provide protection to seagrass beds, areas which aim to specifically protect seagrass habitats and *Zostera sp*. will provide greater protection. Within the UK, 10 MPAs specified seagrass habitats or *Zostera marina* within their listed features.

Table 3: Table listing all MPAs containing seagrass beds within the British Isles. Table shows the conservation site type and size of conservation site. Levels of protection from fishing are indicated and the presence or absence of a management plan depicting how the area will be protected is noted. Where seagrass is a listed feature within the protected area this is noted.

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
England	Berwick to St Mary's	MCZ	634	Less Protected / Unknown	No	
	Blackwater, Crouch, Roach and Colne Estuaries	MCZ	284	Designated & Unimplemented	No	
	Chesil Beach and Stennis Ledges	MCZ	38	Less Protected / Unknown	Yes	
	Chichester and Langstone harbours	SPA	58	Less Protected / Unknown	No	
	Dengie	SPA	31	Less Protected / Unknown	No	

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
	Essex Estuaries	SAC	461	Less Protected / Unknown	No	
	Exe Estuary	SPA	24	Less Protected / Unknown	No	
	Fal and Helford	SAC	64	Less Protected / Unknown	No	
	Falmouth Bay to St Austell Bay	SPA	259	Less Protected / Unknown	No	Yes
	Helford Estuary	MCZ	6	Less Protected / Unknown	No	
	Lindisfarne	SPA	32	Less Protected / Unknown	No	
	Lyme Bay and Torbay	SAC	312	Less Protected / Unknown	No	

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
	Morecambe Bay	SPA	669	Less Protected / Unknown	No	
	Morecambe Bay	SAC	615	Less Protected / Unknown	No	
	Mounts Bay	MCZ	12	Less Protected / Unknown	No	Yes
	North Norfolk Coast	SPA	79	Less Protected / Unknown	No	
	Plymouth Sound and Estuaries	SAC	64	Less Protected / Unknown	No	
	Portsmouth harbour	SPA	12	Less Protected / Unknown	No	
	Purbeck Coast	MCZ	282	Less Protected / Unknown	No	

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
	Solent and dorset coast	SPA	890	Less Protected / Unknown	No	
	Solent Maritime	SAC	112	Less Protected / Unknown	No	
	Stour and Orwell Estuaries	SPA	37	Less Protected / Unknown	No	
	Studland Bay	MCZ	4	Less Protected / Unknown	No	Yes
	Studland to Portland	SAC	332	Less Protected / Unknown	No	
	Swale Estuaries	MCZ	51	Less Protected / Unknown	No	
	Tamar Estuaries Complex	SPA	19	Less Protected / Unknown	No	

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
	The Needles	MCZ	11	Less Protected / Unknown	No	Yes
	Torbay	MCZ	20	Less Protected / Unknown	No	Yes
Wales	Anglesey Terns	SPA	1019	Less Protected / Unknown	Yes	
	Burry Inlet	SPA	67	Less Protected / Unknown	Yes	
	Carmarthen Bay	SAC	661	Less Protected / Unknown	Yes	
	Liverpool Bay	SPA	2528	Less Protected / Unknown	Yes	
	Lleyn peninsula and the sarnau	SAC	1460	Less Protected / Unknown	Yes	

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
	Menai Strait and Conwy Bay	SAC	265	Less Protected / Unknown	Yes	
	North Anglesey Marine	SAC	3249	Less Protected / Unknown	No	
	Northern Cardigan Bay	SPA	823	Less Protected / Unknown	No, but in preparation	
	Pembrokeshire Marine	SAC	1380	Less Protected / Unknown	Yes	
	Severn Estuary	SAC	737	Less Protected / Unknown	Yes	
	Traeth Lafan	SPA	27	Less Protected / Unknown	Yes	
	West Wales Marine	SAC	7376	Less Protected / Unknown	No	

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
N. Ireland	Carlingford Lough	SPA	8	Less Protected / Unknown	No	
	Killough Bay	SPA	1	Less Protected / Unknown	No	
	Larne Lough	SPA	4	Less Protected / Unknown	No	
	Lough Foyle	SPA	22	Less Protected / Unknown	No	
	Murlough	SPA	119	Less Protected / Unknown	No	
	Rathlin	MCZ	91	Less Protected / Unknown	No	
	Rathlin Island	SPA	33	Less Protected / Unknown	No	

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
	Rathlin Island	SAC	33	Less Protected / Unknown	No	
	Skerries and Causeway	SAC	109	Less Protected / Unknown	No, but in preparation	
	Strangford Lough	MCZ	165	Less Protected / Unknown	No	
	Strangford Lough	SPA	156	Less Protected / Unknown	No	
	Strangford Lough	SAC	154	Less Protected / Unknown	No	
Republic. Ireland	Ballyhoorisky Point to Fanad Head	SAC	13	Designated & Unimplemented	No	

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
	Broadhaven Bay	SAC	90	Designated & Unimplemented	No	
	Broadmeadow / Swords Estuary	SPA	8	Less Protected / Unknown	No	
	Bunduff Lough	SAC	44	Designated & Unimplemented	No	
	Cummeen Strand	SAC	49	Less Protected / Unknown	No	
	Horn Head to Fanad Head	SPA	24	Less Protected / Unknown	No	
	Kenmare River	SAC	433	Less Protected / Unknown	No	
	Lower River Shannon	SAC	683	Designated & Unimplemented	No	

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
	Malahide Estuary	SAC	8	Less Protected / Unknown	No	
	Mullet / Blacksod Bay Complex	SAC	141	Less Protected / Unknown	No	
	Mulroy Bay	SAC	32	Less Protected / Unknown	No	
	Roaringwater Bay and Islands	SAC	143	Less Protected / Unknown	No	
	Rutland Island and Sound	SAC	39	Designated & Unimplemented	No	
	Tralee Bay and Magharees Peninsula	SAC	116	Less Protected / Unknown	No	

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
	Valencia Harbour / Portmagee Channel	SAC	27	Designated & Unimplemented	No	
	West Connacht Coast	SAC	659	Designated & Unimplemented	No	
Scotland	Cromarty Firth	SSSi	36	Less Protected / Unknown	No	
	Dornoch Firth	SSSi	20	Less Protected / Unknown	No	
	Firth of Forth	SSSi	61	Less Protected / Unknown	No, but in preparation	
	Firth of Tay	SAC	151	Less Protected / Unknown	No	

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
	Loch Bee	SSSi	8	Less Protected / Unknown	No	
	Loch Fleet	SSSi	7	Less Protected / Unknown	No	
	Loch Nam Madadh	SSSi	3	Less Protected / Unknown	No	
	Loch Nam Madadh	SAC	23	Less Protected / Unknown	No	
	Loch Obisary	SSSi	4	Less Protected / Unknown	No	
	Loch Roag	SAC	4	Less Protected / Unknown	No	
	Loch Siadar	SSSi	1	Less Protected / Unknown	No	

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
	Lochs of Harray	SSSi	18	Less Protected / Unknown	No	
	Longman and Castle Stuart Bay	SSSi	4	Less Protected / Unknown	No	
	Moine Mhor	SAC	12	Less Protected / Unknown	No	
	Montrose Basin	SSSi	10	Less Protected / Unknown	No	
	Obain Loch Euphoirt	SAC	3	Less Protected / Unknown	No	
	Sound of Arisaig	SAC	46	Less Protected / Unknown	No	
	Sound of Barra	SAC	125	Less Protected / Unknown	No	

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
	South Arran	NCMPA	280	Less Protected / Unknown	No	Yes
	Tayport	SSSi	13	Less Protected / Unknown	No	
	Tob Valasay	SSSi	1	Less Protected / Unknown	No	
Isle of Man	Ramsey Bay	MNR	97	Less Protected / Unknown	No	Yes
	Laxey Bay	MNR	4	Less Protected / Unknown	No	Yes
	Douglas Bay	MNR	5	Less Protected / Unknown	No	
	Langness	MNR	89	Less Protected / Unknown	No	Yes

Country	Marine Protected Area	Conservation type	Area Size (Km2)	Fishing Protection level	Management Plan	Seagrass as a listed feature
	Baie ny Carrickey	MNR	11	Less Protected / Unknown	No	Yes
Jersey	Jersey Coast	JMPA	87	Less Protected / Unknown	No	

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2.3.3 Levels of Protection

Documents detailing the protection received within each MPA were retrieved in order to assess levels of protection within each MPA. These standard data forms provided records of any management plans attached to the areas. The plans documented in detail identified threats to listed features along with their ideal conditions were then used to ensure regulations matched the threats. Of the 95 MPAs under analysis, ten had accessible management plans and three had plans in preparation.

Of the 1890 seagrass beds within the UK and Ireland, 93.4% of seagrass beds are found within the geographical boundaries of an MPA. When this figure is reduced to include only MPAs which list seagrass as a protected feature the figure is reduced to just 5%.

2.3.4 Policies addressing threats

Chemical threats such as eutrophication, surface run off and pollution receive protection primarily at an international level. The Water Framework Directive (2000), Marine Strategy Framework Directive (2008) and EU Zero Pollution Action Plan all aim to reduce the use of chemical pesticides and herbicides which lead to increased nutrient loading. The Habitat Directive (1992) and Barcelona Convention (1978) aim to assess and address nutrient input from land-based sources.

Threats such as boating, fishing and coastal development were categorized as physical threats due to their direct impact on effected habitats. International policies such as the Bern (1982) and Barcelona Convention (1978) prohibit the intentional disturbance of seagrass beds preventing coastal construction and the use of benthic fishing within the area. Policies focusing on deliberate behaviours allow for misinterpretation. While the Bern Convention (1982) prevents disturbance due to coastal development and trawling, it could be argued that damage from boating is not intentional or deliberate. If policies focus on the intention behind the actions, it can be easy to forget about the consequences of unintentional actions that also deal a significant amount of damage.

While climate change policies do not directly reference seagrass as a habitat or a species, the threat was the third most prominent within literature due to the effects of turbid conditions, ocean acidification and temperature rise. Global policies such as the International Panel on Climate Change (1988) and Paris Climate Agreement (2016) as well as UK policies such as the Climate Change Act (2008) all aim to reduce atmospheric carbon and work towards carbon neutrality. By reducing global carbon emissions and atmospheric carbon levels countries can prevent the further acidification of oceans as well as extreme storms and weather events leading to turbid, sediment disturbing conditions.

Table 4 Comparison of categorized known threats to seagrass habitats within the UK and their associated protective policies.

Category	Examples of included threats	Location	Mitigating policies	Year	Method of protection
Chemical	Eutrophication, surface run off and pollution	Europe	EU Zero Pollution Action Plan	2021	Works towards the reduction of plastic litter at sea and the use of chemical pesticides by 50% by 2030.
			Marine Strategy Framework Directive	2008	Should work with Water Framework Directive to assess and ensure reduced levels of nutrient input.
			Water Framework Directive	2000	Should work with Marine Strategy Framework Directive to assess and ensure reduced levels of nutrient input.
		Mediterranean	Barcelona Convention	1978	Ensures the assessment of lagoons and coastal areas adjacent to river mouths. Efforts should be taken to reduce nutrient input from land based sources.
		United Kingdom	Habitat Directive	1992	Assessment of nutrient levels should take place when seagrass species are threatened and action should then be taken to resolve the threat.
Physical	Boating, fishing, and coastal development	Global	Bern Convention	1982	Prohibits the deliberate gathering, cutting, or uprooting of *Zostera marina*.
		Mediterranean	Barcelona Convention	1978	Prohibits the deliberate picking or disturbance of seagrass beds.

		United Kingdom	Marine and Coastal Access Act	2009	Restricts fishing and fishing methods. The degree to which they are restricted is dependent on the MPA.
		Northern Ireland	Northern Ireland Habitat Action Plan	2003	Requires damaged or degrading seagrass beds to be recorded and reported.
Climate change	Turbid conditions, ocean acidification and temperature increase	Global	International Panel on Climate Change	1988	Outlines the increase of greenhouse gasses in our atmosphere. Discusses the impact of increased atmospheric greenhouse gasses on the marine environment. Highlight the need to reduce this figure.
			Paris Climate Agreement	2016	Works towards carbon neutrality, decarbonisation of fishing fleet and aims to increase use of offshore energy by 2050.
		United Kingdom	Climate Change Act	2008	Aims to reduce greenhouse gas emissions by at least 100% of 1990 levels by 2050.
Biological	Invasive species and disease	Global	United Nations Convention on Biological Diversity	1992	Identifies invasive non-native species and provides an obligation to address the negative effects of invasive species.
		United Kingdom	Invasive Non-native Species Framework Strategy for Great Britain	2015	Ranks invasive species within UK waters on a scale from 0 to 4 dependent on the abundance of an invasive species within an area. Most abundant species are identified along with their entrance pathways and it is aimed to eradicate them from the area.

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2.4 Discussion

2.4.1 Threats to seagrass habitats

This review furthers work focusing on the prominence of reports of factors within the literature as opposed to the prominence of the threats in nature or the severity of threats. Few papers referenced multiple threats to seagrass habitats within their research therefore knowledge of cumulative threats and how these threats may interact together is limited. Previous research focusing primarily on Baltic regions has gathered and discussed literature quantifying effects of anthropogenic factors such as eutrophication and trawling (Krause-Jensen et al., 2021). Further to this, research has assessed the impacts of anthropogenic behaviors along the Australian coastline (Grech et al., 2011; Rees et al., 2023). Significant research has been completed throughout the Mediterranean coastline (Alvarez, 2020; Balestri et al., 2017; Marbà and Duarte, 2010) however such impacts have yet to be collated and compared to determine the most prominent threats to Mediterranean seagrass habitats.

While some threats such as eutrophication and climate change have been well researched. Such patterns were similarly observed within research along the coast of Australia, suggesting that threats such as eutrophication are threatening seagrass beds globally (Grech et al., 2011). Factors such as habitat fragmentation have received less attention. Habitat fragmentation can occur as a result of other factors such as coastal development and from boating impacts (Fahrig, 2003). Research has stated that seagrass beds which are subject to fragmentation are at an increased risk of being lost completely however little research has been done to evaluate the risks (Swadling et al., 2023).

Factors such as plastic pollution are rapidly increasing yet research has remained limited (Bonanno and Orlando-Bonaca, 2020). It is estimated that there is a total of 710 million tonnes of plastic polluting marine and terrestrial ecosystems, however, little research has been conducted to assess the threats of such pollutants on seagrass habitats (Lau et al., 2020). Effects of plastic pollution on seagrass habitats have lacked focus of research globally (Grech et al., 2011; Krause-Jensen et al., 2021; Rees et al., 2023). Further research is therefore needed globally in order to correctly understand the threats posed to seagrass habitats by plastic pollutants (Dahl et al., 2021).

2.4.2 Policies Protecting Seagrass Habitats

The most prominent policy type protecting seagrass in the UK relies on spatial protection. Spatial protection measures provide protection to habitats through the limitation of direct impacts such as fishing activities and boating (Glasby and West, 2018). The majority of MPAs documented were shown to minimize these activities but fail to prevent them completely (Jones, 2012). Spatial protection policies may not have the ability to mitigate against all threats, however, there is the potential for other policies to mitigate threats further.

Eutrophication is the most recorded threat to seagrass habitats. Policies such as the Bern Convention that focus on the implementation of MPAs are only able to protect and reduce threats that occur as a result of activities undertaken within the area (Sanchirico et al., 2003). Eutrophication occurs from activities undertaken on land and therefore this threat cannot be minimised by the introduction of MPAs. Conversely, research has stated that when localised threats are removed, the habitats may be more resilient to more widespread threats such as climate change (Stobart et al., 2009, Tarrach et al., 2022). The Water Framework Directive (2000) and Barcelona Convention (1978) also aim to reduce the impact of eutrophication. The Water Framework Directive (2000) aims to reduce the use of pesticides and fertilizers that lead to increased nutrient input. The Barcelona Convention aims to assess areas that are most at risk from nutrient input and build up. The removal of additional nutrient inputs such as herbicides and fertilizers will improve the health of the habitat over time however, the long-term goals set in place by current legislation fail to provide immediate protection for existing seagrass habitats (Carus et al., 2021).

Furthermore, many protective policies such as the Water Framework Directive (2000) are enforced by the European Union. After the UK disassociating itself from the European Union in recent years, environmental legislation has been retained until the present (discuss new Environment Bill and any possible repeals). The UK has similar policies previously enacted such as Marine and Coastal Access (2009) which provides similar cover to those provided by the EU. At the time of research current policies set in place by the EU acted as advisories to UK policies however some legislations remained to be updated in order to function independently form the EU.

2.4.3 Spatial Protection of Seagrass Habitats

There are five identifiable MPAs related to the protection of seagrass habitats, SSSIs, SACs, SPAs, MCZs, NCMPAs. These areas are all influenced by the same policies however, each type of area is regulated by a different governing body (Jobstvogt et al., 2014). This leads to discrepancies within the regulations they enforce, for example SACs and SPAs in comparison to MCZs. SACs and SPAs receive their designation from Natura 2000. MCZs receive their designation from the Marine and Coastal Access Act (2009). The process of creating MCZs is much more collaborative with stake-holders and aims to take into account the socioeconomic factors of protection (Pieraccini, 2015).

Within UK waters, designated Special Sites of Scientific Interest (SSSIs) aim to prohibit the damage, disturbance, or destruction of these areas in any way that may harm the habitat or affect wildlife (Gallego et al., 2017). Due to the destructive nature of dredges and trawls (Jones, 1992), trawling and dredging is prohibited within these areas. Such legislation also states that it is an offence to inflict damage to the area through boating activities including anchoring and propeller scars, regardless of if the damage is intentional or as a result of reckless behaviour. Regulations regarding damage to seagrass are clear however, policies fail to protect species living within seagrass which assist in the survival of the habitat (Last, 1999).

Within the UK, most MPAs appear to fail to mitigate the most identified threats to seagrass habitats. MPAs within the UK only regulate activities within the water (Day et al., 2015). This prevents protection from major threats such as eutrophication and climate change. Equally, due to each MPA enforcing its own regulations, there is a lack of consistency between areas and so levels of protection between areas are unequal (Roberts, 2001).

Within the Republic of Ireland, each site contains its own legislation dependent on the species the site aims to protect. All investigated areas identified, noted the impact of increased levels of nitrogen to an area (Skeffington and Jeffrey, 1988, Treacy et al., 2008). These areas then aim to minimize the levels of eutrophication to an area in order to minimize harmful increases in nitrogen (Skeffington and Jeffrey, 1988). Within the South West of Ireland, the use of nitrogen based fertilizers were reduced due to increased levels of nitrates within coastal waters (Treacy et al., 2008). This is an issue not addressed by other MPAs within the UK however, these regulations lack monitoring, and the degree of regulations are unclear.

The use of bottom-contact equipment in commercial fishing has also been frequently identified as causing damage to benthic habitats due to sediment disturbance (Jones, 1992). For this reason, regulations have been enforced which limits the usage of mobile fishing gear within MPAs such as the Burry Inlet SPA and the Traeth Lafan SPA (Rilov et al., 2020). This forbids the use of nets which may become entangled in the seagrass habitat and trawling and dredging which can lead to damage to the benthic habitats (Unsworth et al., 2017).

MPA management plans are often used in order to determine needed legislation and to ensure the appropriate management and enforcement. Such plans contain detailed descriptions of habitats found within the MPA and addresses any possible threats they may face. Only ten of the MPAs found within the UK that were identified to contain seagrass beds had accessible management plans with a further three having plans that are in progress. Of the 95 MPAs containing seagrass within the UK, 82 identified MPAs without a management plan. With 86% of researched MPAs lacking a management plan, the question can be raised as to whether all of the threats that the area could face have been identified and accounted for when implementing legislation.

In addition to this, each MPA has identified species which are named listed features. These features become the priority when employing legislation in order to protect the area. Only six of the investigated MPAs noted seagrass habitats or *Zostera marina* as listed features equating to 586 km^2 of coverage as seen in Table 3. While seagrass may experience similar threats to other species within an MPA, it is unlikely that all the threats experienced by seagrass will be addressed in MPAs where seagrass is not noted as listed feature.

2.5 Conclusion

It is clear that the most common threats to seagrass in within European waters, as reported in literature, were eutrophication and surface run off, aquaculture and climate change. Research highlighted the negative impacts of nitrogen leaching into the habitat and disrupting growth. It must be questioned, however, whether such factors we most frequently reported due to the severity of their nature or due to the biases in research and their reporting as a result of external funding sources and current research focuses. Research had previously focused on the impacts of eutrophication and nutrient input (Duarte, 2002; Marbà et al., 2008) whereas focus has now moved to the effects of plastic and climate change as research interests have changed (Marbà et al., 2022). Further research is needed in order to further quantify the severity of each threat in order to confirm the accuracy of these conclusions.

It is clear that there are certain levels of spatial protection which protect a number of seagrass beds surrounding the UK. These policies prevent the damage and uprooting of benthic species due to trawling in the majority of MPAs. Policies such as The Bern Convention (1982) prohibit the intentional cutting or uprooting of seagrass species. Such policies create a grey area where damage to seagrass may have been unintentional but somewhat unavoidable such as in the case of bottom contact fishing gear. The most debates with regards to whether damage is intentional come from the use of boats within the area. Boating is often unprohibited by MPAs therefore leaving a gap in policy for damage to occur.

Many areas fail to protect seagrass habitats due to their aims to protect other species. With a high proportion of UK MPAs comprising of SPAs which focus on the protection of birds, it is unlikely that such areas will provide adequate protection to benthic species. While there may be overlap within their protection needs, many threats to seagrass have been overlooked.

Chapter 3: Investigating biodiversity within *Zostera noltii* beds across the UK.

3.1 Introduction

Global levels of marine biodiversity have faced a significant decline in previous decades (Chu, 2008, Blowes et al., 2019). Increases in climate change, pollution and overfishing have led to reductions in biodiversity and increased extinction risk to species (Newbold et al., 2015). Climate change has led to changes within marine ecosystems which species are unable to adapt to (Crespo et al., 2017). Increases in temperature have led to reductions in growth rates and seed production rates within seagrass and kelp habitats, limiting their ability to recover and grow (Schlenger et al., 2020, Pereda-Briones et al., 2019). Further to this, the warming of temperate water has allowed the habitats to become a habitat for species that could not previously survive there, creating alterations in the food web (Vergés et al., 2014).

Increases in eutrophication within coastal environments has led to an increase in above sediment biomass however, such increases limits light and nutrient availability to below ground root systems which provide stabilization of the sediment (Deegan et al., 2012). Sublittoral seagrass habitats are particularly at risk from the effects or boating as propeller scars and mooring within the habitat leads to the breakage of seagrass shoots which provide shelter and nutrients to fauna within the habitat (Pergent-Martini et al., 2022, Hansen et al., 2019).

Seagrass beds are home to a wide variety of species including molluscs, crustaceans, echinoderms, and fish (Swadling et al., 2023, Muzaki et al., 2019, Becker et al., 2012). Seagrass shoots provide the ideal holdfasts for seahorse species throughout temperate and subtropical waters (Scapin et al., 2018). Bivalves feed on detritus collected within the seagrass canopy which falls to the sediment (Dame, 2012, Meysick et al., 2022). *Lebensspuren* is a term used by deep sea biologists to refer to the visible life traces left behind by an ecosystem's inhabitants, such as shells and worm casts (Przeslawski et al., 2012). Intertidal seagrass habitats, particularly *Zostera noltii*, are emersed at low tide and so the analysis of *lebensspuren* may allow quantification of surface-active species that are situated below the sediment during low tide (Bell et al., 2013, Hemminga and Duarte, 2000).

Denser seagrass beds have the ability to support increased biodiversity and increased organism abundance (McCloskey and Unsworth, 2015). Increasing shoot density increases food availability for grazers such as *Littorina* and deposit feeders such as *Echinodermata* (Voltolina and Sacchi, 1990, Coulon and Jangoux, 1993, Teoh and Woo, 2021). Further to this, increased shoot density allows for the formation of a larger, denser canopy which traps detritus allowing it to sink to the sediment where it becomes food for surface deposit feeders (Tanaya et al., 2018, Dame, 2012).

This research aimed to investigate the diversity of *lebensspuren* within *Zostera noltii* within beds across the UK. This research aimed to investigate a potential relationship between seagrass density and biodiversity within UK waters.

3.2 Method

3.2.1 Site location

A total of three sites were sampled throughout the investigation: i) Exmouth Bay, England ii) Holyhead, Wales and iii) Sand Haile Flats, England. These sites were chosen in order to represent *Zostera noltii* beds throughout the UK coastline. Each bed chosen has a detailed historical record documenting the abundance of seagrass within the area (Project Seagrass, 2023).

The first site was located where the River Exe meets Exmouth Bay. Exmouth Bay is situated in the Southwest of the UK at the mouth of the Exe Estuary. The site was divided into three areas for sampling as shown in Figure 8.

Figure 8 Map produced in Google Earth showing the surveyed area at Exmouth Beach, Devon. Yellow lines show the location of transects within seagrass bed 1. Blue lines indicate transects sampled within sandy habitats. Red lines show transects within seagrass bed 2.

The second site was located at Holyhead, North Wales. The seagrass bed stretched 800 m along the Cymyran Strait (Figure 9). The area falls within the Anglesey Terns SPA which aims to protect rock pools and *Ruppia cirrhosa (Spiral Tasselweed).* The local Penrhos Coastal Park nature reserve adds additional protection through the addition of a barrier between local farmland and housing which may serve as threats to the seagrass habitats.

Figure 9 Map of Cymyran Strait sampling site produced in Google Earth. Green lines identify the locations of transects taken within the seagrass bed whilst orange lines represent the location of transects taken external to the seagrass habitat.

Sand Haile Flats was also used as a site for sampling (Figure 10). The site is located a short distance along the coast from the town of Cleethorpes and is situated within the Humber Estuary SPA and SAC. While seagrass assemblages are sparse, the area has previously been home to dense seagrass beds which have experienced its most severe deterioration since approximately 2016 (Project Seagrass, 2023).

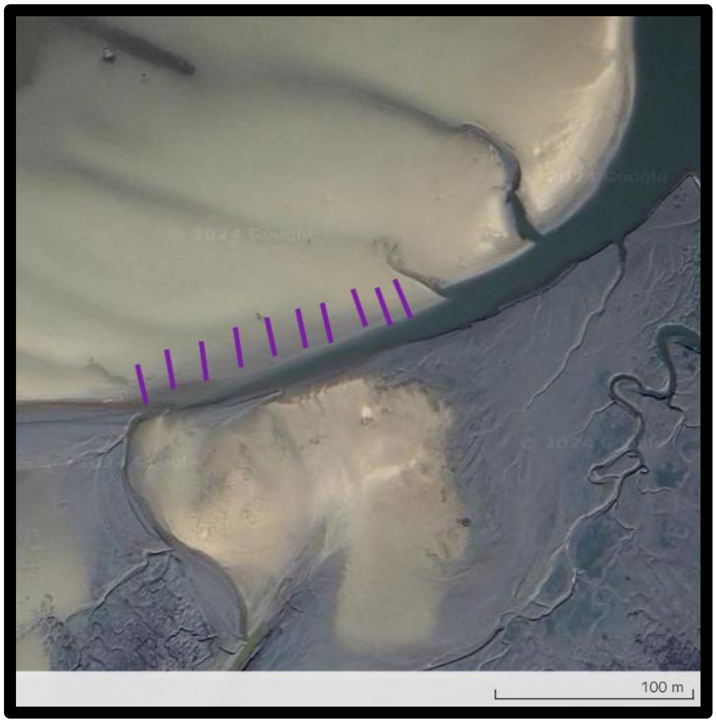

Figure 10 Map produced in Google Earth displaying the location of 10 transects taken from the Sand Haile Flats site.

3.2.2 Habitat identification

In order to identify whether a habitat was deemed a seagrass habitat or a unvegetated habitat, percentage cover of *Zostera noltii* was determined. The densest part of the habitat was identified by eye. Transects began in this location. To identify a boundary of seagrass beds I adopted a method that identified when the percentage cover of *Zostera noltii* fell below 5%, at which the habitat was labelled an unvegetated habitat. If seagrass appeared to become dense again after the sandy habitat, this seagrass bed was labelled as a separate seagrass bed, as seen at the Exmouth site in Figure 8.

3.2.3 Sampling method

Data was collected using photo quadrats. A total of 60 line transects that were 20 m in length were placed parallel to the shoreline. A total of 30 transects were taken at Exmouth with 10 taken from seagrass bed 1, 10 from seagrass bed 2 and a further 10 from the sandy sediment in between. A further 20 transects were taken from the Cymyran Strait site. Of the 20 transects taken, 10 were situated within the seagrass bed and a further 10 were taken from the sandy area next to the seagrass bed. Finally, 10 transects were taken from Sand Haile Flats.

Transects ran parallel to the shoreline and photos were taken at 5 m intervals along the transect allowing for five photos per transect. Each photo was taken from 1 m height to produce a photo quadrat of 0.25 m^2 resulting in 300 quadrats in total. GPS locations and details of each quadrat such as its position within the bed were noted. Coral Point Count (CPC) was used to establish the percentage cover of *Z. noltii* and green algae within each quadrat (Kohler and Gill, 2006). CPC identified 50 points along the transect which were later analyzed to assess the presence or absence of *Zostera noltii*.

Life traces known as *lebensspuren* were counted in each 0.25 m^2 quadrat and identified to determine species living on and within the sediment, while minimizing disturbance to the area. The number of each identified species per quadrat was recorded.

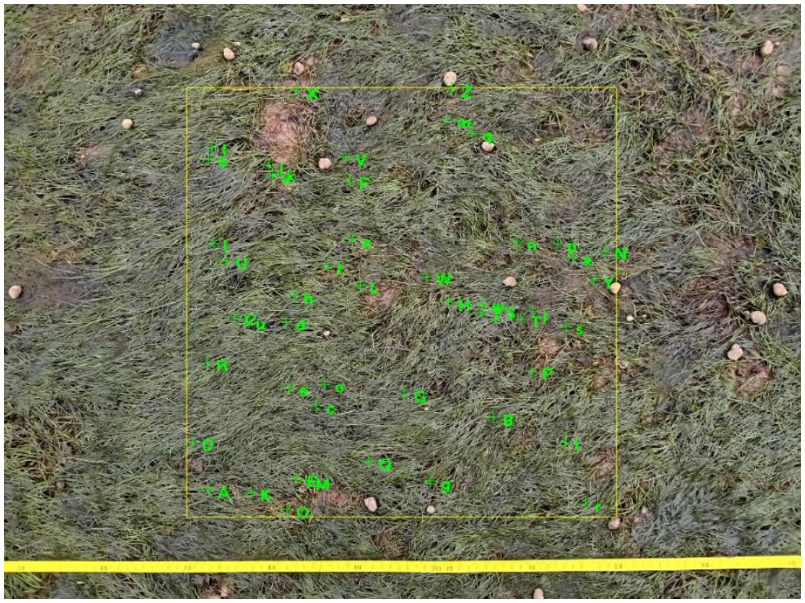

Figure 11: Example of a transect complete with points added by CPC in order to identify seagrass coverage.

3.2.4 Data analysis

3.2.4.1 Community analysis

Plymouth Routines In Multivariate Ecological Research (PRIMER) was used to investigate similarities and differences in community structure within sites due to the ease of replication for future studies and the reliability of the programme (Clarke and Gorley, 2006). A Bray-Curtis similarity index was produced to aid in the production of an MDS plot for each site. This allowed for a visual analysis of similarities between samples and between different areas of the site. An ANOSIM test was conducted to determine if there was a significant difference in similarity within and between sites using the frequency at which each species was found within each sample site. This was conducted to test the hypothesis that there was a similar ecological structure at each seagrass bed. *P* values were reported as a percentage and so any P value found to be less than 5% was deemed significant. A SIMPER test was used to determine similarity of species between habitats at each site and to what extent they are impacting the differences.

3.2.4.2 Measuring relationships

Scatter graphs were created to visually assess potential relationships between seagrass percentage cover and biodiversity. The Shannon diversity index (Equation 1) was used due to its common usage within ecological research allowing for more accurate comparisons to other research (Morris et al., 2014). The index was calculated using PRIMER. A correlation coefficient was then used to measure the significance of the relationship. It was hypothesized that the increase of anoxic conditions produced by algal mats may reduce diversity and so this method was then repeated to investigate the potential impact of algal mats on diversity. Correlation coefficients were also performed to identify the significance of the relationships between seagrass and each species identified through SIMPER tests that contributed most to differences within habitats.

$$H = -\sum[(p_i) \times \log(p_i)]$$

Equation 1: Equation used to calculate Shannon Diversity Index where p_i refers to the number of individuals per species divided by the number of species.

3.3 Results

3.3.1 Species identified

A total of five species were identified including two species of gastropods, two bivalve species and an Annelida species (Table 4). The most abundant species throughout all sites was *Cerastoderma edule* which were identified by shell fragments. The species was also the most abundant species in all but two sites. *Littorina littorea* is the most abundant species within the first seagrass bed within the Exmouth site and *Arenicola marina* was the most abundant species found within the sandy habitat within the Holyhead site.

Table 5 List of identified species found within sites including the mean abundance in terms of individuals per m^2 and standard deviation found at each site.

Site	Exmouth			Grimsby	Holyhead	
Habitat	Seagrass 1	Sand	Seagrass 2	Historic	Seagrass	Sand
Arenicola marina	0.9 ±1.55	0.24 ±0.62	1.12 ±1.76	1.94 ±2.70	0	4.94 ±4.78
Nucella lapillus	0.16 ±0.42	0	0	0	0	0
Littorina littorea	13.96 ±14.86	0	1.62 ±2.67	0	0.52 ±1.78	0.04 ±0.20
Cerastoderma edule	4.38 ±7.10	2.04 ±3.74	2.88 ±4.65	9.72 ±6.43	1.06 ±1.86	0.06 ±0.24
Mytilus edulis	0.06 ±0.24	0	0.02 ±0.14	0	0	0

3.3.2 Community structure within investigated sites

While investigating the Exmouth coastline, three distinct sites were noted. The first identified seagrass bed was subject to more physical disturbances such as mooring of boats and human trampling, whereas the second seagrass bed appeared to have higher rates of algal coverage. Evidence of this can be seen through the extensive boat presence seen within Figure 8. An MDS plot was created in order to investigate similarities in community structures between the two identified seagrass beds and the sand separating them (Figure 12). Assemblages displayed the most similarity within quadrats taken from the first seagrass bed. The second seagrass bed posed more differences both between samples and between seagrass beds. There were few similarities between samples within the sandy habitat. There was a significant difference in the *lebensspuren* identified within each habitat at Exmouth Beach (ANOSIM, Global R = 0.201, P = 0.1%). Pairwise comparison of the habitats revealed that all sites were significantly different (in all cases, P = 0.1%). A SIMPER test revealed that differences in community compositions were in relation to the abundance of organisms present as opposed to additional species found within habitats. *Littorina littorea* were found to be more abundant within the first seagrass bed whereas *Arenicola marina* casts were found to be more abundant within the second seagrass bed.

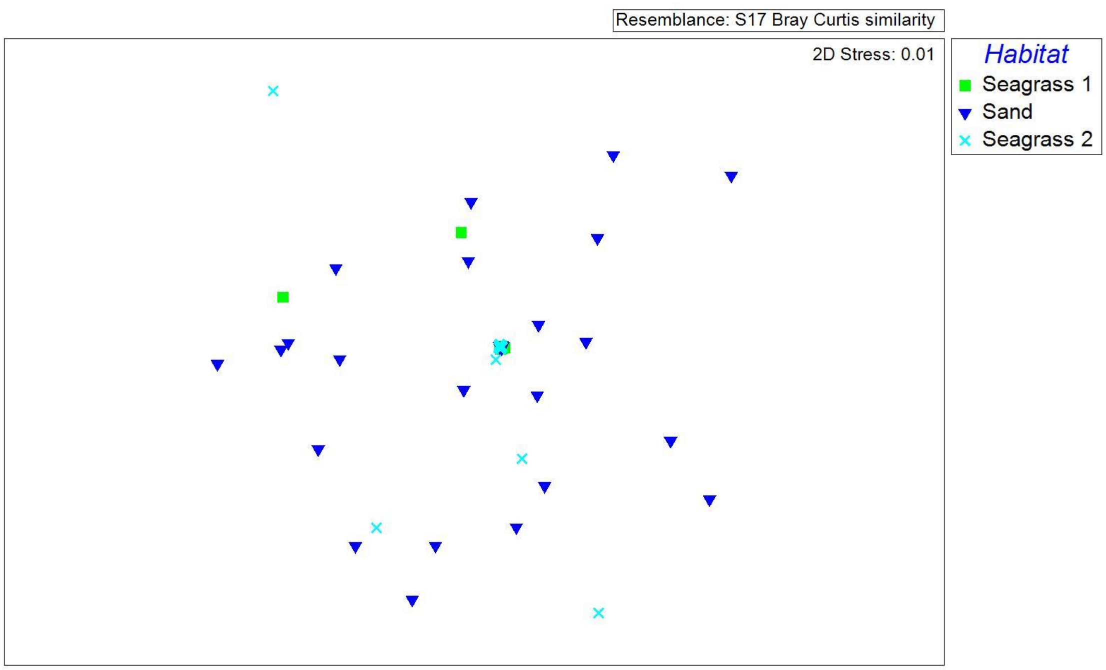

Figure 12 Results of MDS analysis on a Bray-Curtis similarity matrix for quadrats taken from Exmouth Beach.

The Holyhead site was divided into two distinct habitats, seagrass, and sandy habitats. An MDS plot was used in order to establish if there were any identifiable similarities within quadrats or between habitats (Figure 13). There were fewer similarities within quadrats within the Holyhead site. The MDS plot shows more similarities between quadrats within sandy habitats as opposed to seagrass habitats. An ANOSIM test was conducted to determine the similarities. There was a significant difference in identified lebensspuren within each habitat (ANOSIM, Global R = 0.432, P = 0.1%). A SIMPER test found *Cerastoderma edule* and *Arenicola marina* to be contributing the most to these differences with *Arenicola marina* predominantly being found within sandy habitats and *Cerastoderma edule* predominantly being found within the seagrass bed.

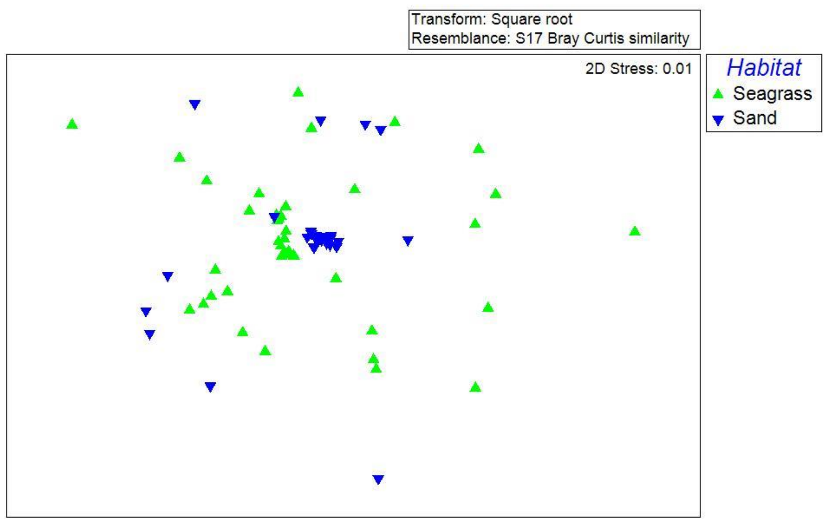

Figure 13 The results of MDS analysis on a Bray-Curtis similarity matrix for Cymryan Strait near Holyhead.

A final MDS plot was created to investigate similarities between seagrass all three investigated seagrass beds (Figure 14). There was a significant difference between the lebensspuren located within each *Z. noltii* bed (ANOSIM, Global R = 0.206, P-value = 0.1%). Pairwise comparison of seagrass beds revealed that all sites were significantly different. A SIMPER test discovered that abundance of *Littorina littorea* and *Cerastoderma edule* had the most significant impact on these differences.

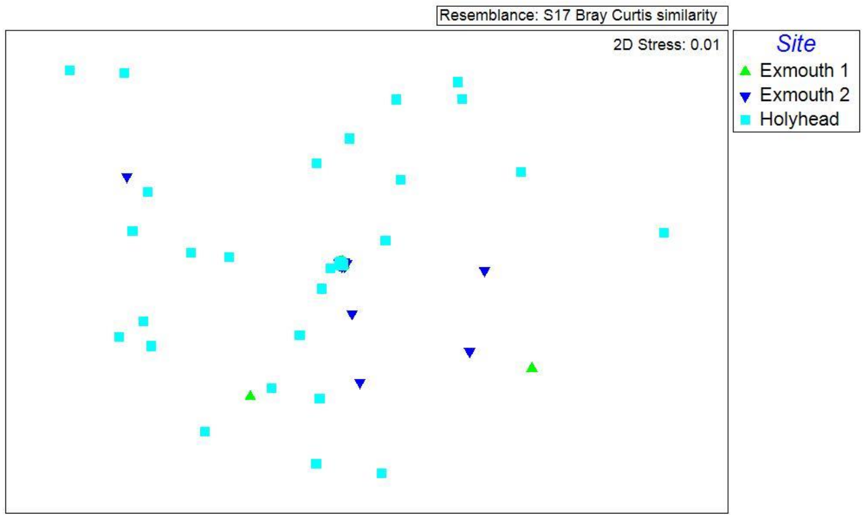

Figure 14 Results of MDS analysis on a Bray-Curtis similarity matrix for seagrass beds located in Exmouth and Holyhead.

3.3.2 Measuring relationships

Scatter plots were created in order to assess the relationship between seagrass and Shannon diversity (Figure 15). A correlation coefficient was conducted which returned a roh 0.22 correlation with p-value < 0.001, showing a significant positive but low regression. This was repeated to assess the relationship between percentage cover of *Z. noltii* and the total number of identified *lebensspuren*.

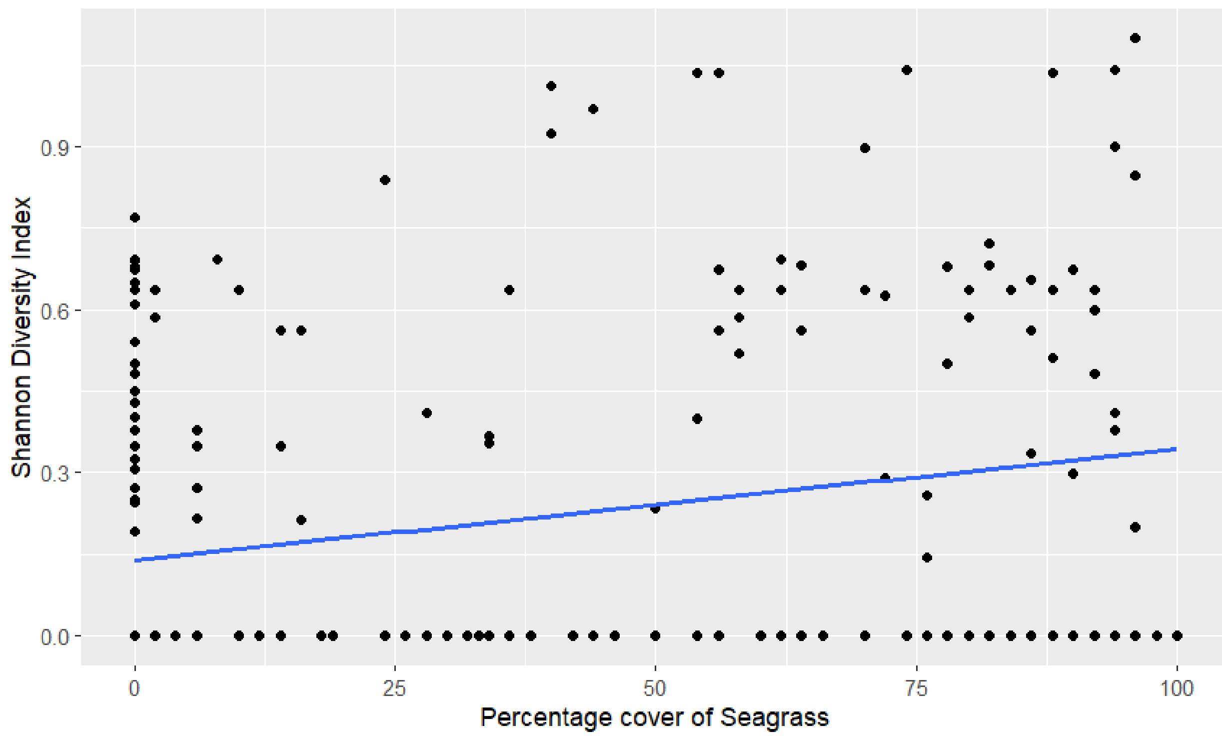

Figure 15 Scatter graph showing a positive regression between percentage cover of Z. noltii and Shannon diversity using a linear model.

Algal mats were quite prominent within the sampled sites with some areas being affected more than others. The question was then raised as to whether algal cover was impacting the organisms found within the habitats. A scatter plot was made to illustrate the relationship between both percentage cover of algal mats and Shannon diversity and also percentage cover of algal mat and total lebensspuren identified (Figure 16). A correlation of -0.05 with a p-value of 0.3516 was identified showing no significant relationship between percentage cover of algal mats and Shannon diversity. A correlation of -0.21 with a p-value of less than 0.001 showing a significant negative relationship between percentage cover of algal mats and the total number of lebensspuren identified.

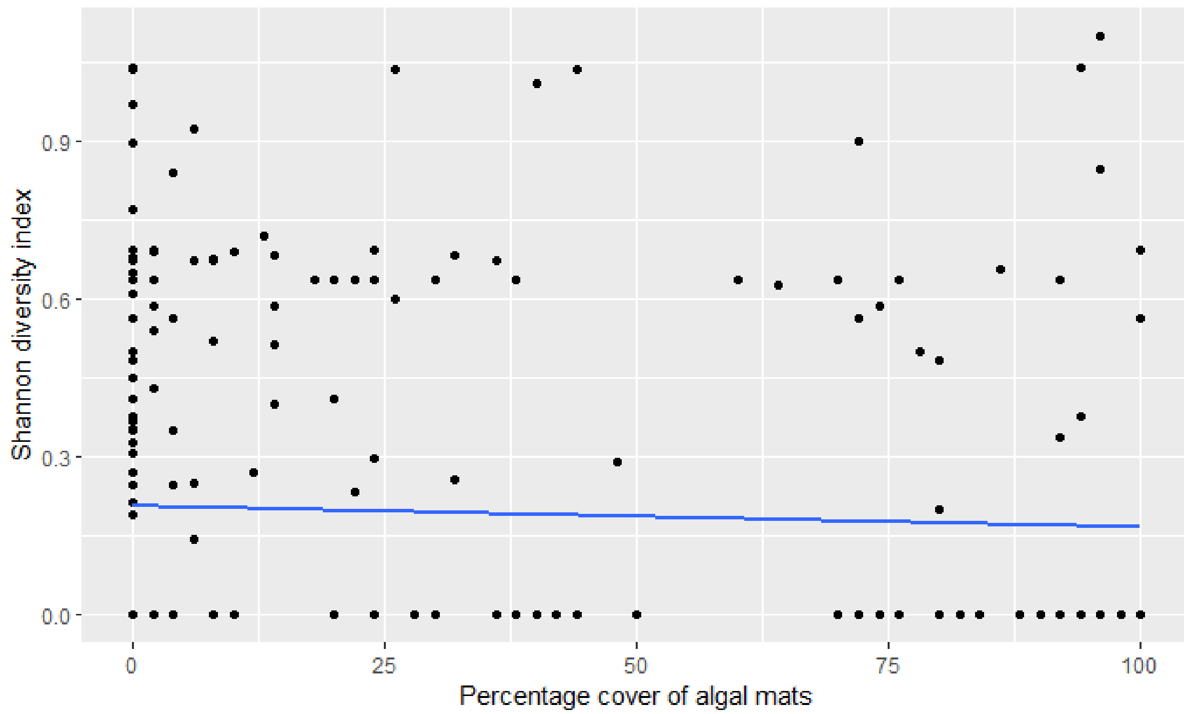

Figure 16 Scatter plot shows an insignificant negative regression between percentage cover of algal mats and Shannon diversity using a linear model.

3.3.3 Relationships between *Zostera noltii* and burrowing organisms

Upon identifying the positive relationship between percentage cover of *Z. noltii* and Shannon diversity, each species was then compared to the percentage cover of *Z. noltii*. A scatter plot was formed to assess a possible relationship between percentage cover of *Z. noltii* and *Arenicola marina* abundance. A correlation coefficient reported a correlation of -0.26 with a p value of less than 0.001 suggesting a significant negative but low relationship between the percentage cover of *Z. noltii* and *Arenicola marina* abundance.

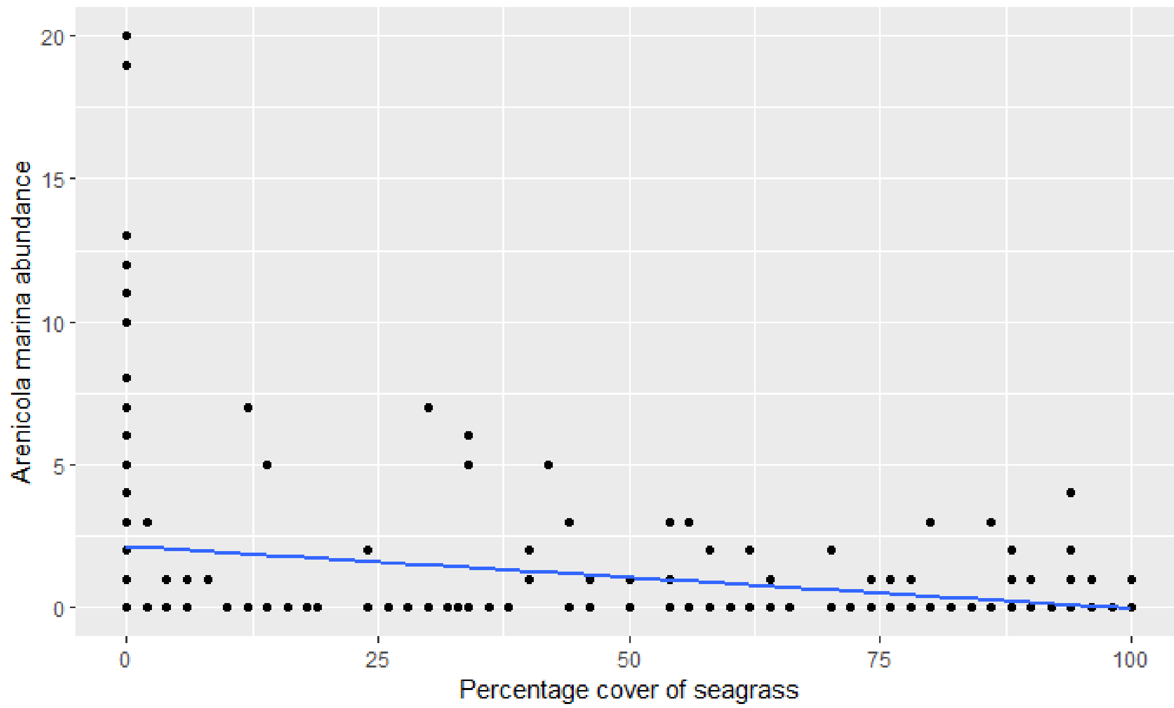

Figure 17 Scatter plot illustrating the negative relationship between Zostera noltii and Arenicola marina.

Littorina littorea was also identified as having a significant influence on the community structure within surveyed habitats. A scatter plot was formed to assess a possible relationship and a correlation coefficient was carried out. A correlation of 0.29 was reported with a P value of less than 0.001 suggesting a significant positive relationship between percentage cover of *Z. noltii* and *Littorina littorea*.

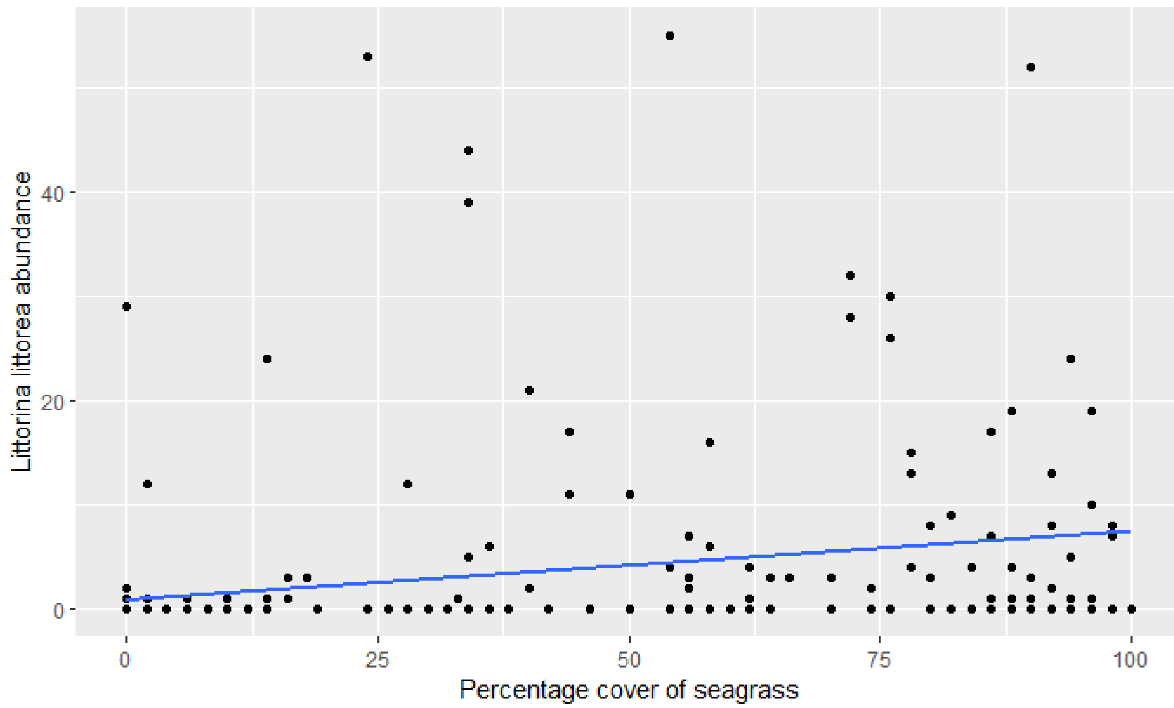

Figure 18 Scatter plot showing the positive relationship between percentage cover of Zostera noltii and Littorina littorea abundance.

3.4 Discussion

Sandy habitats were home to a higher abundance of *Arenicola marina* wormcasts whereas gastropods were found in a higher abundance within seagrass beds. Increased densities of seagrass have been reported to lower the abundance of *Arenicola* species due to the entanglement of rhizome layers within the sediment preventing the burrowing of such species (Goerlitz et al., 2015). This research identified a negative correlation between *Arenicola marina* and *Zostera noltii* however, correlation coefficients showed that variations in the data allowed for an insignificant correlation.

Factors such as water temperature and dissolved oxygen can often act as limiting factors to burrowing organisms such as *Arenicola marina* (Xu et al., 2020). ANOSIM tests have shown that communities within each sampled seagrass bed share few similarities, further supporting the hypothesis that there are other external factors such as increased disturbance from anchoring and mooring that are having a more significant effect on the diversity within the habitats (Duffy, 2006).

The *lebensspuren* diversity of *Z. noltii* habitats changes depending on tidal height however, current data gathered does not allow for in depth analysis surrounding this factor due to the complexities required to allow for thorough representation of tidal height in data (Robbins and Bell, 2000). Further research should take place within shorter timescales allowing for more comprehensive analysis of the impacts of tidal exposure on diversity within the habitats. Seagrass habitats support a rich and diverse habitat for fish species when submerged during high tide (Becker et al., 2012). Research including submersible cameras or beach seining would prove beneficial to investigating the total biodiversity supported by such habitats (Unsworth et al., 2008). Equally, burrowing organisms are found primarily within the sediments meaning there are limitations to investigating biodiversity of burrowing organisms through the identification of lebensspuren found at surface level (Viola et al., 2014). In order to fully investigate the biodiversity of burrowing organisms within seagrass sediments, more invasive sampling techniques such as coring must take place (Pischedda et al., 2011). However, simple methods such as lebenspuren could be useful tools for citizen scientists.

Areas with the highest coverage of algal mats were often situated within 700m of farmland as seen in the most northern transect within the Exmouth Bay area and within the most southern transects of the Holyhead sampling area. Increased rates of eutrophication from farmland has been linked to increases in algal blooms (Schramm, 1999). Increases in nutrient loading into coastal habitats provides ideal nutrients for the replication of bacteria which multiply rapidly creating an algal bloom (O'Neil et al., 2012, Anderson et al., 2002). Increased algal mats limit light availability to flora and fauna found underneath the mat decreasing their rates of survival (Lapointe et al., 2020).

Background levels of nitrates fall below 0.3 mg N/l, however, samples taken within the River Exe prior to sampling reported levels of 1.68 mg N/l (Environment Agency, 2024; Natural Scotland, 2015). Such levels fall under the required figures to categorize waters as experiencing eutrophication in accordance with the Nitrates Directive (1991), however the raised nature of nitrate levels requires monitoring. Further to this, nitrate levels were taken 3 km upstream of the site sampled for this research and so further research is necessary to assess nitrate levels closer to the affected site.

3.5 Conclusion

Investigations have uncovered a significant relationship between *Zostera noltii* and levels of diversity of lebensspuren emphasizing the importance of the protection of the habitats in order to protect biodiversity within UK waters (McHenry et al., 2021). Further research should act to solidify this research by focusing research efforts on the investigation of burrowing organisms through coring and sediment probes (Pischedda et al., 2011) and also through the investigation of species found among the habitat during its submersion at high tide (Unsworth et al., 2008).

Algal mats may have had a significant impact on species found within the area (Schramm, 1999). Through measuring of nitrate levels within the sediment, further research can be conducted in order to confirm the hypothesis that the identified algal blooms were as a result of nutrient loading from eutrophication (Han and Liu, 2014). Such research will help inform our understanding the threats that seagrass habitats within the UK are experiencing and to what extent they are being impacted.

Chapter 4: Evaluating the reliability of previous research investigating carbon burial within UK seagrass habitats.

4.1 Introduction

Global carbon emissions have increased by 2.9% per year (Canadell et al., 2007) until hitting their peak in 2019 (Liu et al., 2022). Such increases in carbon emissions have led to an increased frequency in storms and extreme weather events (Song, 2022) in addition to reductions in oceanic pH (Raven et al., 2005, Strong et al., 2014). Blue carbon habitats have been useful in the offsetting 10% of annual global carbon emissions (Greiner et al., 2013; Macreadie et al., 2021). Blue carbon habitats are marine and coastal ecosystems which have the ability to sequester carbon at faster rates than that of terrestrial sources (Macreadie et al., 2019). Saltmarsh, kelp and seagrass habitats are responsible for the sequestration of more than 30,000 Tg C globally (Macreadie et al., 2021).

Blue carbon habitats are found in a high abundance along areas of the UK coastline (Green et al., 2018). Seagrass and salt marsh habitats make up 0.17% of UK coastal waters (Bertelli and Unsworth, 2014, Burden et al., 2013) while kelp habitats occupy the sunlit subtidal of some rock and boulder habitats (Smale et al., 2016).

A total of 20% of global oceanic carbon sequestration occurs within seagrass habitats (Duarte et al., 2013). Photosynthesis is one way in which seagrass habitats are able to sequester carbon (Raven and Karley, 2006). Seagrass habitats are located throughout shallow temperate and sub-tropical coastlines, typically down to a depth at which irradiance is reduced to around 10% of the surface value. Higher irradiances in shallower water will leads to higher levels of photosynthesis within the area (Dennison, 1987, Zimmerman, 2007).

Carbon burial via particle trapping within seagrass beds is another method of carbon sequestration (Johannessen, 2022). Seagrass canopies slow the flow of water and allow sedimentation of particles, and prevention of resuspension of sediment (Duarte et al., 2013). Prevention of resuspension of sediment and the entanglement of additional particles allows for quicker burial and sedimentation, leading to further burial of organic carbon in the form of coastal and marine debris and detritus (Kennedy et al., 2010).

Degradation of seagrass habitats leads to the release of these previously trapped sediments releasing carbon stores that were previously considered to be in long term storage (Marba et al., 2015). Therefore, degradation of seagrass habitats not only reduces levels of carbon sequestration through photosynthesis and carbon burial (Russell et al., 2013), but also leads to an increase in oceanic carbon as sediment becomes dislodged (Marba et al., 2015).

In this chapter, I investigated current studies used to measure organic carbon levels in relation to seagrass density within the UK. Methods used to conduct the research were analysed, in order to evaluate the reliability of the study and suggest method corrections to improve the reliability of further studies.

4.2 Method

A review was conducted in order to obtain studies that investigated organic carbon levels within UK seagrass. Studies were found using Web of Science using the search terms below (Table 6). Studies were divided by countries and factors regarding the study were recorded. For each study, the size of the core and the organic carbon levels were recorded as reported in the paper. Species of seagrass present and the percentage cover within the area was also recorded. Data was gathered into a table for comparison.

Table 6: Table showing search terms used to find research showing carbon levels within seagrass habitats in the UK.

Primary search terms	Secondary search terms	Tertiary search terms
Seagrass	Carbon	UK
Zostera		England
		Scotland
		Wales
		N. Ireland

4.3 Results

A total of four studies were obtained measuring organic carbon within seagrass sediments within the UK (Table 7). Studies were spread across England, Scotland, and Wales. Current studies investigating the effect of seagrass density on carbon burial within the UK have been limited to the east coast of Scotland, the southwest coast of England and the north of Wales (Figure 19). Studies have investigated carbon burial within seagrass along the west coast of Scotland however, no data was collected within the area to estimate seagrass density.

Table 7: Table showing studies found within literature and the country in which the study took place.

Citation	Country
Do Amaral Camara Lima et al., 2022	England
Green et al., 2018	England
Potouroglou et al., 2021	Scotland
Röhr et al., 2018	Wales

Figure 19 Map displaying sites which have been studied to investigate the relationship between seagrass density and carbon burial within the UK. Blue sites indicate those sampled by Potouroglou et al. (2021). Red sites indicate samples taken by Röhr et al. (2018). Orange sites represent areas sampled by Do Amaral Camara Lima et al. (2022). Green sites were sampled by Green et al. (2018).

Studies reported a core depth ranged between 20 and 50 cm (Table 5). Cores were often unable to be pushed to a greater depth which created variation in the depth of cores within the study as well as creating variation between studies.

1 *Table 8 Studies measuring organic matter within the sediment and the associated recorded abundance of seagrass.*

Country	Site	*Z. noltii* presence	*Z. marina* presence	Seagrass percentage cover	Core depth (cm)	Organic matter	Method	Study
Scotland	Blackness	x	x	50–98	50	1.51	Loss On Ignition	Potouroglou et al., 2021
	Drum Sands	x		3–25	50	1.89	Loss On Ignition	Potouroglou et al., 2021
	Torry Bay	x		50–55	50	1.49	Loss On Ignition	Potouroglou et al., 2021
	Tayport (1)	x		45–60	50	2.52	Loss On Ignition	Potouroglou et al., 2021
	Tayport (2)		x	15–70	50	0.46	Loss On Ignition	Potouroglou et al., 2021
	Montrose	x	x	30–100	50	1.05	Loss On Ignition	Potouroglou et al., 2021
	Beauly	x	x	30–60	50	0.81	Loss On Ignition	Potouroglou et al., 2021
	Findhorn	x		60–70	50	0.28	Loss On Ignition	Potouroglou et al., 2021
	Nairn	x	x	25–65	50	0.26	Loss On Ignition	Potouroglou et al., 2021
	Nigg Bay	x	x	15–45	50	1	Loss On Ignition	Potouroglou et al., 2021
	Dalmore	x	x	30–70	50	0.88	Loss On Ignition	Potouroglou et al., 2021
	Alness	x		25	50	1.39	Loss On Ignition	Potouroglou et al., 2021
	Cromarty	x	x	30–55	50	0.51	Loss On Ignition	Potouroglou et al., 2021
	Tain	x	x	15–65	50	0.47	Loss On Ignition	Potouroglou et al., 2021

Country	Site	*Z. noltii* presence	*Z. marina* presence	Seagrass percentage cover	Core depth (cm)	Organic matter	Method	Study
	Cuthill	x	x	15–65	50	0.38	Loss On Ignition	Potouroglou et al., 2021
England	Looe		x	8.53 ± 6.27	30-35	1.20 ± 0.31	Loss On Ignition	Green et al., 2018
	Cawsands		x	6.08 ± 5.76	30-36	1.25 ± 0.32	Loss On Ignition	Green et al., 2018
	Firestone Bay		x	4.05 ± 5.88	30-37	1.62 ± 0.31	Loss On Ignition	Green et al., 2018
	Drakes Island		x	10.42 ± 8.40	30-38	4.94 ± 2.00	Loss On Ignition	Green et al., 2018
	Jennycliff Bay		x	2.84 ± 4.75	30-39	1.30 ± 0.16	Loss On Ignition	Green et al., 2018
	Yealm CC		x	6.7 ± 7.01	30-40	1.37 ± 0.18	Loss On Ignition	Green et al., 2018
	Tomb Rock		x	4.21 ± 4.51	30-41	1.04 ± 0.21	Loss On Ignition	Green et al., 2018
	Elbery Cove		x	10.63 ± 9.45	30-42	1.33 ± 0.18	Loss On Ignition	Green et al., 2018
	Torre Abbey		x	5.52 ± 5.10	30-43	1.10 ± 0.04	Loss On Ignition	Green et al., 2018
	Fishcombe Cove		x	5.71 ± 7.64	30-44	1.28 ± 0.24	Loss On Ignition	Green et al., 2018
	Hopes Cove		x	7.61 ± 5.98	30-45	0.95 ± 0.68	Loss On Ignition	Green et al., 2018
	Fleet			0	30-46	3.82 ± 1.14	Loss On Ignition	Green et al., 2018
	Studland Bay		x	53.53 ± 10.45	30-47	0.86 ± 0.27	Loss On Ignition	Green et al., 2018

Country	Site	*Z. noltii* presence	*Z. marina* presence	Seagrass percentage cover	Core depth (cm)	Organic matter	Method	Study
	Creek Rythe	x		367.0 ± 115.1 (shoots per m^2)	50	0.50 ± 0.25	Loss On Ignition	Do Amaral Camara Lima et al., 2022
	Hayling Island	x		336.7 ± 95.0 (shoots per m^2)	50	0.38 ± 0.13	Loss On Ignition	Do Amaral Camara Lima et al., 2022
	Porchester	x		302.0 ± 76.1 (shoots per m^2)	50	0.32 ± 0.07	Loss On Ignition	Do Amaral Camara Lima et al., 2022
	Farlington Marshes	x		584 ± 427 (shoots per m^2)	50	0.25 ± 0.14	Loss On Ignition	Do Amaral Camara Lima et al., 2022
	Cowes		x	346 ± 247 (shoots per m^2)	20	0.18 ± 0.07	Loss On Ignition	Do Amaral Camara Lima et al., 2022
	Ryde	x		427 ± 430 (shoots per m^2)	50	0.18 ± 0.03	Loss On Ignition	Do Amaral Camara Lima et al., 2022
Wales	Anglesey		x	774 ± 275 (shoots per m^2)	50	0.7 ± 0.5	Loss On Ignition	Röhr et al., 2018

All three studies used Loss-on-Ignition to analyse obtained samples however, the exact method varied per study. Two out of the four studies found reported the temperature at which the samples were dried (Green et al., 2018, Potouroglou et al., 2021). Samples that were taken from the east coast of Scotland were dried at 60°C for 18 to 24 hours (Potouroglou et al., 2021). Samples taken from the southwest coast of England were dried at 105°C until the mass of the samples ceased to decrease (Green et al., 2018).

Variations were found in the temperature at which samples were placed in the furnace. Samples taken from the east coast of Scotland were placed in a furnace at 500°C for six hours (Potouroglou et al., 2021). Samples that were taken from the south of England were placed in a furnace at the significantly hotter temperature of 950°C for just two hours (Green et al., 2018). Samples taken from the southwest of England were placed in a furnace at 450°C for 24 hours (Do Amaral Camara Lima et al., 2022). Samples taken from North Wales were placed in a furnace 520°C for four hours (Röhr et al., 2018). All samples were acidified with a weak solution of HCl to remove carbonates (Do Amaral Camara Lima et al., 2022; Green et al., 2018; Potouroglou et al., 2021; Röhr et al., 2018).

Table 9: Table showing the drying times, furnace temperatures and time spent in furnace for samples at each site.

Country	Site	Drying Temperature	Furnace Temperature	Furnace Time	Study
Scotland	Blackness	60	500	6 Hours	Potouroglou et al., 2021
	Drum Sands	60	500	6 Hours	Potouroglou et al., 2021
	Torry Bay	60	500	6 Hours	Potouroglou et al., 2021
	Tayport (1)	60	500	6 Hours	Potouroglou et al., 2021
	Tayport (2)	60	500	6 Hours	Potouroglou et al., 2021
	Montrose	60	500	6 Hours	Potouroglou et al., 2021
	Beauly	60	500	6 Hours	Potouroglou et al., 2021
	Findhorn	60	500	6 Hours	Potouroglou et al., 2021
	Nairn	60	500	6 Hours	Potouroglou et al., 2021
	Nigg Bay	60	500	6 Hours	Potouroglou et al., 2021
	Dalmore	60	500	6 Hours	Potouroglou et al., 2021
	Alness	60	500	6 Hours	Potouroglou et al., 2021
	Cromarty	60	500	6 Hours	Potouroglou et al., 2021
	Tain	60	500	6 Hours	Potouroglou et al., 2021
	Cuthill	60	500	6 Hours	Potouroglou et al., 2021
England	Looe	105	950	2 hours	Green et al., 2018
	Cawsands	105	950	2 hours	Green et al., 2018
	Firestone Bay	105	950	2 hours	Green et al., 2018
	Drakes Island	105	950	2 hours	Green et al., 2018
	Jennycliff Bay	105	950	2 hours	Green et al., 2018
	Yealm CC	105	950	2 hours	Green et al., 2018
	Tomb Rock	105	950	2 hours	Green et al., 2018
	Elbery Cove	105	950	2 hours	Green et al., 2018
	Torre Abbey	105	950	2 hours	Green et al., 2018
	Fishcombe Cove	105	950	2 hours	Green et al., 2018
	Hopes Cove	105	950	2 hours	Green et al., 2018
	Fleet	105	950	2 hours	Green et al., 2018

Country	Site	Drying Temperature	Furnace Temperature	Furnace Time	Study
	Studland Bay	105	950	2 hours	Green et al., 2018
	Creek Rythe	72	450	24 hours	Do Amaral Camara Lima et al., 2022
	Hayling Island	72	450	24 hours	Do Amaral Camara Lima et al., 2022
	Porchester	72	450	24 hours	Do Amaral Camara Lima et al., 2022
	Farlington Marshes	72	450	24 hours	Do Amaral Camara Lima et al., 2022
	Cowes	72	450	24 hours	Do Amaral Camara Lima et al., 2022
	Ryde	72	450	24 hours	Do Amaral Camara Lima et al., 2022
Wales	Anglesey		520	4 hours	Röhr et al., 2018

The lowest levels of organic carbon were found in the Nairn site within Scotland. The habitat is home to *Zostera noltii* and *Zostera marina* and samples taken had a percentage cover between 25 and 65% while organic carbon levels within the sediment were 0.26% on average (Potouroglou et al., 2021). The highest levels of organic carbon were found in the Drakes Island, England. The area had an average percentage cover of Z. marina of 10.42% and an average organic carbon level of 4.94% (Green et al., 2018).

4.4 Discussion

Cores used in the UK studies ranged from 20 to 50 cm in depth. The depth of some cores varied between studies depending on how far the core was able to penetrate the sediment (Green et al., 2018). This creates difficulties in the comparison of samples. Organic carbon levels decrease within upper decimeters as the depth of the sediment increases, creating variability in sediment samples taken from different depths (Diesing et al., 2020).

All three studies conducted within the UK used Loss On Ignition to analyse sediment samples. Loss On Ignition is the most common analysis method used to calculate organic matter levels within sediments (Frangipane et al., 2009, Jensen et al., 2018). The Loss On Ignition method has come under criticism as at temperatures between 425 and 520°C, CO_2 can be released from carbonate minerals within the sediment which includes three of the investigations from UK research (Weliky et al., 1983). The removal of CO_2 leads to a loss in mass that is not due to the breakdown of organic matter or organic carbon sources (Frangipane et al., 2009). The Loss On Ignition process can lead to the oxidation of samples which can lead to the increase in mass of a sample invalidating calculations used to calculate organic matter and organic carbon (Vandenberghe et al., 2010). Without the use of another method of sediment analysis, such as thermogravimetric-differential thermal analysis or chemical oxidation, the accuracy of the gathered data cannot be confirmed (Frangipane et al., 2009, Bisutti et al., 2004).

During investigations within the UK the assumption was made that organic carbon comprised of 38% of the organic matter within the sediment as done in other investigations within the UK (Green et al., 2018, Potouroglou et al., 2021). In practice, carbon levels within organic matter can vary and so such assumptions do not accurately predict carbon levels within the sediment (Post et al., 1996, Howard and Howard, 1990).

Carbon storage within the sediment of seagrass habitats is highly influenced by external factors such as structural complexity and impacts of wave action such as wave height and water turbidity (Samper-Villarreal et al., 2016). Complex canopies which include closely entangled shoots allow for the capture of more particles, further encouraging the settlement of sediment and its associated carbon (Tanaya et al., 2018). Increased water turbidity leads to the inability of sediment to settle and allow for the long-term settlement of carbon within the habitat (Hage, 2020). Current investigations have failed to account for factors that may affect rates of carbon burial within seagrass habitats that are unrelated to seagrass density and so further studies must be undertaken in order to truly understand carbon burial rates within UK waters.

Little research has been conducted comparing temperate blue carbon habitats (Duarte et al., 2005). No studies were found that used a similar method to analyse carbon burial within salt marsh habitats in UK waters. In order to understand the efficiency of each habitat with regards to the burial of organic carbon, further studies are needed providing an accurate comparison of organic carbon levels within each habitat (Trevathan-Tackett et al., 2015).

4.5 Conclusion

Few studies have been conducted within the UK investigating the relationship between seagrass density and levels of organic carbon within the sediment creating issues in extrapolating conclusions and assessing the impact of seagrass density nationally (Green et al., 2018). Further studies are required with a particular focus on the east coast of England and the Welsh coastline in order to develop a more accurate, national picture. Studies found within literature reported that there is a significant relationship between the density of seagrass and organic carbon levels within the sediment however, each study has reported a different significance (Potouroglou et al., 2021, Green et al., 2018, Röhr et al., 2018). This may be a result of variations in the methodology of each investigation or due to geographical variation (Nedwell, 1987, Röhr et al., 2018).

In order to determine the impact of seagrass density on the burial of organic carbon a standardised method is required in order to maintain factors such as the depth of sediment cores and ensure national representation. Other blue carbon habitats such as kelp and salt marshes should also be included to provide an accurate comparison of the burial of carbon within blue carbon habitats (Duarte et al., 2005). Methods other than the Loss On Ignition method should be trialled in order to overcome the impact of oxidation and eliminate the need to rely on models to estimate carbon within organic matter (Post et al., 1996, Vandenberghe et al., 2010).

Chapter 5: Discussion

5.1 Blue carbon habitats

Global atmospheric CO_2 levels have increased from approximately 280 ppm to 420 ppm from the time of the industrial revolution to 2023 (IPCC, 2023). Blue carbon habitats are crucial to mitigating against these increases in atmospheric carbon (Howard et al., 2017). Habitats such as seagrass and kelp habitats allow for the stabilisation and burial of carbon within sediments as well as the removal of oceanic CO_2 through photosynthesis (Tanaya et al., 2018).

Seagrass habitats are currently under threat from a range of abiotic and anthropogenic factors such as climate change and habitat fragmentation (Unsworth et al., 2018a). The impact of external factors can lead to reduced levels of carbon storage (Samper-Villarreal et al., 2016). As described in Chapter 2: Investigating threats to UK seagrass habitats and their existing protective policies., there was no relationship between the density of seagrass and the levels of carbon within the habitat. This suggests the possibility of other limiting factors such as water turbidity within the habitats leading to reductions in long-term carbon storage (Duarte et al., 2013).

Further investigations are needed to assess levels of organic carbon within the sediments of other blue carbon habitats such as salt marsh and kelp forests (Pedersen et al., 2021, Sanders et al., 2010). Kelp forests have not received as much research as other blue carbon habitats such as seagrass habitats despite reports that such habitats account for 30% of carbon burial and sequestration within Australian waters (Filbee-

Dexter and Wernberg, 2020). Further research is needed in order to establish the efficiency of each blue carbon habitats and to evaluate appropriate protection for each habitat (Rogers et al., 2019).

5.2 Biodiversity of seagrass habitats

Seagrass canopies provide an intricate network of pathways that provide protection for juvenile fish species (Bertelli and Unsworth, 2014). Seagrass also provides nutrients to coastal organisms (Bell and Westoby, 2004). Studies have shown the importance of seagrass leaves for providing nutrition to juvenile fish and invertebrate species (Jiang et al., 2020). *Littorina* species are herbivores feeding on coastal algae such as seagrass (Voltolina and Sacchi, 1990). Increasing the density of seagrass within an area increases the levels of biodiversity that the habitat can support (Duffy, 2006).

5.3 Protection of seagrass habitats

Current measures of protection focus on intentional physical disturbance of *Zostera* species as seen in the Bern Convention (1982) and the Barcelona Convention (1978) which prevent the intentional dislodging and damaging of such species. Physical disturbances from anthropogenic sources leads to the degradation of seagrass habitats and so such protection is crucial for the protection of such habitats (Cross et al., 2021). While this is beneficial, investigations have now shown eutrophication to have the most significant impact on seagrass habitats (Deegan et al., 2012). In order to provide protection to seagrass habitats from eutrophication there must be increased regulations to waste water treatment and disposal as well as the usage of fertilizers and herbicides in farmland in close proximity to seagrass habitats (Govers et al., 2014, Gao et al., 2011).

A common form of protection for coastal marine habitats such as seagrass in the UK is the use of Marine Protected Areas (Jones, 2012). Each MPA employs its own levels of protection and management designed to protect species and habitats within the area (Maestro et al., 2019). MPAs limit factors such as fishing are limited within the area in order to protect species and habitats (Williams et al., 2015). The use of MPAs has been shown to increase the density of seagrass beds within the areas (Fraschetti et al., 2013). The protection of seagrass habitats is not guaranteed when the habitat is not noted as a listed feature (Tyler-Walters et al., 2016). Many MPAs within the UK list species such as the harbor porpoise *Phocoena phocoena* as a protected feature with only ten identifying seagrass beds as a protected feature meaning that the protection provided to *Phocoena phocoena* is unlikely to benefit seagrass habitats (JNCC, 2020, Tyler-Walters et al., 2016).

5.4 Rewilding of seagrass habitats

Restoration of seagrass habitats leads to the restoration of ecosystem services provided by the habitat (Orth et al., 2020). This understanding has now led to increased importance being placed on the rewilding of seagrass habitats (van Katwijk et al., 2021).

Analysis, protection, and rewilding of *Zostera marina* is currently occurring at Porthdinllaen, North Wales, as a result of funding from Seacology and execution from

Project Seagrass (Project Seagrass, 2023). The project aims to restore ten separate areas totalling 1000 m^2 of seagrass beds from the effects of chains and mooring of boats (Unsworth and Cullen-Unsworth, 2015). Seeds are harvested from donor beds and placed into hessian sacks which are then planted within the area of restoration (Wegoro et al., 2022). Restoration within Loch Craignish began in 2021 through Seawilding, Scottish Association for Marine Science and Project Seagrass as a result of funding from NatureScot Biodiversity Challenge Fund (Project Seagrass, 2023). Single, healthy shoots are planted by hand by divers at a rate of 300 to 400 shoots per hour to restore density within the area (Kent et al., 2021). The Cornish coastline is currently home to a restoration project led by Seasalt Cornwall and Cornwall Wildlife Trust (Cornwall Wildlife Trust, n.d.). Seeds are planted in hessian sacks along the coastline with 1 m distance between them in order to allow for the growth of a rhizome layer to create connectivity within the habitat (Seasalt Cornwall, 2023). Restoration of the seagrass beds aims to increase abundance and diversity of fish species within the area (Cheng et al., 2022).

Seagrass restoration has allowed habitats to achieve the same levels of carbon sequestration within 12 years as naturally occurring seagrass habitats (Greiner et al., 2013). Restored habitats have also acted to increase abundance and diversity of associated fish species within the habitat (Cheng et al., 2022). Such effects are only sustainable if the habitat is maintained through a reduction of anthropogenic impacts within the habitats (Duarte, 2002). Restored seagrass habitats are often still exposed to the threats that caused their decline meaning that the restored habitats will soon again be subject to degradation (Norton, 2009). With increased public education and the increased management and removal of such threats as mooring and eutrophication these restored areas can continue to thrive and grow for years to come (Duarte, 2002). It is worth noting that not all seagrass attempts have proved successful (Van Katwijk et al., 2016). This emphasizes the importance of research and education to ensure that future restoration attempts prove to be successful and allow for the growth and protection of the habitats.

5.5 Conclusion

Seagrass habitats are experiencing significant declines globally as a result of anthropogenic impacts (Green et al., 2021). Seagrass habitats provide a habitat for a diverse range of species (Hyman et al., 2019) and support the growth of commercial fish species (Bertelli and Unsworth, 2014, Unsworth et al., 2018b). Seagrass habitats are also important blue carbon habitats which are able to sequester and bury large quantities of carbon (Filbee‐Dexter and Wernberg, 2020, Green et al., 2018). In order to protect global fisheries, promote biodiversity along coastlines and encourage the burial and sequestration of atmospheric carbon, efforts must be taken to mitigate the anthropogenic factors leading to mass seagrass decline globally (Duarte, 2002, Hughes et al., 2009).

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Appendix

Authors	Article Title	Year
Grech, A; Chartrand-Miller, K; Erftemeijer, P; Fonseca, M; McKenzie, L; Rasheed, M; Taylor, H; Coles, R	A comparison of threats, vulnerabilities and management approaches in global seagrass bioregions	2012
Carus, J; Arndt, C; Schroder, B; Thom, M; Villanueva, R; Paul, M	Using Artificial Seagrass for Promoting Positive Feedback Mechanisms in Seagrass Restoration	2021
Giakoumi, S; Brown, CJ; Katsanevakis, S; Saunders, MI; Possingham, HP	Using threat maps for cost-effective prioritization of actions to conserve coastal habitats	2015
Balestri, E; Menicagli, V; Vallerini, F; Lardicci, C	Biodegradable plastic bags on the seafloor: A future threat for seagrass meadows?	2017
Duarte, CM	The future of seagrass meadows	2002
Jones, BL; Unsworth, RKF	The perilous state of seagrass in the British Isles	2016
Andika, Y; Kawaroe, M; Effendi, H; Zamani, NP	THE EFFECT OF pH CONDITIONS ON PHYSIOLOGICAL RESPONSE OF SEAGRASS LEAVES Cymodocea rotundata	2020
McMahon, K; Collier, C; Lavery, PS	Identifying robust bioindicators of light stress in seagrasses: A meta-analysis	2013
Burke, M	Seagrasses under threat	2006
Hughes, AR; Williams, SL; Duarte, CM; Heck, KL; Waycott, M	Associations of concern: declining seagrasses and threatened dependent species	2009
Short, FT; Polidoro, B; Livingstone, SR; Carpenter, KE; Bandeira, S; Bujang, JS; Calumpong, HP; Carruthers, TJB; Coles, RG; Dennison, WC; Erftemeijer, PLA; Fortes, MD; Freeman, AS; Jagtap, TG; Kamal, AM; Kendrick, GA; Kenworthy, WJ; La Nafie, YA; Nasution, IM; Orth, RJ; Prathep, A; Sanciangco, JC; van Tussenbroek, B; Vergara, SG; Waycott, M; Zieman, JC	Extinction risk assessment of the world's seagrass species	2011
	Marine plastics: What risks and policies exist for seagrass ecosystems in the Plasticene?	2020
Alvarez, A	A mechanistic assessment of the potential ecological risk to seagrass meadows posed by marine echosounders	2019

Authors	Article Title	Year
Marba, N; Jorda, G; Bennett, S; Duarte, CM	Seagrass Thermal Limits and Vulnerability to Future Warming	2022
Perry, D; Hammar, L; Linderholm, HW; Gullstrom, M	Spatial risk assessment of global change impacts on Swedish seagrass ecosystems	2020
Lewis, MA; Devereux, R	NONNUTRIENT ANTHROPOGENIC CHEMICALS IN SEAGRASS ECOSYSTEMS: FATE AND EFFECTS	2009
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Marba, N; Duarte, CM; Holmer, M; Calleja, ML; Alvarez, E; Diaz-Almela, E; Garcias-Bonet, N	Sedimentary iron inputs stimulate seagrass (Posidonia oceanica) population growth in carbonate sediments	2008
Catucci, E; Scardi, M	A Machine Learning approach to the assessment of the vulnerability of Posidonia oceanica meadows	2020
Ruiz, JM; Marco-Mendez, C; Sanchez-Lizaso, JL	Remote influence of off-shore fish farm waste on Mediterranean seagrass (Posidonia oceanica) meadows	2010
Cozza, R; Rende, F; Ferrari, M; Bruno, L; Pacenza, M; Dattola, L; Bitonti, MB	Biomonitoring of Posidonia oceanica beds by a multiscale approach	2019
Chefaoui, RM; Duarte, CM; Serrao, EA	Dramatic loss of seagrass habitat under projected climate change in the Mediterranean Sea	2018
Ontoria, Y; Cuesta-Gracia, A; Ruiz, JM; Romero, J; Perez, M	The negative effects of short-term extreme thermal events on the seagrass Posidonia oceanica are exacerbated by ammonium additions	2019
Zunino, S; Canu, DM; Zupo, V; Solidoro, C	Direct and indirect impacts of marine acidification on the ecosystem services provided by coralligenous reefs and seagrass systems	2019
Pruckner, S; Bedford, J; Murphy, L; Turner, JA; Mills, J	Adapting to heatwave-induced seagrass loss: Prioritizing management areas through environmental sensitivity mapping	2022

Authors	Article Title	Year
Carreno, A; Lloret, J	Environmental impacts of increasing leisure boating activity in Mediterranean coastal waters	2021
Pergent-Martini, C; Monnier, B; Lehmann, L; Barralon, E; Pergent, G	Major regression of Posidonia oceanica meadows in relation with recreational boat anchoring: A case study from Sant'Amanza bay	2022
Traboni, C; Mammola, SD; Ruocco, M; Ontoria, Y; Ruiz, JM; Procaccini, G; Marin-Guirao, L	Investigating cellular stress response to heat stress in the seagrass Posidonia oceanica in a global change scenario	2018
Marba, N; Duarte, CM	Mediterranean warming triggers seagrass (Posidonia oceanica) shoot mortality	2010
Holon, F; Marre, G; Parravicini, V; Mouquet, N; Bockel, T; Descamp, P; Tribot, AS; Boissery, P; Deter, J	A predictive model based on multiple coastal anthropogenic pressures explains the degradation status of a marine ecosystem: Implications for management and conservation	2018
Glasby, TM; West, G	Dragging the chain: Quantifying continued losses of seagrasses from boat moorings	2018
Dahl, M; Bergman, S; Bjork, M; Diaz-Almela, E; Granberg, M; Gullstrom, M; Leiva-Duenas, C; Magnusson, K; Marco-Mendez, C; Pineiro-Juncal, N; Mateo, MA	A temporal record of microplastic pollution in Mediterranean seagrass soils	2021
Unsworth, RKF; Williams, B; Jones, BL; Cullen-Unsworth, LC	Rocking the Boat: Damage to Eelgrass by Swinging Boat Moorings	2017
Chesworth, JC; Donkin, ME; Brown, MT	The interactive effects of the antifouling herbicides Irgarol 1051 and Diuron on the seagrass Zostera marina (L.)	2004
Gao, YP; Fang, JG; Du, MR; Fang, JH; Jiang, WW; Jiang, ZJ	Response of the eelgrass (Zostera marina L.) to the combined effects of high temperatures and the herbicide, atrazine	2017
denHartog, C	Is Sargassum muticum a threat to eelgrass beds?	1997
Martinez-Luscher, J; Holmer, M	Potential effects of the invasive species Gracilaria vermiculophylla on Zostera marina metabolism and survival	2010
Barille, L; Robin, M; Harin, N; Bargain, A; Launeau, P	Increase in seagrass distribution at Bourgneuf Bay (France) detected by spatial remote sensing	2010
Dolbeth, M; Pardal, MA; Lillebo, AI; Azeiteiro, U; Marques, JC	Short- and long-term effects of eutrophication on the secondary production of an intertidal macrobenthic community	2003

Authors	Article Title	Year
Eriander, L; Laas, K; Bergstrom, P; Gipperth, L; Moksnes, PO	The effects of small-scale coastal development on the eelgrass (Zostera marina L.) distribution along the Swedish west coast - Ecological impact and legal challenges	2017
Scarlett, A; Donkin, P; Fileman, TW; Evans, SV; Donkin, ME	Risk posed by the antifouling agent Irgarol 1051 to the seagrass, Zostera marina	1999
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REUSCH, TBH; CHAPMAN, ARO	STORM EFFECTS ON EELGRASS (ZOSTERA-MARINA L) AND BLUE MUSSEL (MYTILUS-EDULIS-L) BEDS	1995
Suykerbuyk, W; Govers, LL; Bouma, TJ; Giesen, WBJT; de Jong, DJ; van de Voort, R; Giesen, K; Giesen, PT; van Katwijk, MM	Unpredictability in seagrass restoration: analysing the role of positive feedback and environmental stress on Zostera noltii transplants	2016
van Katwijk, MM; Bos, AR; Kennis, P; de Vries, R	Vulnerability to eutrophication of a semi-annual life history: A lesson learnt from an extinct eelgrass (Zostera marina) population	2010
Alekseenko, E; Roux, B	Wind effect on bottom shear stress, erosion and redeposition on Zostera noltei restoration in a coastal lagoon; part 2	2019
Valle, M; Chust, G; del Campo, A; Wisz, MS; Olsen, SM; Garmendia, JM; Borja, A	Projecting future distribution of the seagrass Zostera noltii under global warming and sea level rise	2014
Groner, ML; Burge, CA; Couch, CS; Kim, CJS; Siegmund, GF; Singhal, S; Smoot, SC; Jarrell, A; Gaydos, JK; Harvell, CD; Wyllie-Echeverria, S	Host demography influences the prevalence and severity of eelgrass wasting disease	2014

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