MarLIN
Marine Information Network
Information on the species and habitats around the coasts and sea of the British Isles

Zostera (Zostera) marina beds on lower shore or infralittoral clean or muddy sand

MarLIN – Marine Life Information Network
Marine Evidence–based Sensitivity Assessment (MarESA) Review

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***Zostera marina/angustifolia* beds on lower shore or infralittoral clean or muddy sand**
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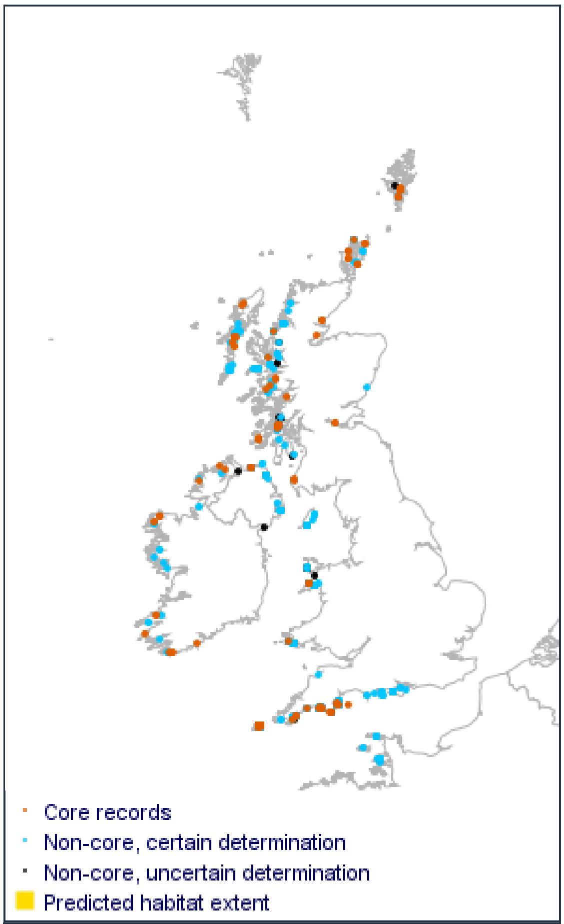

17-09-2018
Biotope distribution data provided by EMODnet Seabed Habitats (www.emodnet-seabedhabitats.eu)

Researched by Emilia d'Avack, Dr Harvey Tyler-Walters & Catherine Wilding

Refereed by Dr Emma Jackson & Dr Leigh Jones

Summary

UK and Ireland classification

EUNIS 2008	A5.5331	*Zostera marina/angustifolia* beds on lower shore or infralittoral clean or muddy sand
JNCC 2015	SS.SMp.SSgr.Zmar	*Zostera (Zostera) marina* beds on lower shore or infralittoral clean or muddy sand
JNCC 2004	SS.SMp.SSgr.Zmar	*Zostera marina/angustifolia* beds on lower shore or infralittoral clean or muddy sand
1997 Biotope	SS.IMS.Sgr.Zmar	*Zostera marina*/angustifolia beds in lower shore or infralittoral clean or muddy sand

Description

Expanses of clean or muddy fine sand and sandy mud in shallow water and on the lower shore (typically to about 5 m depth) can have dense stands of *Zostera marina*/*angustifolia* [Note: the taxonomic status of *Zostera angustifolia* is currently under consideration]. In Zmar the community composition may be dominated by these *Zostera* species and, therefore, characterized by the associated biota. Other biota present can be closely related to that of areas of sediment not containing *Zostera marina*, for example, *Saccharina latissima*, *Chorda filum* and infaunal species such as *Ensis* spp. and *Echinocardium cordatum* (e.g. Bamber, 1993). From the available data, it would appear that a number of sub-biotopes may be found within this biotope dependant on the nature of the substratum and it should be noted that sparse beds of *Zostera marina* may be more readily characterized by their infaunal community. For example, coarse marine sands with seagrass have associated communities similar to MoeVen, SLan or Glap whilst muddy sands may have infaunal populations related to EcorEns, AreISa and FfabMag. Muddy examples of this biotope may show similarities to SundAasp, PhiVir, Are or AfilMysAnit. At present, the data does not permit a detailed description of these sub-biotopes but it is likely that with further study the relationships between these assemblages will be clarified. Furthermore, whilst the *Zostera* biotope may be considered an epibiotic overlay of established sedimentary communities it is likely that the presence of *Zostera* will modify the underlying community to some extent. For example, beds of this biotope in the south-west of Britain may contain conspicuous and distinctive assemblages of Lusitanian fauna such as *Laomedea angulata*, *Hippocampus* spp. and Stauromedusae. In addition, it is known that seagrass beds play an important role in the trophic status of marine and estuarine waters, acting as an important conduit or sink for nutrients and consequently some examples of *Zostera marina* beds have markedly anoxic sediments associated with them. (Information taken from Connor *et al.*, 2004; JNCC, 2015).

Depth range

Lower shore, 0-5 m, 5-10 m

Additional information

The status of *Zostera angustifolia* as a distinct species, a variant of *Zostera marina* or synonym of *Zostera marina* has been the focus of debate. Neither *Zostera angustifolia* nor *Zostera marina* var *angustifolia* are accepted taxonomic names (WorMS, 2015) The current consensus is that *Zostera angustifolia* is a taxonomic synonym of *Zostera marina*. Van Lent & Verschuure (1994) suggest that there is a continuum of life history strategies exhibited by *Zostera marina* for survival in a wider range of environments. Any observed differences in terms of morphology and life history are thus likely to be adaptations to different habitats. A genetic comparison of 'wide-leaved' *Zostera marina* var. *angustifolia* from three locations, as part of a global study using four genetic loci, found that they were indistinguishable from *Zostera marina* (Coyer *et al.*, 2013; Jackson, pers. comm, 2019). Similarly, microsatellite loci did not distinguish the *'angustifolia'* morphotype from *Zostera marina* in Norwegian fjord populations (Olsen *et al.*, 2013).

Listed By

- none -

Further information sources

Search on:

JNCC

Habitat review

Ecology

Ecological and functional relationships

- *Zostera marina* provides shelter or substratum for a wide range of species including fish such as wrasse and goby species (also associated with kelp).
- Leaves slow currents and water flow rates under the canopy and encourage settlement of fine sediments, detritus and larvae (Turner & Kendal,l 1999).
- Seagrass rhizomes stabilize the sediment and protect against wave disturbance and favour sedentary species that require stable substrata and may, therefore, increase species diversity;.
- The leaves are grazed by small prosobranch molluscs, for example, *Rissoa* spp., *Lacuna vincta*, *Hydrobia* spp. and *Littorina littorea*.
- *Zostera marina* bed assemblages may include, in particular, Pipe fish (*Syngnathus typhle*, *Entelurus aequoraeus*), the sea anemones (*Cereus pedunculatus*, *Cerianthus lloydii*) and the neogastropod *Hinia reticulatus*.
- Cuttlefish (*Sepia officinalis*) may lay their eggs amongst sea grass;
- Beds on the south east cost of England may contain distinctive assemblages of Lusitanian fauna such as the hydroid *Laomedea angulata*, Stauromedusae (stalked jellyfish) and, rarely, sea horses *Hippocampus guttulatus*.

Seasonal and longer term change

Zostera beds are naturally dynamic. The population is still recovering from loss of 90 percent of *Zostera marina* beds in 1920s and 1930s as a result of wasting disease. May show marked annual change, for example in the brackish conditions in the Fleet Lagoon (Dorset, UK) leaves die back in autumn and regrow in spring to early summer (Dyrynda, 1997).

Habitat structure and complexity

Seagrasses provide shelter and hiding places. Leaves and rhizomes provide substrata for epibenthic species. These epibenthic species may be grazed by other species (Davison & Hughes, 1998). The sediment supports a rich infauna of polychaetes, bivalve molluscs and burrowing anemones. Amphipods and mysids are important mobile epifauna in seagrass beds. Cockle beds (*Cerastoderma edule*) are often associated with seagrass beds.

Productivity

Seagrass meadows are considered to be the most productive of shallow, sedimentary environments (Davison & Hughes, 1998). The species richness of *Zostera marina* beds in the River Yealm, Devon, UK was significantly higher than that of adjacent sediment (Turner & Kendall, 1999). *Zostera* is directly grazed by ducks and geese. Epiphytes may be as productive as the seagrass they inhabit and are grazed by gastropods. Seagrasses are an important source of organic matter whose decomposition supports detritus based food chains. Seagrass detritus may make an important contribution to ecosystems far removed from the bed itself.

Recruitment processes

Zostera spp. are monoecious perennials (Phillips & Menez, 1988; Kendrick *et al*., 2012; 2017) but may be annuals under stressful conditions (Phillips & Menez, 1988). *Zostera* sp. and seagrasses are flowering plants adapted to an aquatic environment. They reproduce sexually via pollination of flowers and resultant sexual seed but can also reproduce and colonize sediment asexually via rhizomes. Seagrass species disperse and recruit to existing and new areas via pollen, seed, floating fragments or reproductive structures, vegetative growth (via rhizomes), and via biotic vectors such as wildfowl (e.g. geese). Boese *et al.* (2009) found that natural seedling production was not of significance in the recovery of seagrass beds but that recovery was due exclusively to rhizome growth from adjacent perennial beds. However, genetic analysis of populations has revealed that sexual reproduction and seed are more important for recruitment and the persistence of seagrass beds than previously thought (Kendrick *et al*., 2012; 2017). Kendrick *et al.* (2012; 2017) concluded that seagrass species are capable of extensive long distance dispersal based on the high level of genetic diversity and connectivity observed in natural populations.

Zostera sp. flowers release pollen in long strands, dense enough to remain at the depth they were released for several days, therefore, increasing their chance of pollinating receptive stigmas. Pollen are long-lived (ca 8 hours) but not ideally for long-distance dispersal so that the pollen of *Zostera noltei* is estimated to travel up to 10 m, while that of *Zostera marina* travels up to 15 m, although most are intercepted by the canopy within 0.5 m (Zipperle *et al*., 2011; McMahon *et al*., 2014; Kendrick *et al*., 2012; 2017). Pollination occurs mostly within the seagrass meadow or adjacent meadows, and outcrossing is high in *Zostera* sp. (Zipperle *et al*., 2011). Zipperle *et al.* (2011) that the low level of inbreeding observed was due to self-incompatibility resulting in seed abortion or seedling mortality.

Seeds develop within a membranous wall that photosynthesises, developing an oxygen bubble within the capsule, eventually rupturing the capsule to release the seed. *Zostera* sp. seeds are negatively buoyant and generally sink. Hootsmans *et al.* (1987) reported that each flowering shoot of Zostera noltei produces 3-4 flowers containing 2-3 seed each. They estimated a potential seed production of 9000/m based on the maximum density of flowering shoots in their quadrats in the Zandkreek, Netherlands. Most seeds were released in August in the Zandkreek but the actual seed densities were much lower than predicted (Hootsmans *et al*., 1987). However, the density of flowering shoots is highly variable. Phillips & Menez (1988) state that seedling mortality is extremely high. Fishman & Orth (1996) report that 96% of *Zostera marina* seeds were lost from uncaged test areas due to transport (dispersal) or predation. Phillips & Menez (1988) note that seedlings rarely occur within the eelgrass beds except in areas cleared by storms, blow-out or excessive herbivory. Den Hartog (1970) noted that although the seed set was high, *Zostera noltei* seedlings were rarely seen in the wild, suggesting that vegetative reproduction may be more important than sexual reproduction (Davison & Hughes, 1998). Experimental germination was increased by low salinity (1-10 psu) in *Zostera noltei* and no germination occurred at salinities above 20 psu, however, germination was independent of temperature (Hughes *et al*., 2000). Hootsmans *et al.* (1987) noted that potential recruitment was maximal (32% of seeds) at 30°C and 10 psu, and no recruitment occurred at 30 psu and they estimated that, in 1983 <5>Zostera noltei plants in the Zandkreek originated from seed.

Manley *et al.* (2015) reported that seed density in *Zostera marina* meadows in Hog Island Bay, Virginia, USA, decreased with increasing distance from the parent, that seed predation was low regardless of the distance from the edge of the bed, and that the seed density was strongly correlated with seed density from the previous year. They concluded that *Zostera* could quickly rebound from disturbances as long as a seed source remained.

Seeds have a limited dispersal range of a few metres although they may be dispersed by storms that disturb the sediment (Zipperle *et al.*, 2009b, 2011; McMahon *et al.*, 2014; Kendrick *et al.*, 2012; 2017). However, in New York, USA, Churchill *et al.* (1985) recorded 5-13% of *Zostera marina* seeds with attached gas bubbles and achieved an average dispersal distance of 21 m and up to 200 m in a few cases. Seeds can also be dispersed within positively buoyant flowering branches (rhipidia) for weeks or months, and up to 100s of kilometres i.e. 20-300 km (McMahon *et al.*, 2014; Kendrick *et al.*, 2012; 2017). Kendrick *et al.* (2012) noted that genetic differences between seagrass populations (inc. *Zostera marina* and *Zoster noltei*) showed limited differences regionally, i.e. <100>Zostera marina rhipidia fragments could be transported over 150 km (Kendrick *et al.*, 2012; 2017).

Seagrass seeds may also be transported in the gut of fish, turtles, dugong, manatees, and in the gut or on the feet of waterfowl (McMahon et al., 2014; Kendrick *et al.*, 2012; 2017). For example, 30% of freshwater eelgrass (*Naja marina*) seeds fed to ducks in Japan survived and successfully germinated after passage through their alimentary canals and potentially transported 100-200 km (Fishman & Orth, 1996). McMahon *et al.* (2015) noted that *Zostera* seeds are dormant and viable for 12 months or more. However, the extent of their biotic dispersal is unclear.

Seagrass reproduces vegetatively, i.e. by the growth of rhizome. Vegetative reproduction was thought to exceed seedling recruitment except in areas of sediment disturbance (Reusch *et al.* 1998; Phillips & Menez 1988), although genetic analysis suggests a more complex process (Kendrick *et al.*, 2012; 2017). New leaves appear in spring and seedling appear in spring, and eelgrass meadows develop over intertidal flats in summer, due to vegetative growth. However, *Zostera marina* plants are monomorphic, restricted to the horizontal growth of roots and, hence, unable to grow rhizomes vertically. This restriction to horizontal elongation of the roots makes the recolonization of adjacent bare patches difficult and explains why large beds are only found in gently sloping locations. A depression of the seabed caused by disturbance of the sediment can thus restrict the expansion of the bed. The size and shape of impacted areas will also have a considerable effect on resilience rates (Creed *et al.*, 1999). Larger denuded areas are likely to take longer to recover than smaller scars, for example, seagrass beds are likely to be more resilient to physical damage resulting from narrow furrows left after anchoring because of large edge to area ration and related availability of plants for recolonization. Manley *et al.* (2015) reported a rhizome growth rate of 26 cm/yr. in *Zostera marina*.

Recruitment and recovery of seagrass meadows depend on numerous factors and is an interplay between seed recruitment to open or disturbed areas, the seed bank, and expansion by vegetative growth. Recruitment is also affected by local environmental conditions, and isolation due to coastal geomorphology such as islands and inlets, hydrography and even biological structures. For example, ecological genetics studies of *Zostera marina* in False and Padilla Bays on Pacific coast of USA (Ruckelhaus, 1998) detected genetic differentiation between intertidal and subtidal zones and between the bays. Estimates of gene flow suggested that seed dispersal was more important than pollen dispersal, effective migration (2.9 migrants/generation) occurred between the bays (14 km apart) and that the population subdivision was in part explained by disturbance and recolonization. Also, genetic differentiation between *Zostera marina* populations was six times higher between Norwegian fjords than within fjords (Olsen *et al.*, 2013; Kendrick *et al.*, 2017). Reynolds *et al.* (2013) estimated that natural recovery of *Zostera marina* seagrass beds in the isolated coastal bays of the Virginian coast, USA would have taken between 125 and 185 years to recover from the substantial decline due to wasting disease in the 1930s. Although small patches were observed in the 1990s seagrass was locally extinct for 60 years. Seed transplantation in the late 1990s resulted in the restoration of ca 1600 ha of seagrass within 10 years (Reynolds *et al.*, 2013). In addition, examination of seagrass meadows in Ria Formosa, Portugal, suggested that large and non-fragmented seagrass meadows had higher persistence values than small, fragmented meadows and, hence, that smaller patches were more vulnerable to disturbance (Cunha & Santos, 2009). Fonseca & Bell (1998) also suggested that loss of cover (below ca 50%) led to fragmentation, and loss of habitat structural integrity.

Time for community to reach maturity

Zostera marina beds are unlikely to seed and establish rapidly. There has been little recovery of these beds since the 1930s. In Danish waters *Zostera marina* beds could take at least 5 years to establish even when near to established beds. Seeding over distances is likely to be slow.

Additional information

Seagrass beds may act as corridor habitats for species moving from warm waters. Seasonal die back resulted in sediment destabilization as well as loss of cover for fish in the Fleet, Dorset, UK (Dyrynda, 1997).

Preferences & Distribution

Habitat preferences

Depth Range	Lower shore, 0-5 m, 5-10 m
Water clarity preferences	No information found
Limiting Nutrients	Nitrogen (nitrates), Phosphorous (phosphates)
Salinity preferences	Full (30-40 psu), Variable (18-40 psu)
Physiographic preferences	Enclosed coast / Embayment
Biological zone preferences	Upper infralittoral
Substratum/habitat preferences	Mud, Mud and sandy mud, Muddy sand, Sand, Sand and muddy sand
Tidal strength preferences	Moderately Strong 1 to 3 knots (0.5-1.5 m/sec.), Very Weak (negligible), Weak < 1 knot (<0.5 m/sec.)
Wave exposure preferences	Extremely sheltered, Moderately exposed, Sheltered, Very sheltered
Other preferences	

Additional Information

Intertidal *Zostera marina* beds may be damaged by frost, although rhizomes most likely survive. In carbonate based sediments phosphate may be limiting due to adsorption onto sediment particles. *Zostera marina* is also found in reduced salinities, for example brackish lagoons (Dyrynda, 1997).

Species composition

Species found especially in this biotope

- *Cladosiphon zosterae*
- *Entocladia perforans*
- *Halothrix lumbricalis*
- *Laomedia angulata*
- *Leblondiella densa*
- *Myrionema magnusii*
- *Punctaria crispata*
- *Rhodophysema georgii*

Rare or scarce species associated with this biotope

- *Halothrix lumbricalis*
- *Laomedia angulata*
- *Leblondiella densa*

Additional information

Species richness is derived from the number of species recorded in MNCR database for this biotope. *Zostera* beds, in particular *Zostera marina*, are species rich habitats. Species diversity is highest in subtidal, fully marine, perennial populations of *Zostera marina* when compared to intertidal, estuarine or annual beds of *Zostera* spp. Representative and characteristic species are listed by Davison & Hughes (1998). Species lists for major eelgrass beds are available for the Helford Passage (Sutton & Tompsett, 2000) and Isles of Scilly (Hiscock, S., 1984). Hiscock, S. (1987) listed 67 algae in *Zostera marina* beds in the Isles of Scilly. Proctor (1999) lists 63 species of fauna in *Zostera* sp. beds in Torbay. Hiscock, S. (1987) noted that colonial diatoms were the most abundant algae on *Zostera marina* leaves in the Isles of Scilly. However, it should be noted that species lists are likely to underestimate the total number of species present, especially with respect to microalgae epiphytes, bacteria and meiofauna.

Sensitivity review

Sensitivity characteristics of the habitat and relevant characteristic species

Although a wide range of species are associated with seagrass beds, which provide habitat and food resources, these species occur in a range of other biotopes and were therefore not considered by to characterize the sensitivity of this biotope (d'Avack *et al.*, 2014) . However, seagrasses worldwide have been shown to exhibit a three-way symbiotic relationship with the small lucinid bivalves (hatchet-shells, e.g. *Loripea* and *Lucinoma*) and their endosymbiotic sulfide-oxidizing gill bacteria (Van der Heide *et al.*, 2012). In experiments, the sulfide-oxidizing gill bacteria of *Loripes lacteus* were shown to reduce sulfide levels in the sediment and enhance the productivity of *Zostera noltei*, while the oxygen relased from the roots of *Zoster noltei* was of benefit to *Loripes* (Van der Heide *et al.*, 2012). Epiphytic grazers, such as *Hydrobia ulvae*, *Rissoa* spp. and *Lacuna vincta* remove fouling epiphytic algae that would otherwise smother *Zostera* spp. *Hydrobia ulvae* and *Lacuna* spp. have been shown to reduce the density of epiphytes on *Zostera noltei* in the Dutch Wadden Sea (Philippart, 1995a) and *Zostera marina* in Puget Sound (Nelson, 1997) respectively with subsequent enhancement of the productivity of sea grass. Nevertheless, *Zostera marina* is the main species creating this habitat and the removal or loss of *Zostera marina* plants would result in the disappearance of this biotope. Therefore, *Zostera marina* is considered to be the most important species for the development of and, hence, sensitivity of the biotope, although the effects of pressures on other components of the community are reported where relevant.

Resilience and recovery rates of habitat

d’Avack *et al.* (2014) reported that although seagrass species are fast-growing and relatively short-lived, they can take a considerable time to recover from damaging events if recovery does occur at all. Every seagrass population will have a different response to pressures depending on the magnitude or duration of exposure pressure as well as the nature of the receiving environment. In general terms, the resilience of seagrass biotopes to external pressures is low, as shown by the very slow or lack of recovery after the epidemic of the wasting disease in the 1930s.

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Seeds have a limited dispersal range of a few metres although they may be dispersed by storms that disturb the sediment (Zipperle *et al.*, 2009b, 2011; McMahon *et al.*, 2014; Kendrick *et al.*, 2012; 2017). However, in New York, USA, Churchill *et al.* (1985) recorded 5-13% of *Zostera marina* seeds with attached gas bubbles and achieved an average dispersal distance of 21 m and up to 200 m in a few cases. Seeds can also be dispersed within positively buoyant flowering branches (rhipidia) for weeks or months, and up to 100s of kilometres i.e. 20-300 km (McMahon *et al.*, 2014; Kendrick *et al.*, 2012; 2017). Kendrick *et al.* (2012) noted that genetic differences between seagrass populations (inc. *Zostera marina* and *Zoster noltei*) showed limited differences regionally, i.e. <100 km but increased with long-distances of hundreds of kilometres. In Swedish waters, a model predicted that *Zostera marina* rhipidia fragments could be transported over 150 km (Kendrick *et al.*, 2012; 2017).

Seagrass seeds may also be transported in the gut of fish, turtles, dugong, manatees, and in the gut or on the feet of waterfowl (McMahon et al., 2014; Kendrick *et al.*, 2012; 2017). For example, 30% of freshwater eelgrass (*Naja marina*) seeds fed to ducks in Japan survived and successfully germinated after passage through their alimentary canals and potentially transported 100-200 km (Fishman & Orth, 1996). McMahon *et al.* (2015) noted that *Zostera* seeds are dormant and viable for 12 months or more. However, Dooley *et al.* (2013) reported that the viability of one-year-old *Zostera marina* seeds was 77% but that viability dropped to only 32% in four-year-old seeds.

Similarly, 68% of one-year-old seeds in their study germinated but only 15% in three-year-old seeds and successful seedlings resulted from only ca 5% of fresh seeds (Dooley *et al.*, 2013). The extent of the biotic dispersal of seeds is unclear (McMahon et al., 2014; Kendrick *et al.*, 2012; 2017).

Seagrass reproduces vegetatively, i.e. by the growth of rhizome. Vegetative reproduction was thought to exceed seedling recruitment except in areas of sediment disturbance (Reusch *et al.* 1998; Phillips & Menez 1988), although genetic analysis suggests a more complex process (Kendrick *et al.*, 2012; 2017). New leaves appear in spring and seedling appear in spring, and eelgrass meadows develop over intertidal flats in summer, due to vegetative growth. However, *Zostera marina* plants are monomorphic, restricted to the horizontal growth of roots and, hence, unable to grow rhizomes vertically. This restriction to horizontal elongation of the roots makes the recolonization of adjacent bare patches difficult and explains why large beds are only found in gently sloping locations. A depression of the seabed caused by disturbance of the sediment can thus restrict the expansion of the bed. The size and shape of the impacted areas will also have a considerable effect on resilience rates (Creed *et al.*, 1999). Larger denuded areas are likely to take longer to recover than smaller scars, for example, seagrass beds are likely to be more resilient to physical damage resulting from narrow furrows left after anchoring because of large edge to area ration and related availability of plants for recolonization. Manley *et al.* (2015) reported a rhizome growth rate of 26 cm/yr. in *Zostera marina*.

Recruitment and recovery of seagrass meadows depend on numerous factors and is an interplay between seed recruitment to open or disturbed areas, the seed bank, and expansion by vegetative growth. Recruitment is also affected by local environmental conditions, and isolation due to coastal geomorphology such as islands and inlets, hydrography and even biological structures. For example, ecological genetics studies of *Zostera marina* in False and Padilla Bays on the Pacific coast of USA (Ruckelhaus, 1998) detected genetic differentiation between intertidal and subtidal zones and between the bays. Estimates of gene flow suggested that seed dispersal was more important than pollen dispersal, effective migration (2.9 migrants/generation) occurred between the bays (14 km apart) and that the population subdivision was in part explained by disturbance and recolonization. Also, genetic differentiation between *Zostera marina* populations was six times higher between Norwegian fjords than within fjords (Olsen *et al.*, 2013; Kendrick *et al.*, 2017). Reynolds *et al.* (2013) estimated that natural recovery of *Zostera marina* seagrass beds in the isolated coastal bays of the Virginian coast, USA would have taken between 125 and 185 years to recover from the substantial decline due to wasting disease in the 1930s. Although small patches were observed in the 1990s seagrass was locally extinct for 60 years. Seed transplantation in the late 1990s resulted in the restoration of ca 1600 ha of seagrass within 10 years (Reynolds *et al.*, 2013). In addition, an examination of seagrass meadows in Ria Formosa, Portugal, suggested that large and non-fragmented seagrass meadows had higher persistence values than small, fragmented meadows and, hence, that smaller patches were more vulnerable to disturbance (Cunha & Santos, 2009). Fonseca & Bell (1998) also suggested that loss of cover (below ca 50%) led to fragmentation, and loss of habitat structural integrity.

Genetic diversity also influences the resilience of seagrasses in particular when pressure persists over a long period of time. The genetic diversity of *Zostera* populations is very high, particularly in the NE Atlantic (Olsen *et al.*, 2004; Kendrick *et al.*, 2012; 2017). Rice & Emery (2003) showed that evolutionary change in seagrasses can occur within a few generations, suggesting that genetically diverse population would be more resilient to changes in environmental conditions compared to genetically conserved populations. Pressures causing a rapid change in seagrass environments will have a greater impact as the natural ability of the plants to adapt is compromised. Plasticity is a further key element in determining the resilience of seagrass biotopes. Maxwell *et al.* (2014) investigated the response of seagrass ecosystems to severe weather events (i.e. flooding) in order to understand the process that promotes acclimation. The study found that phenotypic plasticity (changes in physiological and morphological characteristics) enabled the species to cope with varying degree of stress to avoid mortality. Phenotypic plasticity can thus increase the length of time seagrass can persist in unfavourable environments such as reduced light availability. Different populations will thus have different resilience to external pressures. Different populations will thus have different resilience to external pressures. For example, Boese *et al.* (2009) examined the recolonization of gaps created experimentally within *Zostera marina* beds. The study looked at two zones, the lower intertidal covered with almost continuous seagrass and an upper intertidal transition zone where there were patches of perennial and annual *Zostera marina*. Recovery started within a month after the disturbance of the lower intertidal continuous perennial beds and was complete after two years. Plots in the transition zone, however, took almost twice as long to recover.

Resilience assessment. The resilience of seagrass beds and the ability to recover from human induced pressures is a combination of the environmental conditions of the site, including the supply of seed or other propagules, the remaining seed bank and vegetative growth but also the hydrodynamics (i.e. local and regional currents or isolation within bays or inlets), growth rates of the seagrass, and the scale, frequency (repeated disturbances versus a one-off event) and intensity of the disturbance. This highlights the importance of considering the species affected as well as the ecology of the seagrass bed, the environmental conditions and the types and nature of activities giving rise to the pressure. Changes in biological communities after seagrass disappear might impact seagrass resilience. A rise in the abundance of sea urchin, for instance, could prevent the recovery of seagrass beds due to increased herbivory (Valentine & Heck Jr, 1991). The removal of seagrass plants can induce a negative feedback loop inhibiting recovery. Indeed the removal of plants can cause chronic turbidity due to continual resuspension of unconsolidated sediments. When water quality conditions do not return to their original state, recovery of seagrass beds may not occur at all (Giesen *et al.*, 1990). Fragmentation of existing meadows may also increase their vulnerability to further disturbance (Fonseca & Bell, 1998; Cunha & Santos, 2009). In addition, recovery from the substantial loss of seagrass beds in the North Atlantic due to wasting disease in the 1930s has been limited (Davidson & Hughes, 1998). Seagrass beds remain nationally scarce in the UK and may have declined 25-45% in the last 25 years (although detailed datasets are lacking) but many beds remain under threat (Jackson *et al.*, 2013; Jones & Unsworth, 2015). Therefore, recovery from long-term, large-scale impacts may take several decades, especially where the loss of the seagrass beds result in changes in the habitat, loss of the seed bank or isolation slows recruitment. Therefore, where resistance is assessed as 'Medium' or 'Low', resilience is probably **'Medium'** and where resistance is 'None', resilience is probably **'Very low'**, depending on the effects of the pressure on the habitat.

It should be noted that the recovery rates are only indicative of the recovery potential. Recovery of impacted populations will always be mediated by stochastic events and processes acting over different scales including, but not limited to, local habitat conditions, further impacts and processes such as larval-supply and recruitment between populations.

Hydrological Pressures

	Resistance	Resilience	Sensitivity

Temperature increase (local)	Medium Q: High A: High C: High	Medium Q: High A: Low C: Medium	Medium Q: High A: Low C: Medium

Temperature is considered the overall parameter controlling the geographical distribution of seagrasses. All enzymatic processes, related to plant metabolism are temperature dependent and specific life cycle events, such as flowering and germination, are also often related to temperature (Phillips *et al*., 1983). For seagrasses, temperature affects biological processes by increasing reaction rates of biological pathways. Photosynthesis and respiration increase with higher temperature until a point where enzymes associated with these processes are inhibited. Beyond a certain threshold, high temperatures will result in respiration being greater than photosynthesis resulting in a negative energy balance. Increased temperatures do also encourage the growth of epiphytes increasing the burden upon seagrass beds and making them more susceptible to disease (Rasmussen, 1977). *Zostera marina* can tolerate temperatures between -1 to 25°C with optimum conditions for growth being around 10 to 15°C, and 10°C for seedling development (Hootsmans *et al*., 1987). Nejrup & Pedersen (2007) found that temperatures between 25 and 30°C lowered photosynthetic rates by 50% as well as growth (production of new leaves by 50% and leaf elongation rate by 75%). High temperatures also resulted in a 12-fold increase in mortality of *Zostera marina* plants. Moore *et al.* (2014) found that short-term exposures to a rapid increase of 4–5°C above normal temperature (25°C) during summer months resulted in widespread diebacks of *Zostera marina*. Recovery was observed to be minimal as the seagrass was replaced by *Ruppia maritima*. Similarly, Salo & Peterson (2014) found that exposure to high temperature for five weeks led to enhanced mortality, reduced formation of new leaves and a lower number of standing leaves per shoot. Orth & Moore (1983) reported that the majority (68%) of *Zostera marina* seeds germinated in the winter months between 0-10°C, and that germination was most rapid between 5-10°C but virtually no germination was observed when temperatures were above 20°C, in Chesapeake Bay, USA.

Other species associated with seagrass habitats are also affected by changes in temperature. For instance, the gastropod *Lacuna vincta*, an important grazer found in seagrass beds, is near its southern range limit in the British Isles. Long-term increases in temperature due to human activity may limit the survival of the snail and restrict subsequent distribution whilst a short-term acute temperature increase may cause death. The loss of grazers could have detrimental effects on seagrass beds as the leaves provide a substratum for the growth of many species of epiphytic algae. These epiphytes may smother the *Zostera* plants unless kept in check by the grazing activities of gastropods and other invertebrates. Healthy populations of epiphyte grazers are therefore essential to the maintenance of seagrass beds.

Sensitivity assessment. High temperatures during hot summer months have caused massive die-off events among seagrasses worldwide (Moore & Jarvis, 2008; Reusch *et al*., 2005). A 5°C change in temperature over one month or a 2°C change over the period of a year is thus likely to result in some *Zostera marina* mortality. In addition, a longer-term or persistent increase in temperature may reduce germination rates and hence reduce recruitment and resilience (Jackson, pers comm., 2019). Therefore, resistance is assessed as **'Medium'**. Recovery will be fairly rapid once conditions return to normal resulting in a **'Medium'** resilience score. If however, temperatures remain elevated for a prolonged period of time, *Zostera marina* can be out-competed and subsequently excluded from the habitat by other species such as *Ruppia maritima*. Overall, the biotope is assessed as **'Medium'** sensitivity to an increase in temperature at the pressure benchmark.

Temperature decrease (local)	High Q: High A: Medium C: Low	High Q: High A: High C: High	Not sensitive Q: High A: Medium C: Low

Temperature is considered the overall parameter controlling the geographical distribution of seagrasses. All enzymatic processes related to plant metabolism are temperature dependent and specific life cycle events, such as flowering and germination, are also often related to temperature (Phillips *et al.*, 1983). For seagrasses, temperature affects biological processes by increasing reaction rates of biological pathways. Photosynthesis and respiration increase with higher temperature until a point where enzymes associated with these processes are inhibited. Beyond a certain threshold, high temperatures will result in respiration being greater than photosynthesis resulting in a negative energy balance. Increased temperatures do also encourage the growth of epiphytes increasing the burden upon seagrass beds and making them more susceptible to disease (Rasmussen, 1977). *Zostera marina* can tolerate temperatures between -1 to 25°C with optimum conditions for growth being around 10 to 15°C, and 10°C for seedling development (Hootsmans *et al.*, 1987). Nejrup & Pedersen (2007) found that low water temperatures (5°C) slowed down the photosynthetic rate by 75%; growth was also affected, with the production of new leaves reduced by 30% and leaf elongation rate reduced by 80% compared to the control, however, mortality was not affected.

Other species associated with seagrass habitats are also affected by changes in temperature. For instance, the gastropod *Lacuna vincta*, an important grazer found in seagrass beds, is near its southern range limit in the British Isles. Long-term change in temperature due to human activity may limit the survival of the snail and restrict subsequent distribution whilst a short-term acute temperature increase may cause death, although it may be replaced by other grazers. Healthy populations of epiphyte grazers are therefore essential to the maintenance of seagrass beds.

Sensitivity assessment. Overall, a decrease in temperature is likely to reduce growth rates but not to cause mortality directly. Frost damage could occur to plants exposed at extreme low tides in the winter months but as the seagrass dies back in winter this is unlikely to be significant. Therefore, a 5°C decrease in temperature over one month or a 2°C decrease over the period of a year is thus unlikely to result in some *Zostera marina* mortality. Resistance is therefore considered 'High'. Recovery will be rapid once conditions return to normal resulting in a 'High' resilience score. Hence, the biotope is considered be 'Not sensitive' to a decrease in temperature at the pressure benchmark.

Salinity increase (local)	Low Q: Medium A: Medium C: Medium	Medium Q: High A: Low C: Medium	Medium Q: Medium A: Low C: Medium

In general, seagrass species have a wide salinity tolerance. Nejrup & Pedersen (2008) reported optimum salinities between 10 and 25 ppt. Hypersaline conditions can affect seagrass performance as changes in salinity may increase the energy requirements due to demanding osmotic adjustments (Touchette, 2007). Den Hartog (1997) stated that *Zostera noltei* has a greater tolerance to extremes salinities compared to Zostera marina due to its intertidal habitat. Vermaat *et al.* (2000) investigating salinity tolerance in *Zostera noltei* found considerable mortalities of plants at a salinity of 35 ppt. These findings suggest that both *Zostera* species are ill-equipped to withstand high saline conditions. A review by d'Avack *et al.* (2014) reported that phenotypic plasticity can play an important role in the ability of seagrasses to withstand external pressures such as changes in salinity. Changes in physiological and morphological characteristics of seagrass plants will enable species to cope with varying degrees of stress for an extended period of time

(Maxwell *et al*., 2014).

Sensitivity assessment. Even though *Zostera* plants display a wide tolerance to a range of salinities, an increase from 35 to 38 units for the period of one year will cause some mortality in *Zostera marina*. The subtidal habitat makes the species more vulnerable to salinity extremes compared to the intertidal *Zostera noltei* resulting in a 'Low' resistance score. *Zostera marina* will thus be adversely affected by activities such as brine discharges from seawater desalination plant. Recovery, enabled by recolonization from surrounding communities, will be fairly rapid once conditions return to normal resulting in a 'Medium' resilience score. The biotope is therefore considered to have a 'Medium' sensitivity to this pressure at the pressure benchmark.

Salinity decrease (local)	Medium Q: Medium A: Medium C: Medium	Medium Q: High A: Low C: Medium	Medium Q: Medium A: Low C: Medium

In general, seagrass species have a wide salinity tolerance. Nejrup & Pedersen (2008) reported optimum salinities between 10 and 25 ppt, while den Hartog (1970) reported tolerance to salinities as low as 5 ppt. Hyposaline conditions (reduced salinity) can, however, affect seagrass performance as changes in salinity may increase the energy requirements due to demanding osmotic adjustments (Touchette, 2007).

A study by Salo *et al*. (2014) found that hyposaline conditions can seriously impair plant performance and survival rates. The study determined that the severity of impact will be population specific as seagrass populations from different areas may substantially differ in their salinity tolerance range with population naturally occurring in low saline areas having greater resistance to this pressure.

Salo & Petersen (2014) experimentally tested the effects of different combinations of salinity and temperature on the physiological performance of *Zostera marina*. The study found that the combination of high temperature and low salinity resulted in high mortality rates, highlighting negative synergistic effects when seagrasses are exposed to multiple pressures.

A review by d'Avack *et al*. (2014) determined that phenotypic plasticity can play an important role in the ability of seagrasses to withstand external pressures such as changes in salinity. Changes in physiological and morphological characteristics of seagrass plants will enable species to cope with varying degrees of stress for an extended period of time (Maxwell *et al*., 2014).

Sensitivity assessment. *Zostera marina* has a wide salinity tolerance. Reduced salinity will, however, impact performance causing some mortality. Resistance is therefore considered 'Medium'. Effects can be exasperated when the seagrass is exposed to multiple stressors at the same time, highlighting the importance to consider negative synergistic effects when conduction assessments. Recovery is considered fairly rapid once conditions return to normal resulting in a 'Medium' resilience score. The biotope is therefore considered to have a 'Medium' sensitivity to this pressure at the pressure benchmark.

Water flow (tidal current) changes (local)	Medium Q: High A: High C: High	Medium Q: High A: Low C: Medium	Medium Q: High A: Low C: Medium

A complex interaction exists between seagrass beds and water flow. Water flow determines the upper distribution of plants on the shore whilst plants mediate the velocity of the flow by extracting momentum from the moving water. Reducing the flow increases water transparency (see 'changes in suspended sediments' pressure) and causes the deposition and retention of fine sediments. Increased flow rates, on the other hand, are likely to erode sediments, expose rhizomes and lead to loss of plants.

The highest current velocity a seagrass can withstand is determined by a threshold beyond which sediment re-suspension and erosion rates are greater than the seagrasses ability to bind sediment and attenuate currents. In very strong currents, leaves might lie flat on the sea bed reducing erosion under the leaves but not on the unvegetated edges which begin to erode. High velocity currents can thus change the configuration of patches within a meadow, creating striations and mounding in the seagrass beds. Such turreted profiles destabilise the bed and increase the risk of 'blow outs' (Jackson *et al*., 2013). Populations found in stronger currents are usually smaller, patchy and more vulnerable to storm damage.

A review by Koch (2001) determined that the range of current velocities tolerated by seagrass lies approximately between a minimum of 5 cm/s and a maximum of 180 cm/s. Fonseca *et al.* (1983) found a lower maximum for *Zostera marina* and estimated the highest current velocity at approximately 120–150 cm/s.

Human activities in coastal waters which alter hydrology have been implicated in the disappearance of seagrass beds. For instance, van der Heide *et al.* (2007) noted that the construction of a dam in the Wadden Sea influencing the hydrological regime inhibited the recovery of *Zostera* plants after their initial decline following the wasting disease in the 1930s. Aquaculture installations can also change water flow and have shown to directly impact seagrass habitats. Everett *et al.* (1995) experimentally altered water flow to investigate the effects of the commercial culture of the oyster *Magallana gigas* on *Zostera marina*, using both stake and rack methods. The study found that both culture methods caused a sharp decline in *Zostera marina* plants with cover being less than 25% compared to control plots after one year of culture due to changes in local hydrological regime. Both culture methods produced strong, although dissimilar, changes in local hydrological conditions, which had clear effects on sediment characteristics. In general, stakes resulted in local sediment deposition while racks produced local erosion, both leading to the reduction and eventual death of nearby seagrass beds.

Sensitivity assessment. Any changes in hydrology will have a considerable impact on the integrity of seagrass habitat. A change in water flow at the level of the benchmark of 10 to 20 cm/s for more than 1 year would cause some mortality in seagrasses resulting in a 'Medium' resistance score. Recovery will depend on the species capacities to adapt to changes in water flow regime but is considered to be fairly rapid. Resilience is thus assessed as 'Medium'. The biotope scores a 'Medium' sensitivity to changes in water flow at the pressure benchmark.

Emergence regime changes	Low Q: High A: High C: High	Medium Q: High A: Low C: Medium	Medium Q: High A: Low C: Medium

Seagrasses are generally not tolerant to exposure to aerial conditions, suggesting that the shallowest distribution should be at a depth below mean low water (MLW) (Koch, 2001). *Zostera noltei* grows predominantly in the intertidal zone and demonstrate higher resistance to desiccation than *Zostera marina* which occurs more frequently in the subtidal. To understand the differences in desiccation tolerance between the two *Zostera* species, Leuschner *et al.* (1998) investigated the photosynthetic activity of emerged plants. The study found that after 5 hours of exposure to air during low tide, leaves of *Zostera noltei* had lost up to 50% of their water content. Decreasing leaf water content resulted in a reversible reduction in light-saturated net photosynthesis rate of the plant. The experiment further showed that photosynthesis was more sensitive to desiccation in *Zostera marina* plants than in *Zostera noltei* under a given leaf water content. The experiment confirmed that *Zostera marina* is most susceptible to local changes in emergence regimes by being less tolerant to desiccation pressure.

Tolerances vary not only between species but also within species. For instance, annual and perennial forms of *Zostera marina* were observed to tolerate desiccation to different extents. Van Katwijk & Hermus (2000) noted that in intertidal areas of the Wadden Sea, annual *Zostera marina* plants tended to lie flat on the moist sediment when exposed at low tide. Perennial plants, on the other hand, had stiffer stems inhibiting contact with the sediment. These upright sheaths desiccate more rapidly when exposed. Morphology is, therefore, a factor partly determining tolerance to desiccation. The same phenomenon was observed by Boese *et al.* (2003) on *Zostera marina* in Aquinas Bay, USA.

The overall low tolerance of seagrass species to aerial exposure means that an increase in tidal amplitudes could force seagrass to grow deeper where there was less chance of exposure to the air. As the depth limit of seagrasses is set by light penetration, this change is likely to reduce the extent of suitable habitat. Changes in seagrass distribution along a depth gradient will have an impact further down the food chain.

Sensitivity assessment. Sensitivity to changes in emergence regimes varies between species and habitats. Species growing in intertidal habitats have greater tolerance to exposure to air than species inhabiting subtidal beds. The resistance of *Zostera marina* to this pressure is therefore assessed as 'Low'. Recovery will be enabled by recolonization from surrounding communities located further down the shore and via the remaining seed bank. Recovery is therefore considered to be fairly rapid resulting in a 'Medium' resilience score. The biotope is therefore considered to have a 'Medium' sensitivity to this pressure at the pressure benchmark.

Wave exposure changes (local)	Medium Q: Medium A: Medium C: Medium	Medium Q: High A: Low C: Medium	Medium Q: Medium A: Low C: Medium

An absolute wave exposure limit and maximum wave height for *Zostera* has not been established (Short *et al.*, 2002) but an increase in wave action can harm the plants in several ways. Seagrasses are not robust. Strong waves can cause mechanical damage to leaves and rhizomes. By losing above ground biomass due to increased wave action, the productivity of seagrass plants is limited. Small and patchy populations, as well as seedlings, will be particularly vulnerable to wave exposure as they lack extensive rhizome systems to effectively anchor the plant to the seabed.

Wave action also continuously mobilises sediments in coastal areas causing sediment re-suspension which in turn leads to a reduction in water transparency (Koch, 2001) (see 'changes in suspended sediments' pressure). Photosynthesis can be further limited by breaking waves inhibiting light penetration to the seafloor. Wave exposure can also influence the sediment grain size, with areas of high wave exposure having coarser sediments with lower nutrient concentrations. Coarser sediments reduce the vegetative spreading of seagrasses and inhibit seedling colonisation (Gray & Elliott, 2009). Changes in sediment type can, therefore, have wider implications for the sensitivity of the beds on a long-term scale.

Sensitivity assessment. No evidence was available to determine the impact of this pressure at the benchmark level. However, exposure models from Studland Bay and Salcombe, where seagrass beds are limited to low wave exposure, show that even a change of 3% is likely to influence the upper shore limits as well as beds living at the limits of their wave exposure tolerance (Rhodes *et al*., 2006; Jackson *et al*., 2013). Change in wave exposure will impact the upper limit of seagrass and thus influence its wider distribution. At the benchmark level, an increase in wave exposure is likely to remove surface vegetation and the majority of the root system causing some mortality. Resistance is thus assessed as ‘Medium’. Recovery will depend on the presence of adjacent seagrass beds and is considered to be fairly rapid scoring a ‘Medium’ resilience. The biotope, therefore, scores a ‘Medium’ sensitivity to changes in wave exposure at the pressure benchmark.

Chemical Pressures

	Resistance	Resilience	Sensitivity
Transition elements & organo-metal contamination	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

This pressure is **Not assessed** but evidence is presented where available.

Growth of *Zostera marina* was inhibited by 0.32 mg/l Cu and 10 mg/l Hg but Cd, Zn, Cr and Pb had measurable but less toxic effects (Williams *et al*., 1994). Davison & Hughes (1998) report that Hg, Ni and Pb reduce nitrogen fixation, which may affect viability. However, leaves and rhizomes accumulate heavy metals, especially in winter. Williams *et al.* (1994) did not observe any damage to *Zostera marina* in the field. *Zostera marina* is known to accumulate TBT but no detrimental effects were observed in the field (Williams *et al*., 1994). TBT contamination is likely to adversely affect grazing gastropods resulting in increased algal growth, reduced primary productivity and potential smothering of the biotope.

Hydrocarbon & PAH contamination	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

This pressure is **Not assessed** but evidence is presented where available.

Zostera marina may be partially protected from direct contact with oil due to its subtidal habitat. Healthy populations of *Zostera* can occur in the presence of long-term, low level, hydrocarbon effluent, for example in Milford Haven, Wales (Hiscock, 1987). The *Amoco Cadiz* oil spill off Roscoff caused *Zostera marina* leaves to blacken for 1-2 weeks but had little effect on growth, production or reproduction after the leaves were covered in oil for six hours (Jacobs, 1980). The *Amoco Cadiz* oil spill did, however, result in the virtual disappearance of Amphipods, Tanaidacea and Echinodermata from *Zostera marina* beds and caused a decrease in numbers of Gastropoda, sedentary Polychaeta and Bivalvia. The numbers of most groups returned to normal within a year except Echinoderms which recovered more slowly and amphipods which did not show any signs of recovery (Jacobs, 1980). Removal of oil intolerant gastropod grazers may result in smothering of seagrasses by epiphytes (Davison & Hughes, 1998). Jacobs (1980) noted a larger algal bloom than in previous years after the Amoco Cadiz spill in Roscoff, probably as a result in increased nutrients (from dead organisms and breakdown of oil) and the reduction of algal grazers. However, herbivores recolonized and the situation returned to 'normal' within a few months. The experimental treatment of *Zostera* sp. with crude oil and dispersants halted growth but had little effect on cover whereas pre-mixed oil and dispersant caused rapid death and significant decline in cover within one week suggesting that dispersant treatments should be avoided (Davison & Hughes, 1998).

Synthetic compound contamination	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

This pressure is **Not assessed** but evidence is presented where available.

Little information on the impacts of synthetic compounds on *Zostera* species is present in the literature. Triazine herbicides (e.g. Irgarol) inhibit photosynthesis and sublethal effects have been detected. Terrestrial herbicides may also damage seagrass beds. For example, the herbicide Atrazine is reported to cause growth inhibition and 50 % mortality in *Zostera marina* exposed to 100 ppb (ng/ l) Atrazine for 21 days (Delistraty & Hershner, 1984; Davison & Hughes, 1998). Bester (2000) noted a correlation between raised concentrations of 4 triazine herbicides and areas where *Zostera* plants had been lost. Chesworth *et al.* (2004) also noted that exposure to antifoulant herbicides Diuron and Irgarol 1051 alone or in mixtures resulted in reduced photosynthesis and growth in *Zostera marina*.

Radionuclide contamination	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

No evidence found.

Introduction of other substances	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

This pressure is **Not assessed**.

De-oxygenation	High Q: Medium A: Medium C: Medium	High Q: High A: Low C: Medium	Not sensitive Q: Medium A: Low C: Medium

The effects of oxygen concentration on the growth and survivability of *Zostera marina* are not reported in the literature. *Zostera* sp. leaves contain air spaces (lacunae). Oxygen is transported to the roots where it permeates into the sediment, resulting in an oxygenated microzone, enhancing the uptake of nitrogen. The presence of air spaces suggests that seagrass may be tolerant of low oxygen levels in the short-term, however, prolonged deoxygenation, especially if combined with low light penetration and hence reduced photosynthesis will have an adverse effect.

Epifaunal gastropods may be tolerant of hypoxic conditions, especially *Littorina littorea* and *Hydrobia ulvae*. Infaunal species are likely to be exposed to hypoxic conditions, especially at low tide when they can no longer irrigate their burrows e.g. *Arenicola marina* can survive for 9 days without oxygen (Hayward, 1994). Conversely, possibly since it occupies the top few centimetres of sediment, *Cerastoderma edule* may be adversely affected by anoxia and would probably be killed by exposure to 2 mg/l oxygen for a week (benchmark). Loss of grazers would result in unchecked growth of epiphytes and other algae which may smother *Zostera marina*.

Sensitivity assessment. De-oxygenation is not likely to adversely affect seagrass beds in areas of adequate light. The loss of grazing gastropods could result in smothering and potential reduction in the extent of the seagrass. At the level of the benchmark, both resistance and resilience are assessed as **'High'** (no impact to recover from). Overall, the biotope is therefore assessed as **'Not Sensitive'** to de-oxygenation at the pressure benchmark.

Nutrient enrichment	Medium Q: Medium A: Medium C: Medium	Medium Q: High A: Low C: Medium	Medium Q: Medium A: Low C: Medium

During the past several decades, important losses in seagrass meadows have been documented worldwide related to an increase in nutrient load. Seagrasses are typically found in low energy habitats such as estuaries, coastal embayments and lagoons with reduced tidal flushing where nutrient loads are both concentrated and frequent. A typical response to nutrient enrichment is a decline in seagrass populations in favour of macroalgae or phytoplankton (Baden *et al*., 2003). Nutrient enrichment, especially of nitrogen and phosphorus, can lead to eutrophication.

The mechanisms responsible for seagrass decline under eutrophication are complex and involve direct and indirect effects relating to changes in water quality, smothering by macroalgal blooms (Den Hartog & Phillips, 2000), and competition for light and nutrients with epiphytic microalgae and with phytoplankton (Nienhuis, 1996). In the Mondego estuary (Portugal), eutrophication triggered serious biological changes, which led to an overall increase in primary production and to a progressive replacement of seagrass *Zostera noltei* beds by coarser sediments and opportunistic macroalgae (Cardoso *et al*., 2004). Nutrients stimulate phytoplankton blooms that compete for nutrients but more importantly increase the turbidity and absorb light, reducing seagrass productivity (discussed in 'changes in suspended solids'). In general terms, algae are able to out-compete seagrasses for water column nutrients since they have a higher affinity for nitrogen (Touchette & Burkholder, 2000). Short and Burdick (1996) found that excessive nitrogen loading stimulated the proliferation of algal competitors that caused shading and thereby stressed *Zostera* plants. Many seagrasses have a positive response to nitrogen and/or phosphorous enrichment (Peralta *et al*., 2003), but excessive loads can inhibit seagrass growth and survival, not only indirectly through light reduction resulting from increased algal growth but also directly in terms of the physiology of the seagrass. Direct physiological responses include ammonium toxicity and water column nitrate inhibition through internal carbon limitation (Touchette & Burkholder, 2000). Indirect effects of nutrient enrichment can accelerate decreases in seagrass beds such as sediment re-suspension from seagrass loss (see pressure on 'changes in suspended solids'). Jones & Unsworth (2015) concluded that seagrass habitats in the British Isles were nutrient enriched, with nitrogen levels 75% higher than the global average for *Zostera marina*, yet phosphate-limited, and concluded that many beds in the vicinity of human populations were in a poor state.

Sensitivity assessment. The loss of seagrass beds worldwide has been attributed to nutrient enrichment, due in part to the likeliness of smothering by epiphytes, and the effects of reduced light penetration caused by eutrophication. For instance, a study by Greening & Janicki (2006) found that in Florida, the USA, recovery of seagrass beds was incomplete 20 years after nutrient enrichment caused an eutrophication event. Seagrass beds are regarded as highly intolerant or **'Low'** resistance to this pressure. However, the benchmark of this pressure (compliance with WFD 'good' status) allows for a 30% loss of intertidal seagrass beds under the WFD criteria for good status. Therefore, at the level of the benchmark resistance of seagrass beds to this pressure is assessed **'Medium'**. The resilience of seagrass beds this degree of impact is assessed as **'Medium'** and sensitivity is, therefore, assessed as **'Medium'.**

Organic enrichment	Medium Q: Medium A: Medium C: Medium	Medium Q: High A: Low C: Medium	Medium Q: Medium A: Low C: Medium

Organic enrichment may lead to eutrophication with adverse environmental effects including deoxygenation, algal blooms and changes in community structure (see 'nutrient enrichment' pressure). Evidence on the effects of organic enrichment on *Zostera* species is limited but abundant for other seagrass species.

Neverauskas (1987) investigated the effects of discharged digested sludge from a sewage treatment on *Posidonia* spp. and *Amphibolis* spp. in South Australia. Within 5 years the outfall had affected an area of approximately 1900 ha, 365 ha of which were completely denuded of seagrasses. The author suggests that the excessive growth of epiphytes on the leaves of seagrasses was a likely cause for reduced abundance. A subsequent study by Bryars and Neverauskas (2004) determined that 8 years after the cessation of sewage output, total seagrass cover was approximately 28% of its former extent. While these results suggest that seagrasses can return to a severely polluted site if the pollution source is removed, they also suggest that it will take many decades for the seagrass community to recover to its former state.

The effects of organic enrichment from fish farms were investigated on *Posidonia oceanica* seagrass beds in the Balearic Islands (Delgado *et al*., 1999). The fish culture had ceased in 1991; however, seagrass populations were still in decline at the time of sampling. The site closest to the former fish cages showed a marked reduction in shoot density, shoot size, underground biomass, sucrose concentration and photosynthetic capacities. The shoot also had high P-concentration in tissues and higher epiphyte biomass compared to the other sites. Since water conditions had recovered completely by the time of sampling, the authors suggest that the continuous seagrass decline was due to the excess organic matter remaining in the sediment (Delgado *et al*., 1999).

It should be noted that coastal marine sediments where seagrasses grow are often anoxic and highly reduced due to the high levels of organic matter and slow diffusion of oxygen from the water column to the sediment. Seagrasses worldwide have been shown to exhibit a three-way symbiotic relationship with the small lucinid bivalves (hatchet-shells, e.g. *Loripes* and *Lucinoma*) and their endosymbiotic sulfide-oxidizing gill bacteria (Van der Heide *et al*., 2012). In experiments, the sulfide-oxidizing gill bacteria of *Loripes lacteus* were shown to reduce sulfide levels in the sediment and enhance the productivity of *Zostera noltei*, while the oxygen released from the roots of *Zoster noltei* was of benefit to *Loripes*. Nevertheless, the negative effects of the experimental addition of sulphide were not fully prevented by the presence of *Loripes* (Van der Heide *et al*., 2012). Therefore, while seagrasses or the *Zostera*-lucinid symbiosis are adapted to these anoxic sediment conditions if the water column is organically enriched, plants are unable to maintain oxygen supply to the meristem and die fairly quickly. The enrichment of the water column could, therefore, significantly increase the sensitivity of seagrasses to this pressure. Worldwide evidence suggests that nutrient enrichment is one of the biggest threats to seagrass populations (Jones & Unsworth, 2015).

Sensitivity assessment. The organic enrichment of the marine environment increases turbidity and causes the enrichment of the sediment in organic matter and nutrients (Pergent *et al*., 1999). Evidence shows that seagrass beds found in proximity to a source of organic discharge were severely impacted with important losses of biomass. Although no study was found on the British species, the evidence suggests that *Zostera marina* will be negatively affected by organic enrichment. No evidence was found addressing the benchmark of this study. A deposition of 100 gC/m2/year is considerably lower than the amount of organic matter discharged by sewage outlets and fish farms. Therefore, resistance to this pressure is assessed as **'Medium'**, recovery as **'Medium'**, and sensitivity as **'Medium'**.

Physical Pressures

	Resistance	Resilience	Sensitivity
Physical loss (to land or freshwater habitat)	None Q: High A: High C: High	Very Low Q: High A: High C: High	High Q: High A: High C: High

All marine habitats and benthic species are considered to have a resistance of 'None' to this pressure and to be unable to recover from a permanent loss of habitat resulting in 'Very Low' resilience. Sensitivity within the direct spatial footprint of this pressure is, therefore 'High'. Although no specific evidence is described confidence in this assessment is 'High', due to the incontrovertible nature of this pressure. Adjacent habitats and species populations may be indirectly affected where meta-population dynamics and trophic networks are disrupted and where the flow of resources e.g. sediments, prey items, loss of nursery habitat etc. is altered.

	Resistance	Resilience	Sensitivity
Physical change (to another seabed type)	None Q: High A: High C: High	Very Low Q: High A: High C: High	High Q: High A: High C: High

A change to another seabed type (from sediment to hard rock) will result in a permanent loss of suitable habitat for seagrass species. Resistance is thus assessed as 'None'. As this pressure represents a permanent change, recovery is impossible as a suitable substratum for seagrasses is lacking. Consequently, resilience is assessed as 'Very low'. The habitat, therefore, scores a 'High' sensitivity. Although no specific evidence is described confidence in this assessment is 'High', due to the incontrovertible nature of this pressure.

	Resistance	Resilience	Sensitivity
Physical change (to another sediment type)	Low Q: High A: High C: High	Very Low Q: High A: High C: High	High Q: High A: High C: High

Seagrass beds occur almost exclusively in shallow and sheltered coastal waters anchored in sandy and muddy bottoms. Coarser sediments reduce the vegetative spreading of seagrasses and inhibit seedling colonization (Gray & Elliott, 2009). Changes in sediment type can, therefore, have wider implications on the distribution of seagrass beds. Hence, change towards a coarser sediment type would inhibit seagrasses from becoming established due to a lack of adequate anchoring substratum. A more mud dominated habitat, on the other hand, could increase sediment re-suspension and exclude seagrasses due to unfavourable light conditions.

Sensitivity assessment. The resistance was assessed as 'Low'. As this pressure represents a permanent change, recovery is impossible without intervention as a suitable substratum for seagrasses is lacking. Consequently, resilience is assessed as 'Very low'. The habitat, therefore, scores a 'High' sensitivity. Although no specific evidence is described confidence in this assessment is 'High', due to the incontrovertible nature of this pressure.

Habitat structure changes - removal of substratum (extraction)	None Q: High A: High C: High	Very Low Q: High A: Low C: Medium	High Q: High A: Low C: Medium

The extraction of sediments to 30 cm (the benchmark) will result in the removal of every component of seagrass beds. Roots and rhizomes are buried no deeper than 20 cm below the surface (see ‘abrasion’ and ‘penetration and/or disturbance of the substratum below the surface of the seabed’ pressures). Resistance is therefore assessed as ‘None’ and resilience is considered ‘Very Low’ resulting in a ‘High’ sensitivity score. The confidence assessment for this pressure is high as it is based on the characteristics of the pressure i.e. complete removal of the feature within the pressure footprint.

Abrasion/disturbance of the surface of the substratum or seabed	Low Q: High A: High C: Medium	Medium Q: High A: Low C: Medium	Medium Q: High A: Low C: Medium

Seagrasses are not physically robust. The leaves and stems of seagrass plants rise above the surface and the roots are shallowly buried so that they are vulnerable to surface abrasion. Activities such as trampling, anchoring, power boating and potting are likely to remove leaves and damage rhizomes. The removal of above-ground biomass would result in a loss of productivity whilst the removal of roots would cause the death of plants. Seagrasses are limited to shallow, protected waters and soft sediments. These areas are often open to public access and are widely used in commercial and recreational activities. Evidence for abrasion impacts is summarised below for activities that give rise to this pressure.

Trampling: human wading in shallow coastal waters is a common activity that inherently involves trampling of the substratum. Trampling may be caused by recreational activities such as walking, horse-riding and off-road driving. These activities are likely to damage rhizomes and cause seeds to be buried too deeply to germinate (Fonseca, 1992). Negative effects of human trampling on seagrass cover, shoot density, and rhizome biomass, have been reported by Eckrich & Holmquist (2000) for the seagrass *Thalassia testudinum*. The study found that recovery occurred within a period of seven months after trampling ceased but the reduced cover was still visually distinguishable 14 months after the experiment. A study by Major *et al.* (2004) found that trampling impact varied depending on substratum type. A significant decrease in shoot density as a result from trampling was only observed at a site with soft muddy substratum with no impact detected on the hard packed sand substratum. Damage from trampling is thus dependent on the substratum type with seagrass beds growing on soft substrata being most vulnerable to this pressure.

Boating activities: boats passing in close proximity to seagrass beds can create waves. Turbulence from propeller wash and boat wakes can resuspend sediments, break off leaves, dislodge sediments and uproot plants. The re-suspension of sediments is further assessed in ‘changes in suspended sediment’ pressure. Koch (2002) established that physical damage from boat wakes was greatest at low tide but concluded that negative impacts of boat-generated waves were marginal on seagrass habitats. The physical impact of the engine’s propellers, shearing of leaves and cutting into the bottom, can also have damaging effects on seagrass communities. In severe cases, propellers cutting into the bottom may completely denude an area resulting in narrow dredged channels through the vegetation called propeller scars. Scars might expand and merge to form larger denuded areas. A study in Florida looking at the seagrasses *Thalassia* *testudinum*, *Syringodium filiforme* and *Halodule wrightei* determined that recovery of seagrass to propeller impact depend on species (Kenworthy *et al.*, 2002). For *Syringodium filiforme* recovery was estimated at 1.4 years and for *Halodule wrightei* at 1.7 years, whilst recovery for *Thalassia testudinum* was estimated to require 9.5 years. Variations in recovery time were explained by different growth rates. However, it is not appropriate to assume that recovery rates are similar from one geographical or climatic region to another and more in-depth research is needed for *Zostera* species around the British Isles.

Potting: static gear is commonly deployed in areas where seagrass beds are found, either in the form of pots or as bottom set gill or trammel nets. Damage can be caused during the setting of pots or nets and their associated ground lines and anchors, by their movement over the bottom during rough weather and during recovery. Whilst the potential for damage is lower per unit deployment compared to towed gear (see 'penetration and/or disturbance of the substratum below the surface of the seabed' pressure), there is a risk of cumulative damage if use is intensive. Hall *et al.* (2008) categorized seagrass beds as being highly sensitive to high intensities of potting (pots lifted daily, with a density of over 5 pots per ha) and medium sensitive to lower levels (pots lifted daily, less than 4 pots per ha). However, no direct evidence was found to confirm these estimates.

Grazing: Nacken & Reise (2000) investigated physical disturbance caused by Brent geese (*Branta b. bernicla*) and widgeon (*Anas penelope*) feeding on *Zostera noltei* in the northern Wadden Sea. To graze on leaves and shoots above the sediment and on rhizomes and roots below, birds reworked the entire upper 1 cm layer of sediment and excavated pits by trampling. As a result, birds pitted 12% of the seagrass bed and removed 63% of plant biomass. Plants recovered by the following year with the authors suggesting that seasonal erosion caused by herbivorous wildfowl was necessary for the persistence of *Zostera noltei* beds (Nacken & Reise, 2000). Similarly, Tubbs & Tubbs (1982, 1983; see Davison & Hughes, 1998) suggested that *Zostera* sp. can rapidly recover from 'normal' levels of wildfowl grazing. Physical disturbance may, however, be detrimental to seagrass beds as soon as the ‘normal’ level caused by grazing birds is exceeded by human activities. In addition, geese and wigeon do not dive so that shoots below the reach of their necks at low tide are 'safe' from grazing pressure.

Experimental: Boese *et al.* (2009) examined the recolonization of experimentally created gaps within intertidal perennial and annual *Zostera marina* beds in the Yaquina River Estuary, USA. The experiment looked at two zones, the lower intertidal almost continuous seagrass and an upper intertidal transition zone where there were patches of perennial and annual *Zostera marina*. The study found that recovery began within a month after a disturbance in the lower intertidal continuous perennial beds and was complete after two years, whereas, plots in the transition zone took almost twice as long to recover.

Sensitivity assessment. In summary, a wide range of activities gives rise to this pressure with intertidal habitat being more exposed as they are more readily accessible than subtidal beds. The resilience and recovery of seagrass beds to abrasion of the seabed surface depends on the frequency, persistence and extent of the disturbance. Factors such as the size and shape of the impact will also influence the sensitivity of seagrass. There is also considerable evidence that the type of substratum plays a role in determining the magnitude of impact. Soft and muddy substratum is thought to be more easily damaged than harder more compact ground. Finally, temporal effects should also be taken into account. The state of the tide will influence the magnitude of damage as will seasonal effects with damage in winter likely to have less impact than the damage that occurs during the growing season. Overall, studies suggest little resistance to abrasion resulting in an assessment of **‘Low’** resistance. Physical disturbance and removal of plants can lead to increased patchiness and destabilisation of the seagrass bed, which in turn can lead to reduced sedimentation within the seagrass bed, increased erosion, and loss of larger areas of plants (Davison & Hughes, 1998). Recovery will, however, be fairly rapid and resilience is assessed as **'Medium'**. Therefore, sensitivity is assessed as **'Medium'** to this pressure.

Penetration or disturbance of the substratum subsurface	None	Low	High
	Q: High A: Medium C: Medium	Q: High A: Medium C: Medium	Q: High A: Medium C: Medium

Seagrass species are vulnerable to physical damage. The leaves and stems of seagrass plants rise above the surface and the roots are shallowly buried. Activities such as digging and raking for clams, anchoring and mooring will penetrate the substratum to an average depth of 5 cm removing plant biomass above and below ground. Abrasion to the substratum to a depth greater than 5 cm will directly impact seagrass habitats and all biomass (leaves, rhizomes) will be completely removed leading to the death of the plant in the area impacted. Seagrass beds are often associated with commercially important bivalves. Fisheries targeting these species are therefore likely to impact seagrass habitats and are the most widespread (and best studied) activities giving rise to this pressure on this habitat. The extent of the damage on seagrass beds depends on the activity.

Clam digging and clam raking: Boese (2002) investigated the effects of manual clam harvesting on *Zostera marina* by raking and digging for clams in experimental plots in Yaquina Bay, USA. After three monthly treatments, measures of biomass, primary production (leaf elongation), and percent cover were compared between disturbed and undisturbed plots. The study found that clam raking treatments visibly removed large numbers of seagrass leaves and some below-ground rhizomes. However, two weeks after the end of the experiment, no statistical difference in percentage cover was observed between disturbed and control plots indicating a fast recovery rate. Clam digging, on the other hand, caused visual differences in percentage cover for 10 months after the end of the experiment, although differences were not statistically significant. Boese (2002) concluded that recreational clamming is unlikely to have a major impact on seagrass beds in the Yaquina estuary. The author calls, however, to view the results with caution as multi-year disturbances were not investigated and differences in sediment characteristics are likely to influence the resistance and resilience of seagrasses to this pressure. Similarly, Peterson *et al.* (1987) found that hand raking and moderate clam-kicking (a commercial harvesting method in which propeller wash is used to dislodge hard clams) resulted in a reduction in *Zostera marina* biomass by approximately 25%. No differences between control and experimental areas were apparent one year after the experiment. However, at a higher intensity, clam-kicking reduced seagrass biomass to about half of control levels and recovery remained incomplete four years after the end of the experiment (Peterson *et al.*, 1987).

Anchoring and mooring: an anchor landing on a patch of seagrass can bend, damage and break seagrass shoots (Montefalcone et al., 2006) and an anchor being dragged as the boat moves driven by wind or tide causes abrasion of the seabed. Milazzo *et al.* (2004) found that the extent of damage depended on the type of anchor with the folding grapnel having the greatest impact. The study further determined that heavier anchors (often associated with larger boats) will sink deeper into the substratum and thereby causing greater damage. A technical paper by Collins *et al.* (2010) using SCUBA divers found bare patches (typically 1–4 m2) were caused by anchoring by leisure boats in Studland Bay, UK. The study further determined that average shear vane stress was significantly higher in intact seagrass beds compared to scars indicating a less cohesive and more mobile substratum caused by anchors. Axelsson *et al.* (2012) also investigated anchor damage in Studland Bay. The study did not provide consistent evidence of boat anchoring impacting the seagrass habitat in this location. The study did, however, observe higher shoot density and percentage cover of seagrass in a voluntary anchor zone compared to a control area where anchoring occurred. The authors recommended longer monitoring in order to determine whether the trend was caused by natural variations or the effects of anchor exclusion. Traditional mooring further contributes to the degradation of seagrass habitats. A traditional swing mooring is a buoy on a chain attached to a static anchoring block fixed on the seabed, to buffer any direct force on the permanent block, the chain lies on the seabed where it moves around with wind and tides, as the chain pivots on the block it scours the seabed. In proximity to seagrass beds, the chain usually removes not only the seagrass above ground parts such as leaves and shoots but also the roots anchored in the sediment. Further sediment abrasion may occur in the vicinity to the anchoring blocks due to eddying of currents. The blocks themselves may increase the competition of seagrass with other algae as they provide ideal settlement surfaces. Boats might also moor on intertidal sediments. When the tide goes out, the boat sits directly on top of the soft sediment. Walker *et al.* (1989) found that boat moorings caused circular or semi-circular depressions of bare sand within seagrass beds between 3 to 300 m2 causing important habitat fragmentation. The scours created by moorings in the seagrass canopy interfere with the physical integrity of the meadow. Though relatively small areas of seagrass are damaged by moorings, the effect is much greater than if an equivalent area was lost from the edge of a meadow. Such mooring scars have been observed for Zostera marina around the UK such as in Porth Dinllaen in the Pen Llyna'r Sarnau Special Area of Conservation, Wales (Egerton, 2011) and at Studland Bay (Jackson *et al.*, 2013).

Trawling: bottom trawling and dragging are industrial fishing methods which scour the seabed to collect target species. Neckles et al. (2005) investigated the effects of trawling for the blue mussels Mytilus edulis on *Zostera marina* beds in Maquoit Bay, USA. Impacted sites ranged from 3.4 to 31.8 ha in size and were characterized by the removal of above- and belowground plant material from the majority of the bottom. The study found that one year after the last trawl, *Zostera marina* shoot density, shoot height and total biomass averaged respectively to 2-3%, 46-61% and < 1% that of the reference sites. Substantial differences in *Zostera marina* biomass persisted between disturbed and reference sites up to 7 years after trawling. Rates of recovery depended on initial fishing intensity but the authors estimated that an average of 10.6 years was required for *Zostera marina* shoot density to match pre-trawling standards.

Dredging and suction dredging: the effects of dredging for scallops on *Zostera marina* beds were investigated by Fonseca et al. (1984) in Nova Scotia, USA. Dredging was carried out when *Zostera marina* was in its vegetative stage on hard sand and on soft mud substrata. The damage was assessed by analysing the effects of scallop harvesting on seagrass foliar dry weight and on the number of shoots. Lower levels of dredging (15 dredges) had a different impact depending on substrata, with the hard bottom retaining a significantly greater overall biomass than the soft bottom. However, an increase in dredging effort (30 dredges) led to a significant reduction in *Zostera marina* biomass and shoot number on both hard and soft bottoms. Solway Firth is a British example of the detrimental effects of dredging on seagrass habitats. In the area, where harvesting for cockles by hand is a traditional practice, suction dredging was introduced in the 1980s to increase the yield. A study by Perkins (1988) found that where suction dredging occurred, the sediment was smoothened and characterized by a total absence of *Zostera* plants. The study concluded that the fishery was causing widespread damage and could even completely eradicate *Zostera* from affected areas. Due to concerns over the sustainability of this fishing activity, the impacts on cockle and *Zostera* stocks, and the effects on overwintering wildfowl, the fishery was closed to all forms of mechanical harvesting in 1994.

Sensitivity assessment. The deployment of fishing gears on seagrass beds results in physical damage to the above surface part of the plants as well as to the root systems. Seagrasses do not have an avoidance mechanism; resistance to this pressure is therefore assessed as 'None'. The recovery of seagrass beds after disturbance to the sub-surface of the sediment will be slow with the speed depending on the extent of removal. Rates may be accelerated where adjacent seed sources and viable seagrass beds are present but can be considerably longer where rhizomes and seed banks were removed. Using a model simulation, it has been suggested that with favourable environmental conditions, seagrass beds might recover from dragging disturbance in 6 years but, conversely, recovery under conditions less favourable to seagrass growth could require 20 years or longer (Neckles *et al.*, 2005). Resilience is thus assessed as 'Low'. The mechanical harvest of shellfish damaging the sub-surface of the sediments poses a very severe threat to seagrass habitats, yielding a 'High' sensitivity score.

Changes in suspended solids (water clarity)	Low Q: High A: High C: High	Low Q: High A: Low C: Medium	High Q: High A: Low C: Medium

Irradiance decreases exponentially with increasing depth, and the suspended sediment concentration has a direct linear effect on light attenuation (van Duin *et al.*, 2001). Changes in suspended solids will thus reduce the light available for seagrass plants necessary for photosynthesis. Impaired productivity due to a decrease in photosynthesis will affect the growth and reproductive abilities of plants. Turbidity also results in a reduction of the amount of oxygen available for respiration by the roots and rhizomes thus lowering nutrient uptake. The resulting hypoxic conditions will lead to a build-up of sulphides and ammonium, which can be toxic to seagrass at high concentrations (Mateo *et al.*, 2006). Giesen *et al.* (1990a,b; Davison & Hughes, 1998) suggested that considerable declines in seagrass populations in the Wadden Sea were related to increases in turbidity from dredging and deposit extraction.

Water clarity is a vital component for seagrass beds as it determines the depth-penetration of photosynthetically active radiation of sunlight. Seagrasses have light requirements an order of magnitude higher than other marine macrophytes making water clarity a primary factor in determining the maximum depth at which seagrasses can occur. The critical threshold of light requirements varies among species ranging from 2% in-water irradiance for *Zostera noltei*, to 11 to 37% for *Zostera marina* (Erftemeijer & Robin, 2006). These differences in the light requirement for *Zostera* are reflected by the position of species along a depth gradient with *Zostera noltei* occurring predominantly in the intertidal and *Zostera marina* found at greater depth in the subtidal. However, differences in light requirements also vary within species. For example, the minimum light requirement for *Zostera marina* in a Danish embayment was 11% in-water irradiance, whereas the estimated light requirement for the same species in the Netherlands was 29.4% in-water irradiance (Olesen, 1993). This variability within species is likely attributed to photo-acclimation to local light regimes. In a six month long experiment in the Dutch Wadden Sea, Philippart (1995) found that shading induced a 30% decrease in the leaf growth rate, a 3-fold increase in the leaf loss rate, and an 80% reduction in the total biomass of *Zostera noltei*. The decreasing growth rate is most probably due to reduced photosynthesis caused by shading. The increased leaf loss may have been the result of enhanced deterioration of leaf material under low light conditions. The study also established that during the summer period, the maximum biomass of *Zostera noltei* under the control light conditions was almost 10 times higher than those under the low light conditions (incident light reduced to 45% of natural light conditions). The summer is a critical period for maintenance and growth of vegetative shoots. The effects of shading may, therefore, be most severe during the summer months. A similar response to reduced light availability for *Zostera* *marina* was observed by Moore & Wetzel (2000).

Increases in turbidity over a prolonged period of time are therefore highly likely to impact seagrass species. Sensitivity will depend on individual seagrass beds. Older, more established perennial meadows have greater carbohydrate reserves and are thus more able to resist changes in light penetration than annual plants (Alcoverro *et al*., 2001). Seagrass plants found in clear waters may be able to tolerate sporadic high turbidity (Newell & Koch, 2004). However, where seagrass beds are already exposed to low light conditions, then losses may result from even short-term events (Williams, 1988). The growth of both *Zostera marina* and its associated epiphytes are reduced by increased shading due to turbidity (reduction of light penetration by 42, 28 and 9%). Backman & Barilotti (1976) further established that intensive shading (reduction of light penetration by 63%) inhibited flowering in *Zostera marina* plants.

Sensitivity assessment. Turbidity is an important factor controlling production and ultimately survival and recruitment of seagrasses. Seagrass populations are likely to survive short-term increases in turbidity, however, a prolonged increase in light attenuation, especially at the lower depths of its distribution, will probably result in loss or damage of the population. Therefore, resistance is assessed as **'Low'**. A loss of seagrass beds will promote the re-suspension of sediments, making recovery unlikely as seagrass beds are required to initially stabilise the sediment and reduce turbidity levels (Van der Heide *et al*., 2007). A high turbidity state appears to be a highly resilient alternative stable state; hence return to the seagrass biotope is unlikely resulting in **'Low'** resilience. *Zostera marina* should be considered intolerant of any activity that changes the sediment regime where the change is greater than expected due to natural events, and sensitivity is assessed as **'High'**.

Smothering and siltation rate changes (light)	Low Q: Medium A: Medium C: Medium	Medium Q: High A: Medium C: Medium	Medium Q: Medium A: Medium C: Medium

Several studies have documented the deterioration of seagrass meadows by smothering due to excessive sedimentation. The consequences of enhanced sedimentation for seagrass beds depend on several factors such as the life history stage as well as the depth and timing of burial.

Early life stages of seagrass, smaller in size than adult plants, are most vulnerable to this pressure as even a small load of added sediment will lead to the complete burial. Wang *et al.* (2016) reported that seed germination and seedling establishment varied with burial depth and sediment type, in *Zostera marina* from Lidao Bay, Shandong Peninsula, China. High germination rates (76-90%) were found in seeds on the sediment surface or buried to 1 cm after seven weeks in the laboratory, regardless of sediment type but germination rates fell to below 40% with increasing burial. Only those seeds in a sand:silt mixture (2:1) were able to germinate when buried to 5 cm. seedling establishment was similar and seeds buried to shallow depths had the highest establishment rates. A maximum of seedling establishment of ca 30% was recorded buried to 1 cm in sand:silt (2:1) (Wang *et al*., 2016). Wang *et al.* (2016) also reported that the burial limit of *Zostera marina* seeds varied between 1 and 9 cm in various studies but noted that survival of seeds probably depended on a range of factors including depth, sediment type and oxygen levels in the sediment.

Vermaat *et al.* (1997) found that adult *Zostera marina* in the Dutch Wadden Sea was able to cope with sedimentation rates between 2 and 13 cm per year as the plant has the capacity to elongate vertical stems enabling it to raise the leaf canopy above the sediment load. However, a study in the USA observed over 50% mortality of plants of *Zostera marina* in field burial treatments of 4 cm (corresponding to 25% of plant height) for 24 days (Mills & Fonseca, 2003). Plants buried 75% or more of their height (16 cm) experienced 100% mortality, which indicated a low resistance of *Zostera marina* to burial. Munkes *et al.* (2015) noted that the *Zostera marina* in the Mill & Fonseca (2003) study were smaller than the normal ca 50 cm leaf length. Munke *et al.* (2015) noted that the effect of burial depended on the actual leaf length rather than species size. In their experimental field study in Kiel Bight, Munke *et al.* (2015) found negative effects on shoot mortality, delayed growth and flowering and reduced carbohydrate storage even after burial under the sand at 5 cm (ca 10% of plant height) and four weeks, the lowest burial depth and shortest duration examined. The effects were significant enough to affect the next year's growth. Burial by greater than 5 cm resulted in shoot mortality but burial by 5 cm reduced maximum biomass, leaf length, starch storage and flowering capacity. Burial by 20 cm (ca 40% of plant height) resulted in high shoot mortality (ca 97%) after 10 weeks. Munke *et al.* (2015) also noted that their experimental seagrass meadow was in good condition and that stressed populations could exhibit more adverse effects.

The timing of the siltation event also plays a role in particular for intertidal beds. At low tide, the seagrass bed is exposed with plants lying flat on the substratum. The addition of material would immediately smother the entire plant and have a greater impact on leaves and stem than if added on plants standing upright. The resistance of intertidal beds to this pressure may thus vary with time of day.

Sensitivity assessment. Above studies suggest that *Zostera marina* is not resistant to smothering with some variation between the critical threshold depths of burial and sediment type. All studies, however, indicate that at the level of the benchmark (5 cm of fine material added to the seabed) some mortality and loss of biomass may occur. Therefore, resistance is assessed as **'Low'**. Some plants will survive by successfully relocating rhizomes closer to the sediment surface. With the benchmark set at 'material added to the seabed in a single event', the sensitivity will be greater than if burial occurred in a continuous way. In addition, seagrass beds are restricted to low energy environments, suggesting that once the silt is deposited, it will remain in place for a long period of time so habitat conditions will not reduce exposure. Resilience is therefore assessed as **'Medium'** and sensitivity as **'Medium'** siltation at the pressure benchmark.

Smothering and siltation rate changes (heavy)	None Q: High A: High C: High	Very Low Q: High A: Low C: Medium	High Q: High A: Low C: Medium

Zostera marina is intolerant of smothering by excessive siltation (see above). Seagrasses can cope with small rates of sedimentation by relocating their rhizomes closer to the sediment surface (Vermaat *et al.*, 1997). Mills & Fonseca (2003) however observed 100% mortality in *Zostera marina* plants buried at a depth of 16 cm. Burial by 20 cm (ca 40% of plant height) resulted in high shoot mortality (ca 97%) after 10 weeks (Munke *et al.*, 2015).

Resistance to sedimentation at the pressure benchmark (30 cm of added material) is therefore assessed as **'None'** as all individuals exposed to siltation are predicted to die and consequent resilience as **'Low'** to 'Very Low'. In addition, seagrass beds are restricted to low energy environments, suggesting that once the silt is deposited, it will remain in place for a long period of time so habitat conditions will not reduce exposure. Sensitivity based on combined resistance and resilience is therefore assessed as **'High'.**

Litter	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

Not assessed

Electromagnetic changes	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

No evidence

Underwater noise changes	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

Species characterizing this habitat do not have hearing perception but vibrations may cause an impact, however no studies exist to support an assessment

Introduction of light or shading	Low Q: Low A: NR C: NR	Low Q: Low A: NR C: NR	High Q: Low A: Low C: Low

An increase in light might be beneficial while shading by artificial structures will decrease incident light and hence reduce photosynthesis and growth rates. For example, in mesocosm experiments, Frederick *et al.* (1995) noted that shading (at 11, 21, 41, 61, and 94% of incident surface light for one week) resulted in a reduction in shoot density and an increase in shoot height. But shading alone did not cause mortality in the experimental time frame. Holmer & Laursen (2002) noted that shading affected *Zostera marina* from a low-light, organic rich sediment population more than light saturated, low-organic sediment population. However, the effects were significant in spring but not in autumn, and were also related to the plant's ability to tolerant anoxic and sulfidic conditions.

Overall, there is little evidence of seagrass mortality resulting from shading directly but the effects of shading and smothering from epiphytes and macroalgae are discussed under nutrient enrichment and the effects of light attenuation under 'water clarity' above. However, the effects of shading could mirror those of reduced water clarity (increased turbidity) depending on the scale of the artificial structure. Therefore, a resistance of 'Low', with a resilience of 'Low' and sensitivity of 'High' is suggested, albeit with low confidence.

Barrier to species movement	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

Not relevant–this pressure is considered applicable to mobile species, e.g. fish and marine mammals rather than seabed habitats. Physical and hydrographic barriers may limit the dispersal of seed. But seed dispersal is not considered under the pressure definition and benchmark.

Death or injury by collision	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

Not relevant to seabed habitats. NB. Collision by grounding vessels is addressed under ‘surface abrasion’.

Visual disturbance	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

Not relevant

Biological Pressures

	Resistance	Resilience	Sensitivity
Genetic modification & translocation of indigenous species	High Q: Low A: NR C: NR	High Q: Low A: NR C: NR	Not sensitive Q: Low A: NR C: NR

Translocation of seagrass seeds, rhizomes and seedlings is a common practice globally to counter the trend of decline of seagrass beds. However, Williams & Davis (1996) found that levels of genetic diversity of restored *Zostera marina* beds in Baja California, USA, were significantly lower than in natural populations. A subsequent study by Williams (2001) determined that the observed genetic bottleneck was a consequence of the collection protocol of source material (i.e. founder effect). Founder effects are likely to occur if seeds used to revegetate restoration sites are collected from a limited number of sources. Similar to episodes of colonization, the ‘founding’ propagules can represent only a portion of the genetic diversity present in the source populations, and they might hybridize with local genotypes (Hufford & Mazer, 2003). The loss of genetic variation can lead to lower rates of seed germination and fewer reproductive shoots, suggesting that there might be long-term detrimental effects for population fitness. Williams (2001) affirms that genetic variation is essential in determining the potential of seagrass to rapidly adapt to a changing environment. Transplanted populations are therefore more sensitive to external stressors such as eutrophication and habitat fragmentation, with a markedly reduced community resilience than natural populations (Hughes & Stachowicz, 2004).

Translocation also has the potential to transport pathogens to uninfected areas (see 'introduction of microbial pathogens' pressure). The sensitivity of the ‘donor’ population to harvesting to supply stock for translocation is assessed for the pressure ‘removal of target species’. No evidence was found for the impacts of translocated beds on adjacent natural seagrass beds. However, it has been suggested that translocation of plants and propagules may lead to hybridization with local wild populations. If this leads to loss of genetic variation there may be long-term effects on the potential to adapt to changing environments and other stressors.

Sensitivity assessment. Presently, there is no evidence of loss of habitat due to genetic modification and translocation of seagrass species, resistance and resilience to this pressure are thus considered to be ‘High’ (no impact to recover from). Overall the biotope is therefore 'Not Sensitive' to this pressure. However, if hybridization occurred, recovery would not be considered possible unless the population is eradicated and replaced. In this case, resilience is thus deemed ‘Very Low’ resulting in an overall ‘Low’ sensitivity score. As there is no direct evidence to support assessments, these are based on expert judgement.

Introduction or spread of invasive non-indigenous species	Low Q: Medium A: Medium C: Medium	Low Q: High A: Low C: Medium	High Q: Medium A: Low C: Medium

The effects on native species on seagrass species were reviewed by d'Avack *et al.* (2014). The review reported several non-native invasive plants as well as invertebrate species negatively impacting British seagrass beds. The (potential) impact of each invasive non-indigenous species (INIS) is reported below.

Non-native invasive plants: among the INIS currently present in the UK, the large brown seaweed *Sargassum muticum* has the most direct impact on *Zostera* species. Druehl (1973) was the first to raise concern about the potential negative effects of *S. muticum* on *Zostera* beds in British waters. *Zostera* and *Sargassum muticum* were thought to be spatially separated due to their preferred habitat. *Zostera* species grow on sand and muddy bottoms, whereas *Sargassum muticum* attaches to solid substratum. However, when the seabed consists of a mixed substratum of sand, gravel and stones both species may occur together. Even though there are no indications of direct competition between the two species (Den Hartog, 1997), *Sargassum muticum* establishes itself within seagrass habitats where beds are retreating due to natural or anthropogenic causes. The invasive seaweed almost immediately occupies the empty spaces thereby interfering with the natural regeneration cycle of the bed. In addition, a study in Salcombe, SW England by Tweedley *et al.* (2008) demonstrated that the presence of *Zostera marina* may help the attachment of *Sargassum muticum* on soft substrata by trapping drifting fragments thereby allowing viable algae spores to settle on the seagrass matrix in an otherwise unfavourable environment. Once the invasive seaweed establishes itself, *Zostera marina* is unable to regain the lost territory indicating that eventually, *Sargassum muticum* is able to replace seagrass beds particularly on mixed substratum (Den Hartog, 1997).

The cord grass *Spartina anglica* is non-native grass, which was recorded to have negative effects on seagrass beds. This hybrid species of native (*Spartina alterniflora*) and an introduced cord grass species (*Spartina maritima*) colonizes the upper part of mud flats, where due to its extensive root system, it effectively traps and retains sediments. *Spartina anglica* has rapidly colonized mudflats in England and Wales due to its fast growth rate and high fecundity. Deliberate planting to stabilise sediments accelerated its spread throughout Britain (Hubbard & Stebbings, 1967). By consolidating the sediments the plant is responsible for raising mud flats as well as reducing sediment availability elsewhere. Butcher (1934) raised concerns that its pioneering consolidation may result in the removal of sediments from *Zostera* beds. Declines in *Zostera noltei* due to the encroachment of *Spartina anglica* were observed in Lindisfarne National Reserve in north-east England (Percival *et al*., 1998). The reduction in *Zostera noltei* beds had a direct impact on wildfowl populations as the food availability for the wildfowl was reduced on the top of the shore. This pressure will affect the upper limits of the intertidal rather than subtidal biotopes.

The invasive green algae *Codium fragile* ssp. *tomentosoides*, now found throughout Britain has been reported to occur in habitats dominated by *Zostera marina* (Gabary *et al*.,1997). It was initially thought that *Zostera* out-competes *Codium* at high *Zostera* densities (Malinowski & Ramus, 1973). But a study by Gabary *et al.* (2004) in Canada found that the invasive alga has morphological adaptations that allow it to compete with *Zostera* even in healthy eelgrass beds. *Codium fragile* ssp. *tomentosoides* have a wide salinity tolerance 12 to 40 ppt and are thus a concern to biotopes in full as well as in reduced salinity. However, direct ecological impacts remain unknown and no quantitative evidence is available to assess resistance at the benchmark.

Non-native invasive invertebrates: benthic macroinvertebrates can have a significant impact on seagrass beds, by either influencing abundance through seed herbivory (Fishman & Orth, 1996) or by influencing seed germination and seedling development by affecting vertical distribution of seeds. Some species have a positive effect by burying seeds to shallow depths and thereby reducing seed predation and facilitating seed germination whilst other species bury seeds too deep to allow germination. The invasive polychaete *Marenzelleria viridis*, a species naturally occurring on the east coast of North America but introduced Europe via transport in ballast waters, was recorded to directly impact seed banks of *Zostera marina* beds in its new territory (Delefosse & Kristensen, 2012). The study carried out on the island of Fyn, Denmark, determined that the impact of *Marenzelleria viridis* on seagrass beds depended on the abundance of worms within a bed. Negative effects were only observed at high abundances (1600 individual per m^2) causing seeds to be buried too deep to germinate. However, the study by Delefosse & Kristensen (2012) is the only publication on the impact of this particular invasive species on seagrass beds, and more evidence is needed in order to determine the ecological implications of this introduced polychaete in UK waters.

The invasive tunicate *Didemnum vexillum* has been reported growing on stalks and blades of *Zostera marina* plants in New England, USA (Carman & Grunden, 2010). The ecological effects of invasive tunicates introduced to seagrass beds remain unassessed, but in general terms, introduced epibionts have been shown to have negative effects on marine flora (Williams, 2007). Their considerable weight combined with their rapid asexual and sexual reproduction and an absence of predators (Carman *et al*., 2009) make them a considerable threat to marine plant communities as they increase the risk of smothering. The absence of predators could be related to anti-fouling microbial compounds present in *Didemnum vexillum (*Tait *et al*., 2007). Although the direct effect of invasive tunicates on seagrass remains unknown and no records of *Didemnum vexillum* growing on *Zostera* plants in the UK exist yet, there are concerns about possible negative interactions. No quantitative evidence regarding the level of impact has been found to assess this pressure.

Other invasive species could affect seagrass beds via indirect pathways. For instance, the Atlantic oyster drill *Urosalpinx cinerea*, a small predatory sea snail is unlikely to have a direct effect on seagrass beds but by preying on mussels and other bivalves, the sea snail could be responsible for a drop in water clarity which in turn will affect *Zostera* species (see sections below on changes in suspended solids). The invasive Pacific oyster *Magallana gigas* can also have negative effects. Oysters physically alter their environment by increasing habitat complexity and altering water flow and causing sulphide to accumulate in the sediment. Sulphide is toxic to eelgrass and a decline in *Zostera marina* as a consequence of invasive oyster growth was observed in British Columbia, Canada (Kelly & Volpe, 2007). The authors did not state the level of effect quantitatively and therefore the level of impact in terms of the resistance benchmarks used in this study is not clear.

Sensitivity assessment. Invasive species are affecting seagrass habitats around the UK with invasive flora having the greatest impact on seagrass beds so far recorded. However, there are extensive knowledge gaps on how invasive species influence the health of *Zostera* beds in UK waters. More research is needed in order to fully comprehend this pressure. Resistance is assessed as **'Low'**. Return to 'normal' conditions is highly unlikely if an invasive species would come to dominate the biotope. Indeed recovery would only be possible if the majority of the INIS were removed (through either natural or unnatural process) to allow the re-establishment of other species. Therefore, actual resilience is assessed as **'Low'** resulting in an overall **'High'** sensitivity score.

Introduction of microbial pathogens	Low Q: High A: High C: High	Low Q: High A: Medium C: Medium	High Q: High A: Medium C: Medium

Historic records show that seagrass species, in particular, *Zostera marina*, are highly susceptible to microbial pathogens. During the 1930s, a so-called 'wasting disease' decimated the eelgrass *Zostera marina* in Europe and along the Atlantic Coast of North America with over 90% loss (Muehlstein, 1989). Wasting disease resulted in black lesions on the leaf blades which potentially lead to loss of productivity, degradation of shoots and roots, eventually leading to the loss of large areas of seagrass (Den Hartog, 1987). Wasting disease is caused by infection with a marine slime mould-like protist, called *Labyrinthula zosterae* (Short *et al*., 1987; Muehlstein *et al*., 1991). Recovery of seagrass beds after the epidemic has been extremely slow or more or less absent in some areas such as the Wadden Sea (Van der Heide *et al*., 2007). The disease continues to affect *Zostera marina* in temperate regions with variable degrees of losses but not to the extent of an epidemic (Short *et al*., 1988). The exact conditions responsible for an outbreak are still unknown but it has been shown that already weakened plants are more susceptible to infection (Tutin, 1938; Rasmussen, 1977) and that salinity plays a role the pathogen activity (Muehlstein *et al*., 1988).

Sensitivity assessment. *Zostera marina* is highly susceptible to microbial pathogens, which were in the past responsible for important reductions in seagrass populations. A sensitivity of 'High' has been recorded ('Low' resistance, 'Low' resilience)

Removal of target species	None Q: Medium A: High C: High	Medium Q: High A: Medium C: Medium	Medium Q: Medium A: Medium C: Medium

Seagrass is not targeted by commercial fishery in the UK at present. Seeds and shoots are, however, harvested for extensive transplantation project aimed at promoting seagrass populations in areas denuded by natural or anthropogenic causes. Divers are most commonly employed to remove material from the source population, an activity with a low overall impact on seagrass habitats. However, in the USA, a mechanical seed harvesting technique was invented and put into practice (Orth & Marion, 2007). The mechanised harvester is able to drastically increase the number of *Zostera* seed collected from a source population (1.68 million seeds in one day compared to 2.5 million seeds collected by divers in one year). However, the large scale removal of seeds, the productive output of seagrasses, can affect the integrity of the natural seagrass beds. To date, no mechanical harvesting has been employed in the UK. The ecological impact of seed collection by divers is low; the harvesting of *Zostera* in British waters has, therefore, a minimal effect on natural seagrass habitats. The effect of the translocation of species is covered in the pressure 'genetic modification and translocation of indigenous species'.

Harvesting of seagrasses as craft material is a small but growing, industry. The present legislation for the conservation of seagrasses will discourage the expansion of this industry (see Jackson *et al.* 2013 for a full list on the political framework for seagrass protection in the UK). Seagrass beds are not considered dependent on any of the organisms that may be targeted for direct removal e.g. oysters, clams and mussels. However, an indirect effect of fisheries targeting bivalves is a change in the water clarity, crucial for the growth and development of *Zostera* species. Indeed bivalves have been shown to significantly contribute to the clearance of the water column which subsequently increases light penetration, facilitating the growth and reproduction of *Zostera* species (Wall *et al*., 2008). Newell & Koch (2004) using modelling, predicted that when sediments were resuspended, the presence of even low numbers of oysters (25 g dry tissue weight/ m^2) distributed uniformly throughout the domain, reduced suspended sediment concentrations by nearly an order of magnitude. A healthy population of suspension-feeding bivalves thus improves habitat quality and promotes seagrass productivity by mitigating the effects of increased water turbidity in degraded, light-limited habitats (see, changes in suspended solids). Bivalves also contribute pseudofaeces to fertilize seagrass sediments (Bradley & Heck Jr, 1999).

Seagrass plant may be directly removed or damaged by static or mobile gears that target other species. These direct, physical impacts are assessed through the abrasion and penetration of the seabed pressures. The sensitivity assessment for this pressure considers any biological/ecological effects resulting from the removal of target species on this biotope.

Sensitivity assessment. Seagrass beds have no avoidance mechanisms to escape targeted harvesting of leaves, shoots and rhizomes. Resistance to this pressure is therefore assessed as ‘None’. Studies of the effects of wildfowl grazing (see resilience and recovery above) suggest that recovery from the removal of target species will be rapid resulting in 'Medium' resilience score. Added anthropogenic disturbance may, however, be detrimental to seagrass beds as soon as the ‘normal’ level caused by grazing birds is exceeded by human activities. Overall the sensitivity of this biotope is deemed ‘Medium’ to this pressure.

Removal of non-target species	None Q: Low A: NR C: NR	Low Q: High A: Medium C: Medium	High Q: High A: Medium C: Medium

Filter-feeders such as mussels, clams and scallops are often associated with seagrass beds. Fisheries targeting these bivalves employ methods such as trawling, dredging, digging and raking which all result in the non-targeted removal of seagrass species. The direct physical effects of such fishing methods on seagrass are described in detail for the pressure ‘penetration and/or disturbance of the substratum’. Seagrass plants and the sedimentary habitat may be directly removed or damaged by static or mobile gears that are targeting other species. These direct, physical impacts are assessed through the abrasion and penetration of the seabed pressures. The sensitivity assessment for this pressure considers any biological/ecological effects resulting from the removal of non-target species in this biotope.

Incidental removal of the key characterizing seagrass species and associated species would alter the character of the biotope. The biotope is characterized by the presence of beds of seagrass, these provide habitat structure and attachment surfaces for epiphytic species. These may also modify local habitats through changes in water flow and the trapping of sediments. The loss of the turf due to incidental removal as by-catch would, therefore, alter the character of the habitat and result in the loss of habitat structure and species richness. The ecological services such as primary and secondary production and habitat engineering provided by seagrass and the associated species would also be lost.

Sensitivity assessment. Incidental removal of seagrass as by-catch would be detrimental, altering the character of the biotope and removing the habitat structure, and could lead to reclassification of the biotope where extensive removal occurs. Therefore, resistance is considered to be 'None', resilience 'Low' and sensitivity 'High'.

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