Marine
Management
Organisation

Identifying sites suitable for marine habitat restoration or creation (MMO1135)


Medmerry managed realignment in 2015 © J. Akerman

MMO1135: Identifying sites suitable for marine habitat restoration or creation

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Contents

Figures

Tables

Executive summary

The Marine Management Organisation (MMO) is currently going through the process of marine planning for English marine waters. During consultations, stakeholders expressed a desire for a national dataset to be developed to identify sites suitable for coastal and marine habitat creation or restoration developed. In response, the MMO commissioned ABPmer to help develop such a dataset and identify which habitats are the 'best' to be created.

The main purpose of this study was to develop Geographical Information System (GIS) datalayers which could be used / uploaded onto the MMO's Marine Information System (MIS). Six datalayers were consequently produced.

In order to facilitate the informed use of the datalayers from this project, a brief literature review on the ecology of these habitats has also been presented, focusing on the environmental conditions required for their restoration or creation. Furthermore, the status of the habitats, and techniques which have been employed to create or restore them, have also been summarised.

The six datalayers developed for this project have been grouped in relation to the three habitats or habitat groups which formed the focus of the datalayer tasks:

- Mudflats and saltmarshes:
    - Potential habitat creation sites within the current floodplain (applying the techniques known as 'managed realignment' or 'regulated tidal exchange');
    - Potential beneficial use (mud) – stretches which may benefit;
    - Potential beneficial use (mud) – potential material sources (maintenance dredge disposal sites);
- Biogenic reefs:
    - Potential honeycomb worm (*Sabellaria alveolata)* restoration – historic and current sites;
    - Potential European flat oyster *(Ostrea edulis*) restoration – historic and current sites;
- Seagrass beds:
    - Potential seagrass creation / restoration – historic sites.

For each datalayer, a detailed description of the methodology for its development has been summarised in the report, and broad guidance for its use provided, as well as limitations / caveats discussed.

These datalayers can all be used to aid searches for potential restoration or creation sites. They would generally be most useful during the initial stages of a search for potential sites, and further investigations and consultation of local knowledge would always be required to confirm whether or not a site is actually suitable for the restoration or creation of a given habitat.

With regard to creating a list of the 'best' habitats for use in (re)creation / restoration, the report presents a high-level matrix which broadly assesses the restoration feasibility of all marine and coastal habitats which are considered to be of principal national importance. This is in line with recommendations from stakeholders, and available guidance and reviews.

1. Introduction

The UK Marine Policy Statement (2011) highlights the Government's aim to ensure a sustainable marine environment which promotes healthy, functioning marine ecosystems and protects marine habitats. This aim includes the creation of habitat to improve and extend the amount of habitat available for species and where appropriate the recovery of biodiversity. According to the Marine Management Organisation (MMO), marine planning stakeholders have '*identified the lack of a national 'dataset of sites' as an evidence gap that limits the ability to understand the potential of sites to create or restore habitats adequately and in a strategic way. In addition, throughout the marine planning process an opportunity has been identified to enhance the environment in addition to mitigating for adverse impacts.'*

The MMO commissioned ABPmer to develop such a national dataset of sites that are suitable for marine habitat restoration or conversion. For some of the tasks, ABPmer was supported by Aquatic Environmental Research (AER) Ltd and the Royal Society for the Protection of Birds (RSPB).

The objectives of the project were as follows:

- To create a complete, consolidated, list of the sites potentially available for marine habitat creation, using information from all available sources that can be integrated into the existing MMO Marine Information System (MIS); and
- To create a list of the best types of habitat for use in creation and detail the trade-offs, limitations and considerations.

The MMO envisages that this dataset will help inform marine plan policy development that may help increase the amount of ecologically important habitat, where appropriate and in line with current legislation. Marine plans are forward looking and ensure flexibility to anticipate, and accommodate, a range of future demands and scenarios, including enhancing the marine environment. The outputs of this research will enable decisions to be made that integrate social, economic and environmental considerations.

This report is structured as follows:

- Methods – Section 2;
- Habitat background (ecology and creation techniques) – Section 3;
- The datalayers (methodology for each, results and limitations, guidance) – Section 4;
- 'Best' habitats for restoration / creation – Section 5; and
- Summary and conclusions – Section 6.

2. Methods

As noted above, this project involved two tasks:

1. The creation of datalayers to support the development of a complete dataset of sites potentially available for marine habitat creation, and
2. The creation of a list of ‘best’ habitats for the use in (re)creation / restoration.

The methods employed to undertake these tasks are now briefly outlined in turn.

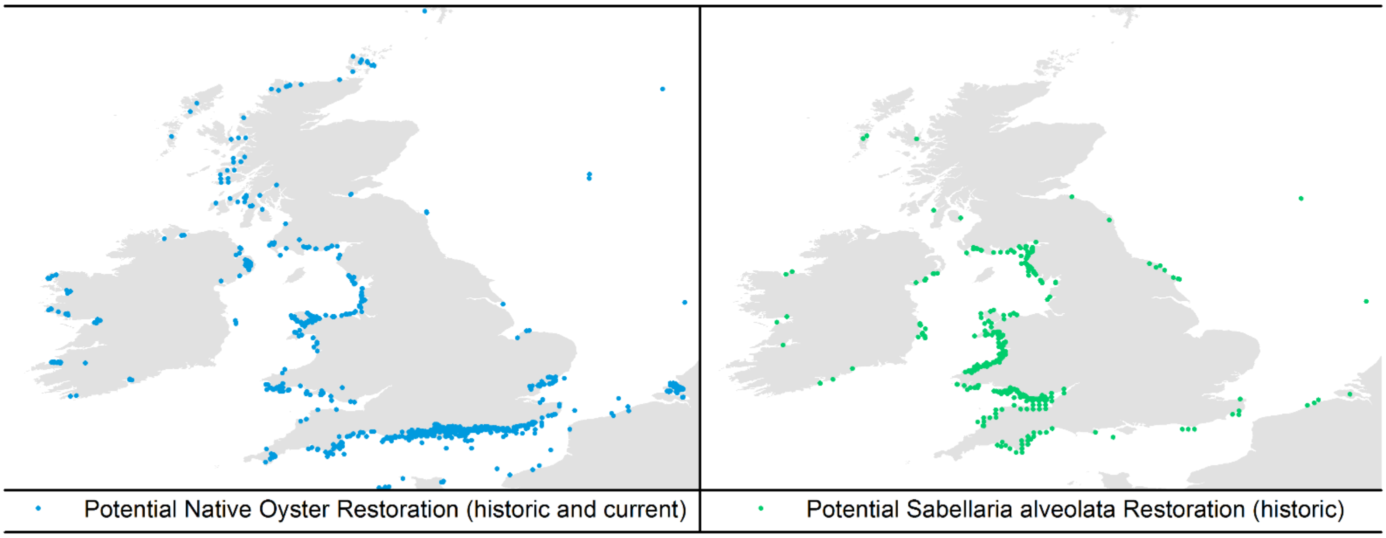
2.1. The datalayers

In order to fulfil the requirement of creating a dataset of available sites, six Geographical Information System (GIS) datalayers were produced, with a view to them being uploaded onto the MMO’s MIS. The methodology for their creation was confirmed and refined at a methodology workshop which was attended by the project’s consultant team and members of the Defra group (see Section 4 for more detail). The detailed methodology employed for each of the datalayers is described in Section 4.

For the datalayers, the project focussed on the following habitats:

- Mudflats and saltmarshes;
- Seagrass meadows; and
- Biogenic reefs (focusing on those formed by European flat oyster (*Ostrea edulis*) and honeycomb worm (*Sabellaria alveolata*)).

Figure 1: Example data layers (“© Marine Management Organisation 2019 and Collins Bartholomew 2018 copyright.”)



In order to facilitate the informed use of the datalayers generated by this project, a brief literature review on these habitats was undertaken, focussing on their ecology, in order to provide background on the environmental conditions required for their restoration or creation. Furthermore, the status of the habitats and techniques which have been employed to create or restore them, were also investigated. This review is provided in Section 3.

2.2. A list of ‘best’ habitats

In order to define such a list of ‘best’ habitats for use in (re)creation / restoration, the following methodology was applied by ABPmer.

- Step 1 – A review of previous and ongoing initiatives / literature. A summary of this review is provided in Section 5.
- Step 2 – A workshop to discuss the opportunities and issues surrounding habitat equivalence with expert stakeholders (see Section 5 for more detail).

For the review, habitat creation / restoration was set in the context of habitat offsetting and biodiversity and environment net gain, as these are topic areas where habitat accounting and issues of habitat equivalency would be encountered, and related literature may thus aid in identifying a methodology for creating a list of ‘best’ habitats.

Rather than only presenting a list of a few habitats, it was decided to develop a high-level matrix (see Table 9), which broadly assesses the restoration feasibility of all marine and coastal habitats / features which are considered to be of principal importance (under the 2006 Natural Environment and Rural Communities (NERC) Act). This is in line with recommendations from the workshop participants, and available guidance and reviews (see Section 5).

3. Habitat background

In order to facilitate the informed use of the datalayers generated by this project, this section provides a brief background on the ecology of, and the specific environmental conditions required for, each of the habitats which were the focus of this project. The section also reviews the status of the habitats and techniques which have been employed to create or restore them. Intertidal mudflats and saltmarshes are discussed first in Section 3.1, before background on seagrasses and biogenic reefs is provided in Sections 3.2 and 3.3 respectively.

For a number of reasons, other habitats were not considered for this project. The potential locations for artificial islands and rocky reefs are relatively unconstrained, and hence creating a spatial datalayer would be difficult. Similarly, small scale enhancement interventions, such as designing in rock pools within man-made structures to increase habitat complexity, were considered to be outside the scope of this project given the spatial scale over which habitat creation / restoration opportunities have been identified. However, a brief summary of the range of ecological enhancements that are feasible in the marine environment is provided in Section 3.4.

3.1. Intertidal mudflats and saltmarshes

3.1.1 Intertidal mudflat and saltmarsh ecology

Environmental conditions

In England, intertidal habitats (mudflats and saltmarshes) are generally understood to develop between the levels of the highest astronomical tides (HAT) and mean low water springs (MLWS) on tide-dominated, sheltered, coasts, where the predominant sediments are silts and muds. These fine sediments are kept in suspension where currents are fast, and settle when the currents are slowest (French, 1997).

Tidal flats are un-vegetated ‘banks of mud or sand that are exposed at low tide’, which generally slope gently seawards. Figure 2: Generalised division of intertidal habitats based on elevation in relation to tidal height (from Davis et al., 2018).

Figure 2 describes how, as elevation increases, and thus tidal inundation frequency decreases, so vegetation can become established and saltmarshes develop throughout temperate regions (with mangroves in the tropics) (Trenhaile, 1997). Sediments needed for this elevation increase (‘accretion’) can be derived from marine, coastal, and fluvial sources, as well as *in situ* reworking (Pethick, 1984).

Intertidal habitats are generally divided into several zones; these zones broadly correspond to the frequency of tidal inundation and the associated effects of salinity and tidal scouring. Four main saltmarsh zones can be distinguished, and are commonly referred to as ‘pioneer’, ‘low’, ‘middle’ and ‘higher’ marsh. These saltmarsh types are typically associated with a characteristic number of tidal inundations per year (Table 1), which can broadly be related to tidal levels as shown in Figure 2. For context, approximately 705 tidal cycles occur in a year on most UK coast.

Figure 2: Generalised division of intertidal habitats based on elevation in relation to tidal height (from Davis et al., 2018).

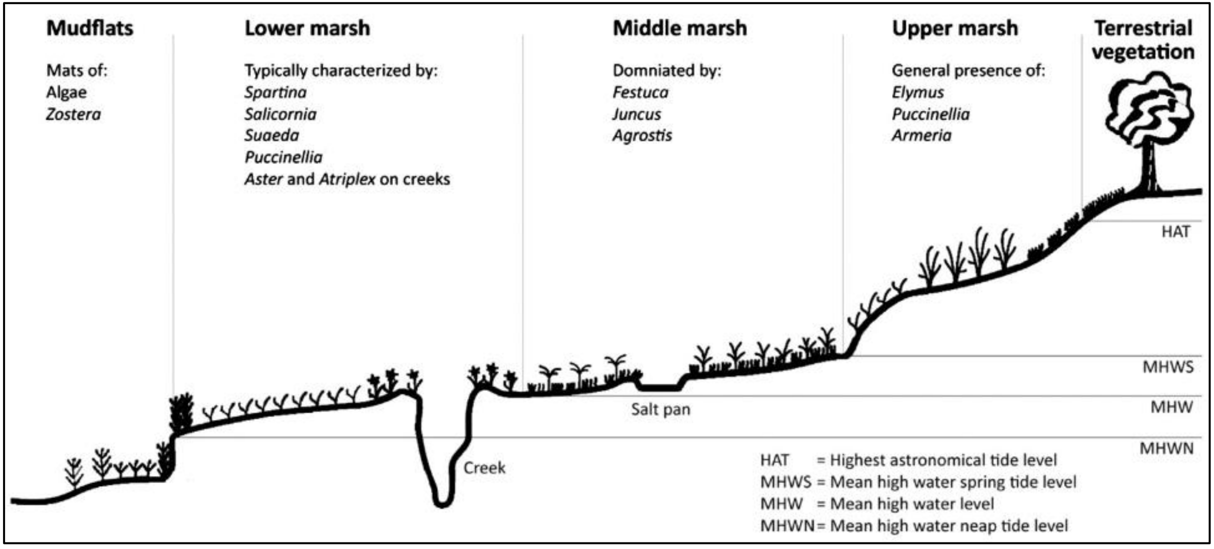


Table 1: Habitats associated with number of tidal inundations per year (from Toft et al., 1995; Leggett et al., 2004).

Inundations Per Year	Habitat
More than 450	Mudflat
450 to 360 (maximum continuous exposure: 9 days; minimum daily daylight submergence of 1-2 hours)	Pioneer marsh
Less than 360 (typical 300) (minimum continuous exposure: 10 days; maximum daily daylight submergence of 1 hour)	Low, mid and upper saltmarsh

Each saltmarsh species has a different tolerance to tidal flooding, and therefore a different, although often overlapping, vertical range. *Spartina* is the saltmarsh species most tolerant of tidal inundation, and its distribution was hence found to be slightly less closely related to tidal levels than other plants (Clarke et al., 1993).

Intertidal habitat zonation can also be impacted by other physio-chemical variables; these, however, tend to be less important than the inundation frequency or duration of coverage by seawater discussed above. For example, the nature of the sediment may influence the elevation that saltmarsh species occur. On sandier substrates, lower marsh zones tend to be at higher elevations due to the lower nutrient contents of sand (Adam, 1990). In larger estuaries, the limits of species also tend to be farther up the shore than predicted by the level of Mean High Water Neaps (MHWN) alone. This is due to the generally greater degree of exposure to wind and wave action, increased velocity of flows and higher turbidity variation (Leggett et al., 2004).

Where disturbance from the action of tides and waves during seed germination and plant establishment is too high (e.g. due to stormy weather), substantial erosion of young plants with less effective root anchorage, and inhibited germination at lower elevations, might occur; consequently, the lower limit of the pioneer zone might shift upwards (Boorman, 2003). Chemical factors can also have an influence, for example, excessively waterlogged sediments due to over-consolidation can lead to low oxygen diffusion rates, and consequently low sediment redox potential. Garbutt et al. (2006) thought this was the most likely explanation as to why halophytes in the Tollesbury managed realignment site (in the Blackwater Estuary) had established at elevations 0.19m higher than in the immediately adjacent areas.

Status

Nationally and globally, there have been extensive historic losses of intertidal habitats, chiefly due to human activities, mostly related to land claim, and the construction of sea defences, ports and harbours. It has been estimated that some 100,000 ha of British saltmarshes were lost between 1600 and 1900, mainly to gain more land for agricultural production (Toft et al., 1995). More recently, comparatively small-scale industrial and residential developments have been the main driver for land claim (Adam, 2002). Environmental assessment and conservation measures afforded over the last few decades have largely eliminated land claim as a pressure on intertidal habitats. More recent losses in saltmarsh extent at various locations have been attributed to other factors – coastal squeeze, isostatic tilt, sea level rise and/or increased storminess (Pye and French, 1993; Cope et al., 2008). A 2011 review of saltmarsh extent in England concluded that the rate of recent saltmarsh loss at national levels may have been slower than previously thought, but cautioned that further work was required (Environment Agency, 2011).

3.1.2 Intertidal mudflat and saltmarsh creation and restoration techniques

Four main techniques have been identified for the creation and restoration of intertidal mudflat and saltmarshes. These include:

- Managed realignment;
- Regulated tidal exchange (RTE);
- Beneficial Use / Sediment recharge; and
- Manipulation of natural processes.

Hybrids of these techniques are also feasible, and have been implemented. The first three bullets above are the focus of the datalayers produced for this project; however, a brief description for the manipulation of natural processes has been included for completeness.

The principles of each of these techniques are summarised below. A significant proportion of the information provided has been derived from ABPmer's Online Marine Registry (OMReg - www.omreg.net); this is not individually referenced below. Where information has been taken from other sources, these are given.

Managed realignment

The term 'managed realignment' is most commonly understood to involve a deliberate breaching, or removal, of existing seawalls, embankments or dikes in order to allow the waters of adjacent coasts, estuaries or rivers to inundate the land behind (see, for example, Leggett et al., 2004). In most instances, the newly flooded land is low-lying coastal floodplain and therefore a new seawall is needed to clearly define the inundated area and protect the hinterland behind. However, in areas with rising ground either no new line of defences or only a partial counterwall is required. RTE is often interpreted as a subset of managed realignment, however, for the purpose of this review, it is listed as a separate technique (and discussed further in the next sub-section).

There are essentially two different managed realignment methods which can be applied; these are: (1) managed breaching (or breach realignment) and (2) defence removal (or bank realignment).

Figure 3 shows an aerial image of the UK's first coastal (breach) realignment at Medmerry (near Selsey, West Sussex), which was implemented in 2013. To date, at least 100 managed realignment schemes have been implemented across Northern Europe, 51 of these are in the UK. Most of these, around 43 in total, have applied the breach realignment technique. The UK schemes were generally implemented on uninhabited agricultural land without significant existing infrastructure or nature conservation designations (though the fronting estuarine habitats have frequently been highly designated).

Figure 3: Aerial image of Medmerry managed realignment (January 2014).

(Image Credit: J. Akerman)

Regulated tidal exchange (RTE)

RTE involves the controlled exchange of estuarine or coastal waters onto a previously terrestrial site using a variety of exchange structures (sluices, culverts or weirs), rather than breaches, to control the tidal exchange volumes and the extent of hinterland flooding. These techniques are employed where there are concerns about the tolerances of the neighbouring waterbody to hydrodynamic change and/or in order to achieve particular ecological aims. These RTE schemes do not always need to have new counter walls; nevertheless, this tactic is often less sustainable in that it requires more ongoing management than an open breach scenario.

The variety of applied RTE approaches is large. They range from simple tidal gates with gaps to allow a finite amount of tidal water through to more complex structures with articulating panels, buoys, and counterweights that can be used to exert control over the timing of tidal exchange (e.g. to ensure that saline waters are extracted from an adjacent tidal river which has a freshwater/marine salt wedge feature). A self-regulating sluice gate is shown in Figure 4 as one example.

To date, a large number of generally small-scale RTE projects (25) have been undertaken in the UK, with the largest being the recently (November 2018) created Cell 4 dynamic lagoon complex on Wallasea Island (132 hectares (ha), Crouch Estuary). The main habitats created have been saline lagoons, saltmarshes and mudflats. The propensity of RTEs leading to saline lagoon creation is related to the reduced tidal amplitude experienced due to the exchange pipes/culverts severely restricting exchange, and the pooling of water in lower lying areas.

Figure 4: Self-regulating tide gate at the Goosemoor RTE (River Clyst, Devon).

Low tide High tide

(Image Credit: RSPB)

Sediment recharge / beneficial use

Sediment recharge in intertidal areas is a process by which dredged sediments are placed over or around intertidal mudflats and saltmarshes to either create habitat (most often saltmarshes), or restore or protect intertidal habitats from ongoing erosion (Nottage and Robertson, 2005; Defra and Environment Agency, 2007). This approach is particularly valuable for protecting habitats that are sediment starved or subject to erosion and where the introduction of dredge arisings will allow the habitat to cope with, or respond to, sea level rise.

In the UK, approximately 20 intertidal recharge projects have been undertaken to date; some of which recur on a regular basis. These have been mainly in Essex, Suffolk and on the South Coast. Two of these projects (Allfleet’s Marsh and Trimley Marsh) are managed realignment schemes which included the beneficial use of dredged sediment as land forming materials prior to breaching the sea walls (see Figure 5 for an image showing one of the recharge campaigns at Allfleet’s Marsh). None of the known schemes have involved intertidal mudflat or saltmarsh creation from subtidal habitats.

Figure 5: Large-scale (550,000m^3) beneficial use of dredged silt for pre-breach land-forming at Allfleet's Marsh on Wallasea Island.

(Image Credit: Defra)

In many estuaries in the UK, fine materials dredged during maintenance and capital dredging campaigns are deposited in a subtidal location within the same estuary; not to create mudflat from subtidal, but to essentially trickle charge sediment back into the estuarine system. Some notable national examples of this 'sustainable relocation' of dredged sediment include the Humber Estuary (Lonsdale et al., 2012), the Stour and Orwell Estuaries, the Dee Estuary and Poole Harbour. The hypothesis behind this sediment retention approach is that there is a net balance between the amount of material being deposited and eroded in many tidal estuaries. Such a balance may be disturbed when an estuary is dredged, and continuous permanent removal of materials could eventually lead to erosion of intertidal habitats (Cefas, 2009).

The direct placement of material onto the subtidal in order to elevate an area into the intertidal, and thus create mudflat, has never been practiced in the UK. There have, however, been examples of this in the US and Japan, where recharge has been very widely practiced for decades (PIANC, 2009). Several large-scale port expansion projects have also recently demonstrated that elevations can be built up from subtidal placement of material, albeit requiring significant engineering effort (for example, the 2,000ha Maasvlakte 2 expansion at the port of Rotterdam).

Carrying out beneficial or 'alternative' use projects using fine/silt sediments can be technically challenging and costly. For this reason, the extent to which such materials are used is very limited (ABPmer, 2018). Several initiatives have been, and continue to be, undertaken to address the known barriers to implementation and facilitate the increased actual use of this technique (e.g. the RSPB's SEABUDS project (Precipitating a SEA Change in the Beneficial Use of Dredged Sediment), as well as regional initiatives such as the Solent Forum's BUDS project (Beneficial Use of Dredging in the Solent).

Manipulation of natural processes

The manipulation of natural processes encompasses projects which alter the existing sedimentary regime along a shoreline in order to protect habitat and possibly create mudflat. This includes a wide range of possible techniques such as introducing obstructions or altering shorelines. There are techniques which can potentially be used to expand mudflat seawards, onto existing subtidal areas, though there are no known (intentional) examples of this in the UK. Such structures are installed in areas which are exposed to relatively high tidal or wave energy forces which would normally prevent the settling of sediments, or re-suspend any that had settled during slack periods. This is provided that the suspended sediment concentration in the system is high enough for accretion to take place. Thus, the artificial import of sediment is not necessary, but instead, structures are put in place to reduce energy and encourage sediments to settle and accrete. In the past, the main methods used for increasing sedimentation in intertidal areas have included brushwood fencing, polders/sedimentation fields, wave breaks or groynes.

Wavebreaks are generally located some distance offshore in more exposed situations, usually in parallel to the shoreline. They can be constructed from a variety of materials, including brushwood, sandbags, geo-tubing and redundant barges (see Figure 6 for such barges at Horsey Island (Hamford Water, Essex)). Many examples of such wavebreaks exist along the UK’s coast, though numbers and case examples are relatively difficult to determine. A notable example includes the Thames lighter barges that were sunk at various locations off the coast of Essex in the 1980s (see French, 2001).

Sedimentation fields/polders have been used extensively along the Dutch and German Wadden Sea coasts for centuries. Land claim would have traditionally been the ultimate aim of this method; although nowadays it is undertaken to build up saltmarsh in front of coastal defences. This technique was trialled in at least 17 locations in Essex in the 1980s (e.g. at Horsey Island, see Figure 6), with mixed success (see French, 2001).

Figure 6: Image of Horsey Island (north shore), showing recharge area, wave breaks, and sedimentation fields.

(Image Credit: Google Earth – annotated by ABPmer)

Hybrids

The creation of mudflats and saltmarshes can potentially be achieved through the combined use of a number of the four main techniques described above. The use of multiple techniques within the same scheme/location has been implemented in a number of locations around the UK. The Allfleet's Marsh and Trimley managed realignment schemes, for example, also incorporated sediment recharge (ABPmer, 2011; John, 2013). Furthermore, sediment recharge schemes generally involve the construction of retaining structures. These have often taken the form of substantial shore parallel structures which also act as wave breaks (e.g. Horsey Island, Essex; see Figure 6). The combined use of managed realignment and RTE has been included within scheme designs at Steart and Cherry Cobb Sands (Black and Veatch, 2012), and has been implemented on Wallasea Island by the RSPB (ABPmer, 2018). Using sedimentation fields to retain recharged sediments on site and encourage further accretion is also conceivable, though largely un-trialled.

Summary of experience

Collectively, there is now a comprehensive evidence base providing a robust practical technical foundation for future schemes. The main message from this accumulated experience is that, when properly designed, these initiatives are effective, particularly with regard to managed realignment and RTE schemes. Mobile invertebrates and fish can colonise within a few days and weeks while some slower colonising species can take months to a few years. Plants also establish within the first year and then flourish over a few years so that a visibly vibrant and diverse wetland can be achieved within 3 to 4 years (albeit not necessarily with the same diversity as adjacent established habitats, at least not initially). In addition, the majority of sites are stable and accrete with sediments (at rates which vary depending on the estuary suspended sediment load and the elevation of the intertidal habitats inside the site), making them sustainable and able to cope with sea-level rise in the medium to possibly long-term. Longer-term mudflat creation can be problematic, particularly in high-turbidity estuaries such as the Severn and Humber Estuaries, where mudflats typically progress to saltmarsh within a decade, or quicker. However, longer-term mudflat has been achieved in lower turbidity environments such as the Crouch/Roach Estuary (at Allfleet's Marsh, Essex) (ABPmer, 2017b).

Implementing such projects, however, often requires a lot of preparatory analysis, consultation, planning and assessment work so projects can be costly, especially at a large scale. A cost of £40,000 to £50,000 per hectare is typical for managed realignment schemes, though some schemes have had much higher per hectare costs of £100,000 or more. Costs of beneficial use of muddy dredge material for intertidal sediment recharge are site specific and may be cheaper, the same, or more expensive compared to 'at sea' disposal (ABPmer, 2017a, b).

Numerous scheme design lessons have been learned and are outlined elsewhere (e.g. Leggett et al., 2004; Nottage and Robertson, 2005; ABPmer, 2017b). In summary however, ensuring that the site is of a suitable elevation, in a sheltered location, and has well designed creeks and channels to facilitate efficient inundation and drainage (particularly for higher lying sites), are the most crucial factors leading to the successful creation of intertidal habitats. Careful design is also required to ensure that the adjacent estuary or coast are not subject to undue impacts, amongst others by calculating the anticipated tolerance of the estuary to the volumetric changes during the design phase, and having a thorough understanding of baseline conditions.

3.2. Seagrass meadows

3.2.1 Seagrass ecology

Environmental conditions
There are two species of seagrass found in UK waters, both belonging to the genera *Zostera* (Family: *Zosteraceae*). *Zostera marina* is the largest of the British seagrasses and typically occurs in the shallow sublittoral down to about 4m depth, in fully marine conditions and on relatively coarse sediments. Dwarf eelgrass, *Z. noltii* occurs higher on the shore than the other two species, typically on mixtures of sand and mud.
Z. marina is an intertidal to sublittoral species that forms dense beds, with trailing leaves up to 1m long. It is generally found in shallow, fully marine conditions on muddy to relatively coarse sediment (occasionally with a mixture of gravel) (Davison and Hughes, 1998; Dale et al., 2007). *Z. noltii* generally inhabits sandy and muddy substrates at the upper extent of the intertidal.

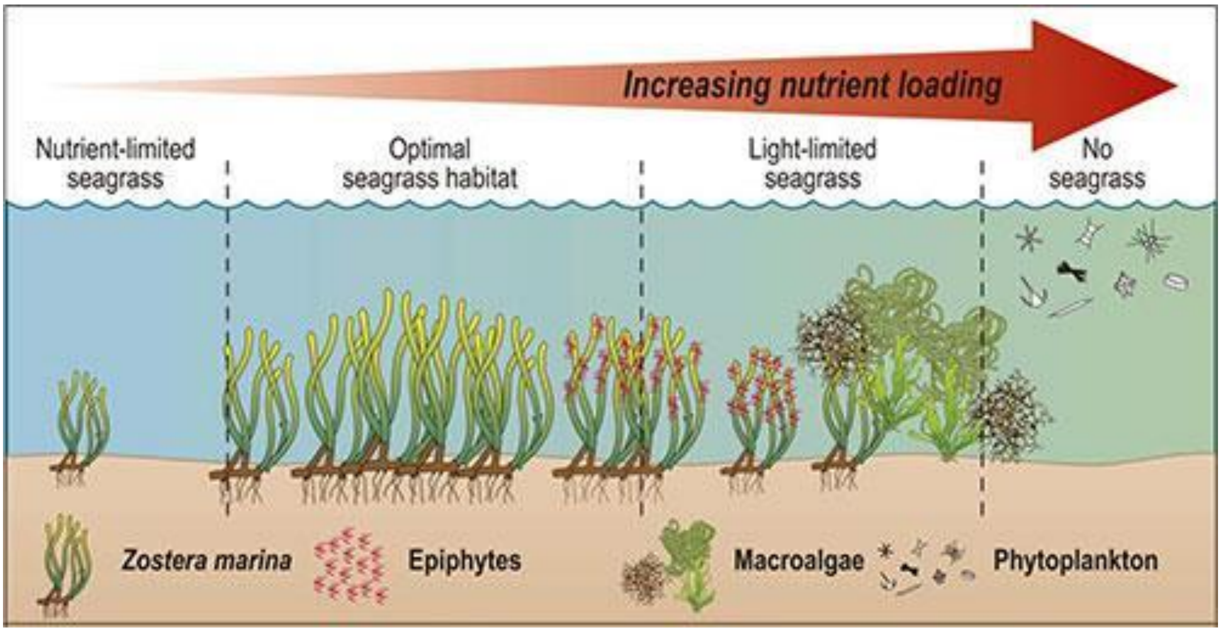
Although *Z. marina* requires marine conditions, reduced levels of salinity (e.g. 20ppt) can be tolerated, although lower salinities for long periods of time reduce plant performance (Salo et al., 2014). *Z. noltii* is more tolerant to large fluctuations in salinity and highly resistant to desiccation and so can be found high up in intertidal zones and further into estuarine environments (Charpentier et al., 2005).

Experimental and observational research finds that *Z. marina* is generally adapted to temperatures ranging from -1 to 25°C, however optimum conditions for growth may be more restricted and range from 13 and 24°C (Lee et al., 2007). There has been far less experimental work conducted on the environmental requirements of *Z. noltii* relative to *Z. marina*. As a result, there is less information on the effect of temperature on this species (Pérez-Lloréns and Neill, 1993).

In the UK, *Z. marina* is most commonly restricted to a maximum of about 7m of water depth (below chart datum), however the maximum known depth of *Z. marina* in the British Isles is at 10m (Dale et al., 2007). This shallow depth distribution is the result of its high light requirements as a photosynthetic organism. Estimates from across the biological range of *Z. marina* suggest it requires between 12% and 37% of surface irradiance (SI) to survive in the long-term with a mean SI of 18% (Lee et al., 2007 and Erftemeijer and Lewis, 2006). UK *Z. marina* appear to have similar light thresholds to those found in the literature (Bertelli and Unsworth, 2018). There is limited literature on the light requirements of *Z. noltii*, though what information is known indicates that light requirements are very low (Lee et al., 2007; Erftemeijer and Lewis, 2006) explaining the capacity of *Z. noltii* to live in estuarine conditions that are commonly turbid. Light available to seagrass reduces with increasing depth and this availability can be influenced by a range of factors such as turbidity, epiphytic growth, plankton blooms, shading from algae (such as *Ulva* spp. or *Enteromorpha* spp.), the proximity of man-made structures, and the presence of epiphytic organisms attached to seagrass leaves (Dennison et al., 1993; Brodersen et al., 2015).

Seagrasses, like any angiosperm, require a sufficient supply of nutrients. Nutrients in seagrass are critical in the establishment of a meadow and its maintenance and growth. However, as nutrient loads increase they have an increasingly negative impact upon the light availability for seagrass photosynthesis (Figure 7). This is a fine balance and elevated nutrients can result in reduced water quality and smothering by macro and microalgae (Burkholder et al., 2007), epiphytic algal growth on the seagrass, macroalgae becoming more dominant, and the density of phytoplankton in the water column increasing.

Figure 7: Schematic representation of the influence of increasing nutrient levels on seagrass (from Burkholder et al., 2007).



Seagrasses generally live in areas sheltered from wave action, although there are reports of *Zostera* species living in locations of wave stress (Jackson et al., 2013). It is likely that in such areas *Zostera* is not subjected to shallow wave action, but instead the seagrass lives in slightly deeper environments to avoid the waves.

Large tidal amplitudes force subtidal seagrass to grow in deeper environments (where there is less chance of exposure to the air) (Jackson et al., 2013). In a review of the literature on physical drivers of seagrass current velocity was found to influence seagrass distribution within a viable range of 5-180cm.s^{-1} (Koch, 2001).

Seagrasses cannot cope with persistent high sedimentation rates. This is both as a result of the suffocation this causes to the leaf (Cabaco et al., 2008) and the negative impacts of sediments landing on seagrass, reducing the capacity of the leaves to remain aerated (Brodersen et al., 2017). A meta-analysis of the literature indicates that *Z. marina* can by the very nature of its larger size (longer leaves) cope with higher amounts of sedimentation than *Z. noltii*. *Z. marina* had 50% mortality at 4cm depth of sedimentation whilst *Z. noltii* had similar mortality but at 2cm of sedimentation (Cabaco et al., 2008).

Intertidal exposure / emersion creates a potential environmental stress to seagrass due to the risks of desiccation. *Z. marina* and *Z. noltii* (when growing intertidally or in the shallows) both use low tide periods for rapid assimilation of atmospheric CO_2 as long as their leaf water status is favourable, but desiccation can significantly reduce carbon gain at low tides (Davison and Hughes, 1998). Experimental studies indicate that *Z. marina* is more sensitive in this respect than in *Z. noltii* (Leuschner et al., 1998). This contributes to explaining why *Z. noltii* can live higher up on the intertidal relative to *Z. marina*.

Status

Seagrass meadows are declining at an unprecedented rate (Waycott et al., 2009; Orth et al., 2006). In the UK, a series of research papers have clearly defined seagrasses to be under threat and in a perilous state (Jones and Unsworth, 2016; Jones et al., 2018; Unsworth et al., 2017). Seagrass was thought to be once abundant and widespread around the British coasts, but serious declines have occurred, in particular due to poor water quality (eutrophication and other pollutants), land claim and a severe outbreak of 'wasting disease' in the early 1930s (Davison and Hughes, 1998; Butcher, 1933). Such an outbreak of disease was probably exacerbated by poor coastal water quality (Short and Wyllie-Echeverria, 1996). Recovery of eelgrass beds in the UK has been slow and patchy, with loss still continuing in many places, although cases of extensive recovery have occurred such as within intertidal *Z. noltii* beds in the Milford Haven Water way in West Wales (Bertelli et al., 2017).

3.2.2 Seagrass creation and restoration techniques

Seagrass restoration has been conducted for over 50 years, and the means of doing this can principally be split into two major techniques:

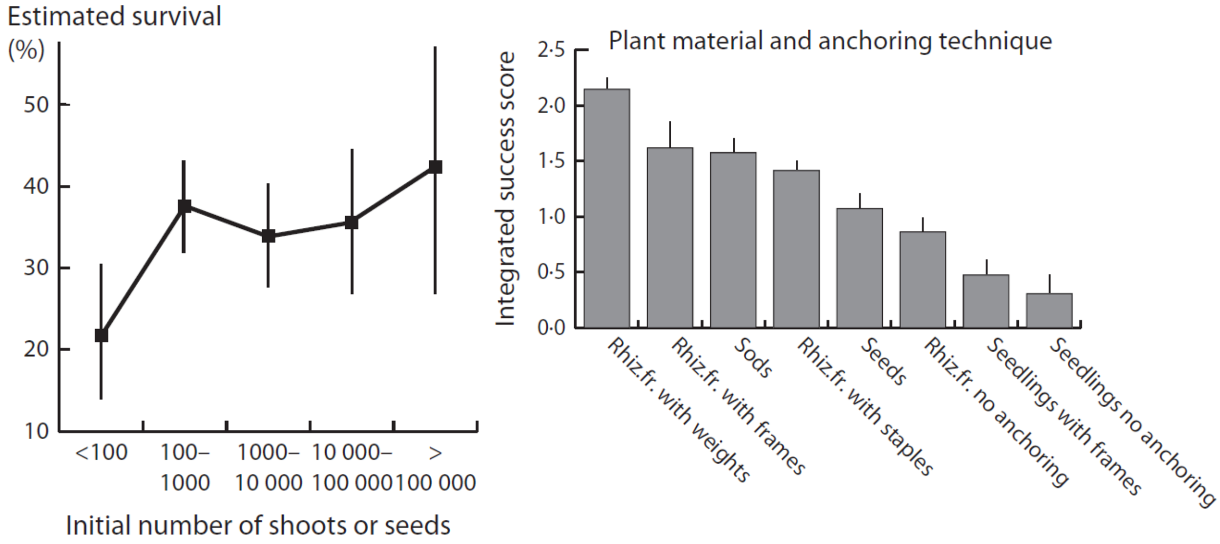
- Replanting;
- Reseeding.

Both techniques have their relative merits and have exhibited varying levels of success. Although a lot is now known about seagrass restoration, much more remains to be researched and, as a result, the success rate of restoration projects is still often very low. The use of re-seeding generally relates to the collection and targeted redistribution (and sometimes processing) of wild seeds. Adult shoot replanting normally involves harvesting plants from an existing meadow and transplanting them to the restoration site. This is because there is no readily available source of nursery grown plants.

In most cases, some means of anchoring the shoots to the bottom is necessary until the roots can take hold (root into the bottom). Replanting uses either labour-intensive diving techniques or various mechanistic approaches to planting various sizes and ages of seagrass plants into new localities. In the US, reseeding and replanting techniques have sometimes been used together. Using seeds possibly in conjunction with adult plants, may in some instances prove more effective (van Katwijk et al., 2016).

Seagrass restoration has the capacity to be both very expensive and have a high risk of project failure. Failures in many projects historically have been the result of limited consideration of the habitat requirements for seagrass and the continued presence of the stressor that caused the original seagrass loss. A recent review of the success of restoration projects globally found that success relates to the severity of the habitat degradation (eutrophication being worse than the combined impacts of dredging and filling or construction). The review also highlights the need for restoration to occur at sufficient scales in order to facilitate positive feedbacks and to spread the chances of success (see Figure 8) (van Katwijk et al., 2016). With regard to techniques, seeds, adult plants and intact units of native sediment with roots (sods) are not significantly different, although seedlings show lesser planting results. A short distance to the donor site is also related to success. Whereas transplantations (replanting) frequently fail (60%) or have limited success, a substantial number of transplantations show huge expansion rates as well (van Katwijk et al., 2016).

Figure 8: Influence of restoration scale and method on seagrass survival and growth (from Katwijk et al., 2016).



Summary of experience

In summary, there is limited experience of seagrass restoration in the UK. Elsewhere, restoration/creation has been attempted, with very varying levels of success. van Katwijk et al, (2009) describe a series of guiding principles laid out by the Wadden Sea restoration project in order to maximise success rates:

1. Ensure long-term survival by promoting self-facilitation through implementation at a large-enough scale (hectares);
2. Focus on facilitating natural recovery through alleviating recruitment limitation (‘let nature work for you’);
3. Spread risks through space and time by restoring multiple sites on multiple occasions;
4. Keep the costs of restoration (per hectare) as low as possible to achieve an as-large-as-possible scale of success; and
5. Minimize impacts on source meadows while avoiding introductions of invasive species at restoration sites.

3.3. Biogenic reefs

3.3.1 Biogenic reef ecology (focussing on the European flat oyster *Ostrea edulis* and the honeycomb worm *Sabellaria alveolata*)

The European flat oyster *O. edulis* and the honeycomb worm *S. alveolata* are two species that create reefs on lower intertidal shores, although both species also colonise subtidal regions. These biogenic reefs are defined as ‘solid massive structures which are created by accumulation of organisms’, and ‘clearly forming a substantial, discrete community or habitat which is very different from the surrounding seabed’ (Holt et al*.,* 1998). Intertidal reefs can be seen as biodiversity hotspots where species diversity contrasts with that of surrounding sediments (Dubois et al*.*, 2002). Due to their high density of organisms, they can also be seen as large biological filters and they play a significant role in trophic webs (Bruschetti et al*.*, 2008; Dubois et al*.*, 2009). Biogenic reefs are consequently of high importance to the ecological functioning of the habitats and areas in which they are found, and it is for this reason that reefs are of particular interest to nature conservation.

Sabellaria alveolata (Linnaeus, 1767), the honeycomb worm (see Figure 9) is a sedentary polychaete worm that constructs tubes of coarse sand grains and fragments of mollusc shells. It is one of the most prolific reef builders and colonises lower to mid-intertidal shores from Scotland to Morocco. *S. alveolata* creates topographic complexity and thereby generates spatial and ecological niches for other species. *S. alveolata* tube-reefs reduce physical and chemical stresses for intertidal species, create refuge from predation and competition and alter resource availability. The known spatial extent of *S. alveolata* habitat covers less than 0.5% of the British coastline making it by definition a “nationally rare” habitat (Sanderson, 1996; Naylor and Viles, 2000).

Figure 9: The largest *Sabellaria alveolata* reef in Europe in the Bay Mont St. Michelle.

(Image Credit: AER)

Although typically found within the low to mid-shore and *S. alveolata* populations have also been found in subtidal areas such as within the Severn Estuary (Mettam et al*.*, 1994; Holt et al*.*, 1998). It requires firm attachment surfaces and colonises different types of hard substrata including bedrock, boulders or pebbles and large bivalve shells such as oysters. *S. alveolata* also attaches to anthropogenically created hard structures such as coastal defences, settling mainly on or near established tube structures or degraded reef scars (Firth et al., 2015). It has also been known to settle on sedimentary substrata that has been stabilised sufficiently, for example by the sand mason worm *Lanice conchilega* (Larsonneur, 1994).

The reproductive biology of *S. alveolata* indicates the importance of the hydrodynamic regime for dispersal and recruitment. Larvae spend between six weeks and six months in the plankton before settling, but depending on local conditions, larvae may remain within a discrete area and colonise neighbouring reefs (Ayata et al., 2009).

The pelagic larvae of *S. alveolata* settle preferentially on existing honeycomb worm reefs. Following initial settlement, reefs grow rapidly. When individuals age, reefs may enter a stagnation phase with little expansion. Reefs are sustained by new recruits, but in the absence of primary or secondary settlement, reef destruction can begin (Wilson, 1976; Gruet, 1986). Individual *S. alveolata* worms live for 3-5 years (Gruet, 1986), but their sand tubes and the reef structures can persist over longer temporal scales in varying degrees of reef-health, partially dependent on environmental conditions (Naylor and Viles, 2000). When conditions become unsuitable, the reef generated by *S. alveolata* can outlive the polychaete itself, although species diversity tends to decline as the reef degrades (Hastings et al., 2007).

Temperature affects the growth and mortality of *S. alveolata*. The metabolism of the species, and the associated growth, of *S. alveolata* increases with temperature, with a plateau at 20°C. Below 5°C growth becomes constrained; worms often die during prolonged periods of low temperatures (Egerton, 2014). Mass mortality occurs during exceptionally cold winters (Holt et al., 1998).

S. alveolata is a filter feeder and requires suspended food particles. It is therefore associated with exposed coastal conditions and turbulent high current velocity waters created by wave or tidal actions, which transport sufficient food. The species also requires sand as tube building material. Both the food supply and sand particles are needed for the colonisation and growth of *S. alveolata* and development of reefs (Egerton, 2014).

***Ostrea edulis* Linnaeus, 1758, the flat oyster**

The bivalve *O. edulis* (see Figure 10) is found from the low intertidal shore down to sublittoral zones throughout the Atlantic and Mediterranean coasts of Europe. It is the UK's native oyster species. *O. edulis* is a protandrous alternating hermaphrodite species, meaning in its lifecycle it is first male and, when older, the oyster alternates between female and male functions (Laing et al., 2005). In temperate UK waters oysters reach sexual maturity in the third summer after settlement (Kamphausen et al., 2011; Korringa, 1952). Females brood fertilized eggs and larvae in their mantle cavity for 6 - 15 days, until the larvae have a fully formed shell of about 0.17mm (Hedgecock et al., 2007; Newkirk and Haley, 1982, Andrews, 1979; Orton, 1927; Walne, 1974). The larvae are then released into the water column.

Figure 10: *Ostrea edulis* cemented to intertidal boulder.

(Image Credit: AER)

Oysters usually spawn between late June and mid-September and remain dormant during winter; eggs or sperm are formed in spring (Hedgecock et al., 2007; Kennedy and Roberts, 1999). When released into the water column larvae drift in the plankton for approximately two weeks. Larvae then develop a “foot”, which enables them to settle on firm surfaces, followed by metamorphosis into a fully formed juvenile oyster (Laing et al., 2005; Sobolewska and Beaumont, 2005).

Although *O. edulis* have been shown to settle on a variety of substrates, including hard silt, muddy gravel with shells, sand and rocks, larvae favour other oyster shells (Korringa, 1946; Airoldi and Beck, 2007). Broodstock oysters are therefore used to attract larvae in restoration projects. Recruitment of *O. eduli*s is, however, sporadic and varies with environmental and physiological factors. Populations undergo natural phases of expansion and contraction. Successful recruitment appears to vary between one to three years (Loch Ryan, Scotland), or even every 6-8 years (Lough Foyle).

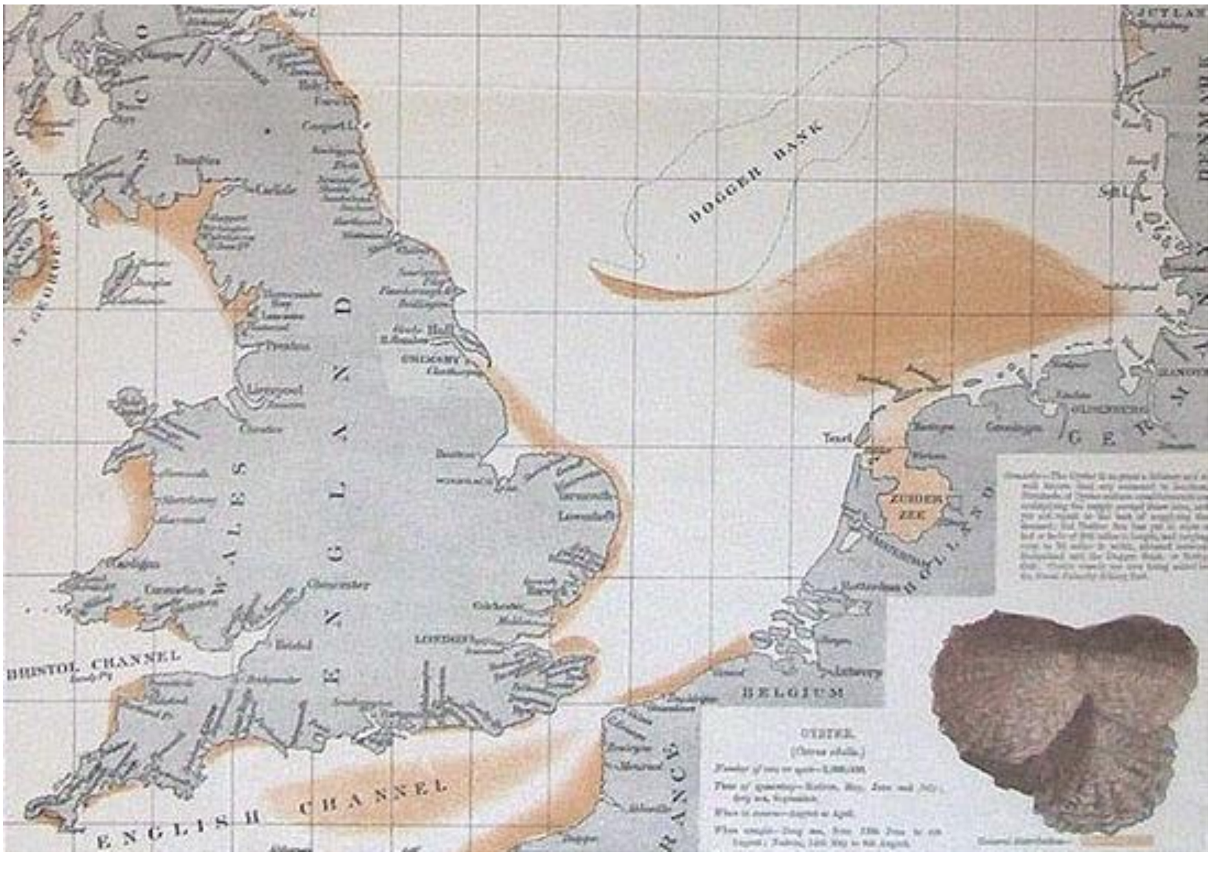
While oysters tolerate a spectrum of environmental conditions, factors such as temperature, salinity, food availability and hydrodynamic conditions affect growth and morphology (Andrews, 1979). Reproduction is driven by temperature and *O. edulis* require 8–9°C to start growth (Korringa, 1957; Loosanoff, 1962; Wilson and Simons, 1985, Laing et al., 2005). The species is able to survive in a wide range of salinities (18-40), although low salinity may inhibit feeding. Similar to other bivalves, they inhale water and filter it through a gill chamber, thereby removing suspended food particles. Growth of oysters depends on food availability, and microalgae or organic matter are important nutritional sources (Grant et al., 1990). Although oysters are adapted to turbid waters, high concentrations of suspended inorganic particles and sediment can result in reduced feeding efficiency. Given the impact of environmental conditions on the physiological and reproductive condition of *O. edulis,* these need to be considered in restoration projects for their long-term success.

Current Status of *S. alveolata* & *Ostrea edulis*

For *S. alveolata*, JNCC (2008) report that there had been '*a significant contraction in range on the south coast of England over a period of at least 20 years until 1984*'. In England, declines have also been reported in the western part of the north Cornish coast, the upper parts of the Bristol Channel and in the Dee Estuary. Causes had not been postulated and it was considered difficult to assess the true significance of these changes given the natural variability of the species.

Ostrea edulis has been harvested and cultivated in Europe since the Roman Empire (Gunther, 1897). The Piscatorial Atlas of the North Sea from 1883 highlights an area of 24,000 km^2 as "oysters" (see Figure 11). Stocks of the bivalve declined throughout their entire geographical range due to over-exploitation and consequently low rates of recruitment, as well as declining water quality due to industrial and municipal effluents (Edwards, 1997, Mackenzie et al., 1997, Tubbs, 1999). Episodic extremely cold winters in the 1960s and 1970s diminished oyster beds, and during the past 40 years, the production of flat oysters has additionally been negatively affected by the parasites and invasive species (Baud et al., 1997; Harding, 1996; Utting and Spencer, 1992). Landings of native oysters in Europe started to decline in the early 18th Century (Grizel and Heral, 1991), and annual landings in England fell from more than 2,000 tonnes in the 1920s to a few hundred tonnes by the early 1990s (Laing et al., 2006). Today, most of the remaining UK populations are situated on the Scottish west coast, the south-east and Thames Estuary, the Solent and the River Fal (Jackson and Wilding, 2007).

Figure 11: *Ostrea edulis* beds during the 19th Century (from Olsen, 1883).



3.3.2 Biogenic reef creation and restoration techniques

Given the ecological value of *O. edulis* and *S. alveolata*, attention focuses on both species not only for conservation, but also for biodiversity enhancement and restoration projects. However, successful projects remain sparse.

Sabellaria alveolata **reefs**

To date there have been no published substantial restoration initiatives for *S. alveolata* (Reach et al., 2015). The company Tidal Lagoon Swansea Bay supported an academic MSc thesis project by Swansea University in 2014, which trialled the translocation of boulders covered with *S. alveolata* from a donor site that would have been negatively affected by the lagoon development to a receptor site. The results of the initial pilot study showed that, in general, the translocated *S. alveolata* survived at the receptor site and reefs even appeared more vigorous five weeks post translocation. Given that this pilot study was a 6-month student project, there is no information about the sustainability and longevity of the intervention. While the project highlighted the possibility of direct translocation of *S. alveolata*, any considerations and discussions of restoring reefs are at an early stage. Value for money needs to be considered, since direct translocation of substantial reefs is likely to be costly, with a high risk of failure.

Since artificial structures and materials seem to be readily colonised by *S. alveolata* (see Figure 12), creating new reefs by providing suitable settlement substratum in close proximity to existing reefs is likely to be feasible (provided environmental conditions, including hydrodynamics and food supply are also suitable). This approach, together with other concepts for the restoration of such reefs, would likely need further research before they could be recommended to a developer.

(Image Credit: AER)

Figure 12: *Sabellaria alveolata* colonising artificial material in intertidal area.

Summary of *Sabellaria alveolata* restoration experience

In summary, there is essentially no experience of *S. alveolata* restoration in the UK, although the species readily forming reefs on artificial structures would indicate such restoration to be feasible provided suitable environmental conditions and larval supply are in place. There is thought to be only one analogous example in the literature base where another reef building polychaete worm has been translocated, which was unsuccessful (Reach et al., 2015).

Ostrea edulis

Oysters are one of the most commercially attractive marine species which has motivated restoration efforts throughout Europe for centuries. However, despite many attempts to restore *O. edulis* populations the species remains in a precarious state (Kamphausen et al., 2011).

Restoration techniques can be summarised as follows:

- Re-laying of adult oysters;
- Re-laying of spat (very young oysters); and
- Provision of shell cultch (substratum for larvae to settle) directly on the sea bed.

On natural oyster beds, settlement surfaces include the shells of living and dead oysters, other shellfish and other hard substrata such as stones and wood. Old bivalve shells are often added as cultch in managed fisheries to encourage Native Oyster settlement, with oyster, scallop, slipper limpet and mussel shells apparently providing particularly viable surfaces (Laing et al., 2005; Key and Davidson, 1981).

While maintaining or enhancing seabed habitats to support spat settlement is an essential starting point for Native Oyster restoration, a viable broodstock must also be present if the spat are to be produced in the first place (Reach et al., 2015). Where there are few adults, or where the adults are too dispersed, regular successful spawning may be impaired and broodstock enhancement may thus be needed. Where available, this enhancement could take the form of aggregating adults by collecting them from the wild and depositing them together in specific locations. Alternatively, when available, broodstock may be sourced from hatchery stocks. Both approaches have merit, but moving wild shellfish from one site to another increases the risk of introducing disease or non- native species to an area. Reliance on hatchery sourcing also increases the risk of limiting the genetic diversity of the population (Laing et al., 2005).

Historically, European restoration projects were based on relaying of parental stocks, but the method has not stopped the decline of *O. edulis* populations. It also resulted in the depletion of some donor stocks, for example in the Essex beds and in the Firth of Forth (Key and Davidson, 1981).

In France, early restoration efforts created ‘oyster parks’, which were based on the settlement preferences of oyster larvae. Shell cultch, including adult oysters was placed on the seabed. This approach was moderately successful, although sedimentation may have compromised spat settlement and survival of new recruits (Yonge, 1966).

In Ireland, bed rotation has been trialled, i.e. the collection of spat on cultch for seeding, which is then transferred to other suitable areas, for example in Tralee Bay. In 1991, 250 bags with 1,000 spat each on native oyster cultch from Tralee Bay were transplanted to Lough Swilly and grown on trestles for over a year. They were subsequently seeded onto the sea bed. These projects increased the catch rates the following years, but limited spat supply and the challenge of relocating oysters whilst minimising the spread of diseases remain significant challenges (OSPAR, 2009).

In 1997, the EU funded an oyster relaying project in Strangford Lough, Northern Ireland in collaboration with the local fishing community (Kennedy and Roberts, 2001, Laing et al., 2005). Cultch, seeds and adult oysters were placed at 9 sites. The oyster population increased from 100,000 individuals in 1998 to 1.2 million individuals in 2003, but stock levels were not sustained due to unregulated harvesting and infestation by the Bonamia *Ostreae* parasite, leading to a decline to 650,000 individuals by 2005 (Smyth et al., 2009).

In South Wales, 40,000 adult oysters from Loch Ryan were placed in an un-fished area of Swansea Bay in 2013 at densities of 10 individuals/m^2 in a total area of 2 ha (Gravestock et al., 2014). The site was selected on the basis of historic oyster bed locations, the presence of existing oysters and substrates for larval settlement; 4-5 tonnes of cockle shells were laid on the sea bed as cultch materials. Beds were monitored for oyster growth, mortality and predators, but to date there is no published evidence that the attempt was successful.

In England, numerous *O. edulis* restoration projects have been conducted in The Solent, including in Chichester Harbour (Vause, 2010; Eagling, 2012) and Southampton Water. With regard to the latter, several attempts were carried out in the northern part of Stanswood Bay with the aspiration to increase the larval supply to surrounding areas using re-laying of cultch and deposition of broodstock oysters on the sea bed. In 1973 the Stanswood Bay Fishermen’s Cooperative planted 375 tonnes of multiple types of mollusc shells on the sea bed to enhance settlement, and it was estimated that the minimum benefit of this project to oyster fishery was 10.5 tonnes of saleable oysters (Key and Davidson, 1981). A second restoration programme was conducted in the same location in 1999, where cultch, consisting of scallop and whelk shells, was planted in the area (Laing et al., 2005). In 2003 and 2004, seed oysters were added in fine sacks. Seed oysters grew from 5 mm to almost 7 cm in six months, but there is no information about the sustainability of the project (Laing et al., 2005). There have also been various recent trials in the Solent, including trials installing suspended cages in various marina and pontoon locations, as part of the wider Solent Oyster Restoration Project (Solent Forum, 2018).

In 2010, the Chichester Harbour Oyster Partnership Initiative was established through the cooperation between Sussex and Southern IFCA, Chichester Harbour Conservancy, Natural England and the local fishing community (Vause, 2010). The purpose of this collaboration was to survey and manage the *O. edulis* stock in Chichester Harbour with academic support from Cefas and the National Oceanography Centre (Eagling, 2012). 2,298kg of broodstock oysters were relayed on the sea bed at a density of 40m^{-2}. Oysters reproduced successfully up until the spawning of larvae, but the sex ratio (male: female) of the broodstock was 3:1, differing significantly to what was naturally expected (1:1). Two years after relaying, an increased mortality of the relayed oysters was reported, especially after the spawning season, and one third of the population had died post spawning. It appears that the environmental conditions at the seabed might have negatively affected oyster physiology, reducing growth and leading to increased mortality.

Other recent initiatives are also noteworthy. These include the Essex Native Oyster Restoration Initiative, whereby more 25,000 mature native oysters were re-laid in the Blackwater, Crouch, Roach and Colne Marine Conservation Zone (MCZ) in 2016

(Essex Wildlife Trust, 2016). A recently initiated Scottish trial is also of interest. In the Dornoch Firth, work has begun to restore oyster reefs which were fished to extinction over 100 years. Here, shell clutch has been provided, and about 20,000 oysters are being placed on this in a grid formation (see Figure 13). The aim is for the reefs to become self-sufficient and sustain 4 million oysters in a 40 ha area (BBC, 2018).

Figure 13: Divers placing oysters on shell clutch in the Dornoch Firth (from BBC, 2018).

Summary of *Ostrea edulis* restoration experience
In summary, current restoration efforts based on re-laying of adult oysters and shell cultch directly on the sea bed seem insufficient to solve the precarious status of *O. edulis*. Therefore, to enhance and restore the flat oyster populations, new management approaches need to be identified and tested. Sustainability and resilience of the restored *O. edulis* beds appears to be the main challenge.

3.4. Ecological enhancements

This section provides a brief overview of possible enhancement measures in the marine environment. Within the marine environment, ecological enhancement needs to be considered in the wider context of marine ecosystems. In contrast to terrestrial ecosystems, marine systems are more open and dynamic with ecosystem processes generally working over much larger spatial scales and at a range of temporal scales. There is a considerable body of research that has explored ecological enhancement of marine structures, particularly, for example, rock armour defences and some vertical wall structures (piles, quay walls), although the actual application of research ideas has often been more limited. In the UK to date, ecological enhancement has tended to be undertaken at very local, small, scales. Examples include:

- Creation of artificial rockpools and textured surfaces to enhance colonization (e.g. porous blocks) (e.g. Firth et al., 2016; Naylor et al., 2017, 2011);
- Creation of artificial reefs (e.g. Fabi et al., 2011);
- Translocation and placement of shingle/cobble to create biodiverse scar-skear habitat (e.g. ABPmer, 2006);
- Bioengineering erosion protection (e.g. River Severn (Longney) case study cited by Environment Agency, 2009);
- Use of timber battens on sheet steel piled quay walls to promote colonisation (e.g. Deptford Creek case study cited by Environment Agency, 2009);
- Incorporation of rock pools into quay wall designs (e.g. Wightlink, 2018);
- Installation of tern rafts (e.g. RSPB, 2017, 2018a) and other bird nesting/roosting structures such as Kittiwake nesting shelves on structures;
- Management of shingle island and rafts to keep vegetation clear for tern species (RSPB, 2013);
- Management of saltmarsh to increase diversity and bird use (e.g. including scrapes and mounds for feeding and roosting); and
- Creation of vegetated urban terraces, amongst others to provide refuges for estuarine fish (e.g. Greenwich case studies cited by Naylor et al., 2017 and Environment Agency, 2009);

4. The datalayers

This section provides background information on the 6 datalayers which were created for this project. It is structured according to the habitats which formed the focus of the datalayer tasks, namely:

- Mudflat and saltmarsh datalayers (Section 4.1);
- Seagrass meadow datalayer (Section 4.2); and
- Biogenic reef datalayers (Section 4.3).

For each datalayer, the methodology for its creation is summarised, and broad guidance for its use provided, as well as limitations / caveats discussed.

The datalayers which were created are summarised in Table 2.

Table 2: Summary of datalayers created for this project.

Habitat	Datalayer name	Description	Main creator	Main input datalayers
Mudflats and saltmarshes	Potential habitat creation sites within the current floodplain	Currently defended floodplain areas which could be suitable for managed realignment and / or RTE. Polygon data.	ABPmer	Environment Agency tidal and tidal/fluvial floodplain
	Potential beneficial use (mud) – stretches which may benefit	Stretches of SSSI-designated low-energy shorelines thought to be eroding due to coastal squeeze. Line data.	ABPmer	Natural England SSSI Units, OS Mean High water Springs (MHWS) line
	Potential beneficial use (mud) - potential material sources (maintenance dredge disposal sites)	Review of disposal site data, highlighting those from which materials suitable for mudflat or saltmarsh restoration could potentially be diverted. Point data.	ABPmer	Cefas disposal sites
Biogenic reefs	Potential *Sabellaria alveolata* restoration - historic and current sites	Historic and current sites where *Sabellaria alveolata* or *Ostrea edulis* records have been logged in the OBIS database. Extracted by AER and classed according to age of record by ABPmer. Point data.	OBIS	OBIS (a global open-access database on marine biodiversity)
	Potential *Ostrea edulis* restoration - historic and current sites			
Seagrass beds	Potential seagrass creation / restoration – historic sites	Sites where seagrasses are either no longer present, or highly degraded, and where restoration may be beneficial. Point data.	AER	None

The methodology for the creation of each of the datalayers was confirmed and refined during a dedicated workshop, which was held in London on 14 June 2018. In addition to the consultancy team (ABPmer, the RSPB and AER), representatives from the MMO, Natural England, and the Environment Agency attended this workshop, to help provide a steer on the methodology. Draft versions of these layers were presented to the organisations present during the June 2018 workshop, and at a results workshop in October 2018[1]. At the results workshop, no changes to the datalayers were requested; the main actionable comments related to the need for a clear summary of caveats and limitations, which is provided in the relevant sections in the 'results and limitations' sub-sections in Sections 4.1 to 4.3. Each of the datalayers has been delivered to the MMO as an Esri compatible GIS file (ArcGIS 10.2), accompanied by a detailed processing log, as well as MEDIN -compliant metadata. These datalayers will be made available for viewing in MIS and Data.gov.uk.

4.1. Mudflat and saltmarsh datalayers

Mudflat and saltmarsh creation can be achieved through a number of mechanisms, with the main techniques being managed realignment, RTE and beneficial use. In this section, the datalayers are discussed in relation to these techniques. Firstly, the datalayer related to the managed realignment and RTE is discussed (Section 4.1.1), before the two datalayers development in relation to beneficial use are described (Section 4.1.2). Within each section, in order, are the methodology for the creation of the datalayer, results and limitations and then high-level guidance (on its use and further steps which may be required)

4.1.1 Managed realignment and Regulated Tidal Exchange – datalayer showing 'Potential habitat creation sites within the current floodplain'

Methodology

This polygon datalayer was created to show where intertidal habitats (mainly mudflats and saltmarshes) could be created in the current floodplain. It highlights areas where the techniques known as 'managed realignment' and/or RTE (see Section 3.1.2) could be utilised to inundate land which is currently defended.

The creation of this datalayer was a very resource-intensive process, which is fully described in the detailed processing log delivered with the datalayer, and also outlined in the metadata information. In summary, the following main steps were undertaken to create this layer:

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[1] Noting that no representatives from Natural England could attend the second workshop in October 2018.

- Use of the Environment Agency's 'Flood Zone 3'[2] polygon datalayer as the basis for the datalayer;
- From the base datalayer, only those areas / types labelled as 'tidal' and 'tidal/fluvial' were then taken forward (excluding / deleting floodplains classed as 'fluvial' only);
- From this base datalayer, the following areas were then deleted / excluded using a mixture of automated and manual processes:
  - Floodplain areas more than 10 km from the coast (or more than 2 km from the banks of constrained tidal rivers and estuaries);
  - Urban areas, and major road and rail infrastructure;
  - Industrial sites, significant residential areas, and caravan parks;
  - Sites which are less than 10 ha in size;
  - Sites which are not currently defended (from flooding - by an embankment, seawall, or similar);
  - Sites which have already been realigned (i.e. existing managed realignment / RTE sites);
  - Sites which are deemed too exposed, or of the wrong substrate (i.e. open coast / beaches, shingle, dunes); and
  - Disconnected floodplain areas (cut off by urban areas, major roads (motorways and category A roads) and major railways).

Various datalayers were used to aid this process, including Ordnance Survey (OS) Mastermap products, the OS Mean High Water Springs (MHWS) line, aerial imagery provided by Esri, as well as the Environment Agency's national defence and saltmarsh datalayers.

It should be noted that international and national conservation designations were not used to exclude sites, neither were greenfield sites or golf courses. Sites with scattered settlements containing less than approximately 20 buildings were also not taken out of the dataset. Limitations of the datalayer are discussed below.

The attributes table[3] for the GIS datalayer was populated with the following additional / background information for each of the identified sites:

- Site location (estuary / coastal stretch);
- Size (ha);
- Potential habitat types;
- Potential techniques;
- Percentage of site which is internationally or Sites of Special Scientific Interest (SSSI) designated; and
- Shoreline Management Plan (SMP) policy (second round, 1st and 2nd epochs).

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[2] This datalayer displays 'land assessed as having a 1 in 100 or greater annual probability of river flooding (>1%), or a 1 in 200 or greater annual probability of flooding from the sea (>0.5%) in any year' (Environment Agency, 2015).

[3] An ArcGIS attributes table resembles an Excel worksheet, and displays information on spatial features of a selected layer. These feature characteristics are contained in columns. Attributes such as size can be easily calculated in ArcGIS, and overlap analysis with other datalayers can help gain additional information (e.g. designations, as undertaken for this datalayer); furthermore, columns can be manually added or imported from other databases.

Validation

The Flood Zone three datalayer was chosen as the base layer, as it was considered to provide a good indication of suitable elevation in the tidal frame. The only other conceivable alternative to this, the interrogation of LiDAR (Light Detection And Ranging) data in conjunction with tidal level interpretation, was considered too data/resource intensive on a national scale. The methodology outlined above was applied whilst anticipating an overestimation of intertidal habitat creation potential, due to the Flood Zone 3 layer incorporating land of an elevation with up to a 1:200 annual probability of flooding (please see Section 3.1.1 for a discussion of tidal levels generally associated with saltmarshes and mudflats). Thus, it was expected that the sites would contain some areas which are too high for intertidal habitat creation (at least without further intervention, notably excavation). In order to provide an indication as to how much the layer essentially over-estimates the potential for intertidal habitat creation, a 'LiDAR validation' exercise was undertaken for seven representative sites around the country (with the locations agreed at the June 2018 workshop). The results of this exercise are shown in Annex A.

In summary, this validation exercise confirmed that, generally, using the Floodplain 3 datalayer would lead to the identification of sites which could support mostly intertidal habitat if they were breached, though actual habitats would depend on the regional and site conditions. The highest lying site among the validation sites was the site in Northumberland (see Figure A.24 in Annex A), where a large percentage of the site would not initially develop into mudflat or saltmarsh without intervention. However, at all the other sites, at least 90 % of intertidal habitat would initially be anticipated across the sites, with the lowest sites found in Essex and the Thames Estuary, and higher sites elsewhere. Thus, the exercise confirmed that the datalayer provides a good initial indication of whether or not intertidal habitat creation may be feasible at a given site, though further investigation is recommended once a site has been selected.

Results and limitations

This datalayer contains over 700 sites where intertidal habitat creation through managed realignment or RTE could potentially be undertaken. The size of these sites ranges widely, from 10 ha to over 10,000 ha (noting that the largest managed realignment site in the UK to date measures 370 ha[4]). Such large sites are unlikely to ever be required, nor would it be likely that undertaking managed realignment or RTE across the whole extent of such a large site would be feasible. For example, in certain areas of a given estuary (particularly in narrower confined upper reaches), undertaking too big a scheme would likely be unduly detrimental to the adjacent system, and thus be undesirable. Thus, in reality, large sites could be sub-divided, and those areas closer to the shoreline preferentially targeted. In practice, an organisation wishing to undertake managed realignment in a given area would be guided by available resources and required/desired size and type of habitats with regard to what site, or section of a given site, might be most suitable.

The datalayer created for this project shows sites with theoretical potential for managed realignment or RTE; it has not ranked areas according to their suitability, nor does it indicate whether or not land is actually available for intertidal habitat

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[4] Alkborough on the Humber.

creation (e.g. landowners may well not be interested in selling). Given the national scale of the datalayer, there may also be areas which may have changed (e.g. built on) since the mapping consulted to inform the exclusion / deletion exercise have been created. In summary, the datalayer should very much be seen as an aide to initiating a search for a potential site, rather than the sole tool for identifying a suitable site.

The datalayer contains sites which lie behind minor roads and railway lines (defined as category B roads or less, and single-track railways). Without realigning these structures, RTE would be the only technique which could facilitate intertidal habitat creation. Depending on local conditions, this may be achievable using existing drainage outfalls (e.g. by retrofitting 2-way exchange gates), or by laying new pipes / culverts. The latter could be relatively costly. With RTE, it is also worth reiterating (as previously stated in Section 3.1.2), that tidal amplitude is significantly reduced, depending on the dimension of the exchange medium. For example, at the RSPB's Goosemoor RTE site on the Clyst (Devon), an on-site tidal variation of 1.3 m is achieved during spring tides (pers. comm., RSPB Goosemoor), whereas the unconstrained spring tidal range in the estuary itself is around 3.1 m. Thus, it is unlikely that the full extent of a given site included in the datalayer would be inundated if RTE were to be employed. Even for sites where managed realignment is being considered, and where water exchange would be relatively unconstrained, it should be noted that the longer the distance the water has to travel from the existing shoreline, the less water height it achieves (due to lag / resistance forces), and thus, where only some sections of a given potential site may be required, then those areas nearest to the shoreline should preferentially be chosen.

Guidance for use

This datalayer should be used as an initial aid at the start of a search for a potential site. The motivation for undertaking a given managed realignment or RTE scheme, or a suite of schemes, would determine the level of further investigation required. For example, developers needing to find a site with specific habitat requirements would need to take a different approach to a non-governmental organisation (NGO) wanting to create additional saltmarsh habitat at a site it was considering to acquire.

Some local knowledge, and at least a high-level understanding of the system in which managed realignment is to be undertaken, are beneficial during site selection exercises. In this respect, it is worth highlighting Natural England's Healthy Estuaries project, which aims to address coastal squeeze in a selection of designated English estuaries, and, in the process, identifies areas where managed realignment might be most appropriate in a given estuary[5]. Other searches which may have previously been undertaken in a given region should also be consulted, if accessible. This includes those undertaken by the Environment Agency in relation to Regional Habitat Creation Programmes and Flood Risk Management Strategies. The RSPB's (2018b) Sustainable Shores reports are also of note[6].

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[5] Published report available for 7 designated sites to date, with more in development: http://publications.naturalengland.org.uk/publication/4734703644966912 [last accessed November 2018]

[6] In its Sustainable Shores project report, the RSPB set out how historic and ongoing marine habitat loss could be addressed through habitat creation and restoration. Read more https://www.rspb.org.uk/our-work/conservation/projects/sustainable-shores/ [last accessed February 2019]

It is beyond the scope of this project to provide detailed advice on how to undertake a site selection exercise and initial feasibility review for managed realignment and RTE sites. Such guidance can, for example, be found in Nottage and Robertson (2005) and Leggett et al. (2004). However, in summary, the following broad steps outline those which may be required when undertaking a search for a potential site, and highlight how the datalayer produced for this project could fit into such a search process. The order, and need for, some of the steps would to a large extent depend on the motivation for a given search.

- Identify sites broadly suitable for managed realignment or RTE in the region of interest (using the datalayer created for this project);
- Create a list of most desirable sites (e.g. in terms of vicinity to damaging development);
- Obtain LiDAR data to create a digital elevation model for some, or all of, the selected sites, and relate the elevation to tidal levels to gain a high level understanding of potential habitats (see Section 3.1.1 and Annex B);
- Refine site boundaries to reflect field / ownership boundaries;
- Undertake a site characterisation process for short listed sites, investigating constraints, opportunities and potential habitats, by researching aspects such as: existing nature conservation designations, shoreline management policy, cultural heritage records, contamination, land use and infrastructure, drainage outfalls, need for landward defence, likely impacts on fronting system (e.g. by calculating a high level tidal prism value[7]), etc. (consult local knowledge if possible);
- Rank the sites according to suitability and project specific drivers; and
- Make enquiries with landowners (if not already the owner).

The attributes table in the datalayer produced for this project can be used to inform the initial search for a potential site by, for example, only displaying sites in a given estuary, providing an indication of how much of a given site is designated, and what the short and medium-term shoreline management policies are.

Once a site, or a set of sites have been identified as the preferred site(s), further studies would generally be required; these would typically involve the commissioning of specialist consultancies, and may include the development and assessment (including numerical modelling) of preliminary designs.

4.1.2 Beneficial use datalayers

Two datalayers were created in relation to beneficial use, a technique generally used to restore degraded intertidal habitats, although it has the potential to create intertidal habitat from subtidal.

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[7] The tidal prims of a site essentially refer to the volume of water exchanged over a given tide. Gaining an understanding of the tidal prim of a given potential site in relation to that of the adjacent waterbody can be used as a key indicator of the likely scale of effects and viability of a site. Such a value is obtained by using elevation data and calculating the fill volume, typically to MHWS (GIS exercise) (see, for example, Leggett et al., 2004).

Methodology

I. Beneficial use (mud) – stretches which may benefit

This line datalayer was created in order to highlight stretches of low energy shorelines where intertidal habitats are currently present and appear to be eroding due to a process known as 'coastal squeeze'[8]. It was decided to use the site condition reports for coastal and estuarine Sites of Special Scientific Interest (SSSIs) as a proxy to identify such stretches. Those sections of littoral and supralittoral sediment habitats whose condition is considered unfavourable due to coastal squeeze/erosion were collated, as these could presumably benefit from intertidal recharge.

In the future, saltmarsh erosion layers which are being produced by the Environment Agency in order to monitor saltmarsh extent in waterbodies designated under the Water Framework Directive Regulations 2017 could be utilised; however, these are still under development.

A number of spatial datasets were used to undertake this task and were obtained from the OS and the Environment Agency spatial data catalogue, these included:

- The OS MHWS line as a representation of the coastline;
- The current SSSI sites and SSSI units for England; and
- The current saltmarsh extents for England (Environment Agency layer).

The Environment Agency's saltmarsh extents layer was used to highlight those SSSIs which contained saltmarsh, and for which condition reports would need to be reviewed.

Subsequently, a desk-based review of the individual condition unit reports published on Natural England's 'Designated Sites View' was undertaken for each of the identified sites. All units which were in unfavourable condition due to intertidal habitat erosion or coastal squeeze were determined and mapped.

A line datalayer was consequently created, which contains the following columns in the attributes table (populated from Natural England's SSSI units shapefile):

- SSSI name;
- Unit ID;
- Unit status (as of October 2018); and
- Condition date (i.e. date of last assessment).

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[8] Coastal squeeze is commonly understood as the 'process by which coastal habitats and natural features are progressively lost or drowned, caught between coastal defences and rising sea levels' (Defra, 2003).Please note that this definition may be considered to be overly simplistic, as it implies that coastal narrowing results entirely from cross shore processes associated with sea level rise. Other factors which can cause a reduction in the width of the coastal zone include shifting offshore banks / channels, changes in the wind–wave climate, etc. (Pontee, 2011).

II. Potential beneficial use – potential material sources (maintenance dredge disposal sites)

This point datalayer serves to indicate regions where significant volumes of fine materials could regularly be available for beneficial use to restore mudflats and saltmarshes. It provides an indication of fine material availability at open marine disposal sites in England and Wales, by providing background information on ongoing maintenance dredge activities. Welsh disposal sites were included as these are often in relatively close proximity to English shorelines, and materials deposited there may have originated in England (and be available for beneficial use in England).

This datalayer was created by undertaking the following steps:

- Use of the Cefas 'disposal site' polygon layer as the basis for the datalayer;
- Interrogation of the latest version of the Defra 'Disposal At Sea (DAS)' database (2014 version, compiled by Cefas, which contained data for the period 1986 to 2012) to:
  - Identify disposal sites which regularly received maintenance dredge material (i.e. those that are used at least once a year);
  - Delete all other sites from the base datalayer (i.e. all closed sites and all sites which do not regularly receive maintenance dredge materials);
  - Calculate average annual volume of sediments received at each sites (this data was captured in the attributes table under the column heading 'Dredge volume[9]');
- Use review of available information/data (e.g. maintenance dredge protocols, technical reports, news articles etc.) and expert knowledge to:
  - Identify the predominant material type likely to be received at each disposal site (mud, sand, shingle) (adding this to attributes table under the column heading 'main materials');
  - Identify the expected primary distributors (i.e. name of port / marina) (adding this to attributes table under column heading 'key depositors'); and
  - Classify whether the material disposed of is likely to be suitable for mudflat/ saltmarsh restoration / creation (deemed to be mud or sandy mud) (adding a 'Suitable for restoration?' column to attributes table).

These aspects are displayed in the respective attributes table that accompanies the datalayer, as is information on whether the disposal site is in an estuary or along the coast (to inform proximity).

Results and limitations

I. Beneficial use (mud) – stretches which may benefit

This datalayer, which identifies stretches of coast which may benefit from beneficial use, highlights shorelines which are currently believed to suffer from erosion. Some 1,690 km of English coastline are highlighted in the final datalayer. Locations identified included:

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[9] Noting that, in ArcGIS attributes tables, column headings have a limit of 10 characters (which the programme automatically shortens a longer name to), and that spaces are replaced with underscores.

- Parts of the southeast coast extending approximately from the Thames Estuary up to The Naze in Essex;
- Most of the Solent, including parts of Southampton Water, and Portsmouth, Chichester and Langstone Harbours;
- Sections of the Severn Estuary and Inner Bristol Channel; and;
- Along parts of the River Humber.

Limitations of this datalayer include:

- It only highlights SSSI-designated shorelines (although around 90 to 95% of England's soft shorelines are SSSI designated[10]);
- It relies on site condition assessments by Natural England officers, which may be subjective, and dated (with some of the assessments being up to 8 years old). It may be prudent to update the datalayer when new/additional data becomes available.

Despite these limitations, from known historic trends (e.g. Environment Agency, 2011), and feedback during the results workshop, it appears that the datalayer successfully identifies the main eroding soft/intertidal shorelines in England.

II. Potential beneficial use – potential material sources (maintenance dredge disposal sites)

This datalayer shows at which marine disposal sites, and from which ports, muddy materials may be available for use in beneficial use schemes from current maintenance dredge activities. The datalayer contains information on 81 English and Welsh disposal sites which regularly receive large quantities of maintenance dredge materials.

The main limitations of the datalayer include:

- It only provides a high level indication of where, and how much, sediments may be available;
- The review is based on a database which contained records up to 2012, so trends may have changed since then; and
- For some sites, primary depositors could not be identified.

Despite these limitations, the datalayer presents spatially referenced data identifying where sediment could potentially be available, and indicates trends which are likely to continue into the future, as maintenance dredge requirements do not tend to fluctuate substantially.

Guidance for use

The two datalayers developed in relation to beneficial use are intended for use as initial aides in a search for a potential site. Some local knowledge and at least a high level understanding of the system in which beneficial use is to be undertaken are highly beneficial during site selection exercises. Two particular related studies are

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[10] Based on an ABPmer comparison of the Environment Agency saltmarsh layer and Natural England's SSSI layer. Notable exceptions include several Cornish estuaries, such as the Camel, Looe and Fowey; and parts of the upper Thames and Medway Estuaries.

worth noting, including highlighted in Section 3.1.2, namely the RSPB's SEABUDS project, as well the Solent Forum's BUDS project.

It is beyond the scope of this project to provide detailed advice on how to undertake a site selection exercise and initial feasibility review for beneficial use sites. Such guidance can for example be found in ABPmer 2017a and 2018. However, the following bullets summarise the broad steps which are likely to be required when undertaking a regional search for a potential site, and highlight how the datalayers produced for this project could fit into such a search process (with the order, and need for some of the steps, to a large extent depending on the motivation for a given search):

- Identify eroding soft shorelines in the region of interest (using the 'stretches which may benefit' datalayer created for this project, whilst also loading available datalayers showing existing mudflats and saltmarshes[11]);
- Identify sediment sources for beneficial use in the area of interest (using the 'potential material sources (maintenance dredge disposal sites)'[12] datalayer created for this project');
- Contact organisations which may have suitable (muddy) sediments in close proximity of the area of interest to confirm availability / timings / willingness to provide / fees (based on 'primary distributors' identified in the 'beneficial use – potential material sources (maintenance dredge disposal sites)' datalayer)[13];
- Employ / consult local knowledge to determine particularly suitable sites along the broad stretches of shoreline identified (e.g. those in a particularly poor condition, or those which front vulnerable sea defences which could benefit);
- Create a list of most desirable sites (e.g. in terms of vicinity to damaging development);
- Obtain bathymetry and LiDAR data to create a digital elevation model for the sites, and relate the elevation to tidal levels to gain a high level understanding of potential habitats and accessibility by dredging vessels;
- Undertake a site characterisation process for short listed sites, investigating constraints, opportunities and potential habitats, by researching aspects such as: existing nature conservation designations, shoreline management policy, distance to dredge site[14], value of hinterland, value of site for fisheries, level of recreational boating / mooring, etc. (consult local knowledge if possible);
- Rank the sites according to suitability and project specific drivers; and
- Consult the landowner / The Crown Estate.

Once a site, or a set of sites have been identified as the preferred site(s), further studies would generally be required; these would typically involve the commissioning

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[11] Where available. For example, the latest 'saltmarsh extents' datalayer can be obtained from the 'data.gov.uk' website.

[12] Maintenance dredging campaigns tend to yield high percentages of fine sediment, and rarely coarser materials (as opposed to capital dredging campaigns, where the opposite is typically the case (e.g. MMO, 2014).

[13] Noting that inquiries should also be made with regulators and ports about upcoming capital dredge campaigns which could yield muddy sediments.

[14] As distance travelled by a dredger can have a large impact on the cost of a beneficial use project, keeping distances as small as possible can be a key factor in making such a project possible (e.g. CIRIA, 2010).

of specialist consultancies, and may include the development and assessment (including numerical modelling) of preliminary designs.

4.2. Seagrass datalayer – ‘Potential seagrass creation / restoration - historic sites’

As shown in Table 2, a point datalayer was created in relation to seagrass meadows, which serves to indicate areas where seagrass meadows are either currently present, but very degraded, or where they may have previously existed, and where their restoration was perceived as potentially feasible.

Methodology

In order to create this point datalayer, a comprehensive literature review was undertaken, and broad locations identified. The literature review was based on a number of extensive and detailed reviews of seagrass loss in the UK; as well as a review of historic records of seagrass recorded by numerous biological recording programmes and individuals that date back to the 1900s and 1800s. The attributes table for the datalayer displays the following information:

- County;
- Type of seagrass (subtidal versus intertidal);
- Whether there is the potential for seagrass restoration through the replacement of moorings;
- Species of seagrass to target for restoration (*Z. marina, Z. noltii,* or both); and
- Key references.

Background information on these potential future restoration sites is provided in Annex B of this report.

Results and limitations

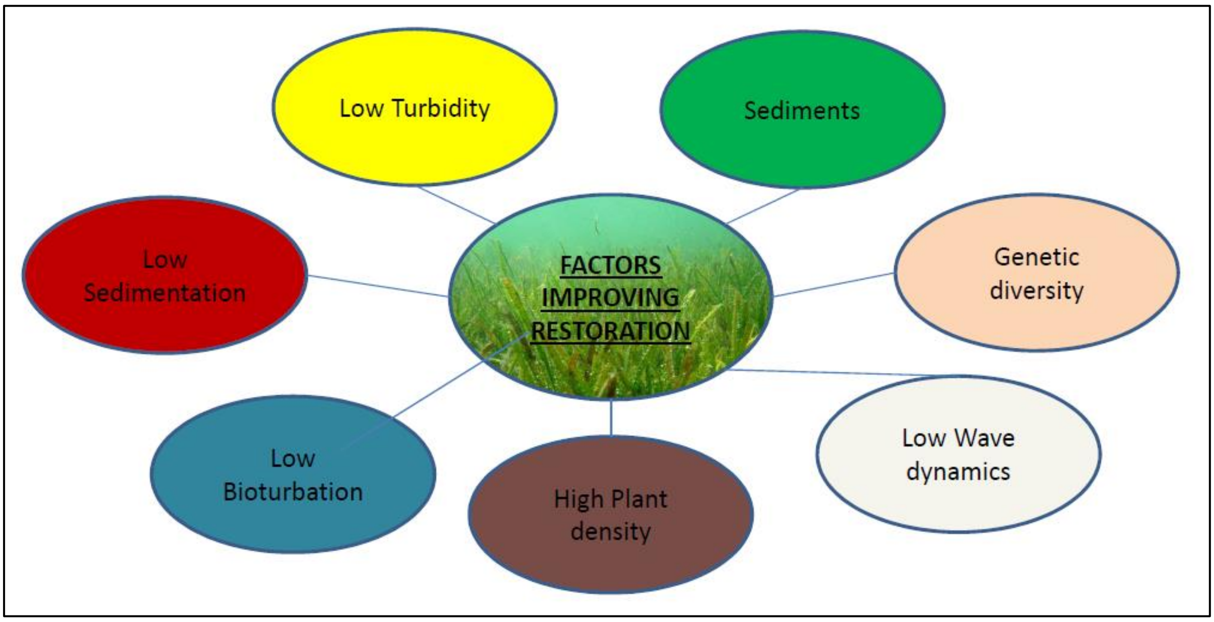
Sixty one restoration sites have been identified in England, where restoration is considered feasible. This datalayer identifies broad areas, and does not pinpoint exact sites where restoration / creation might be feasible. Furthermore, inclusion of a site in the datalayer does not imply that all environmental conditions required for seagrass (re)establishment will have been met at a given location (see Section 3.2.1). Most notably, water quality may (still) not be suited for seagrasses to (re)establish; this would require further investigation.

Guidance for use

This datalayer is intended as an initial aid in a search for a potential restoration site. As noted above in Section 3.2.2, to date, no full-scale seagrass restoration scheme has taken place in the UK, although there is considerable evidence from elsewhere in the world (e.g. van Katwijk et al., 2016), and site selection models have been developed, for example in the US (e.g. Short et al., 2002). It is furthermore noteworthy that the Environment Agency is currently developing a ‘seagrass suitability’ map for England based on modelling of environmental characteristics (pers. comm., Environment Agency).

Based on the review provided in Section 3.2.1, the main environment parameters which improve the likelihood of seagrass restoration success are summarised in Figure 14 below.

Figure 14: Key factors improving the likelihood of seagrass restoration success (From AER).



Whilst it is beyond the scope of this project to provide detailed advice on how to undertake a site selection exercise and initial feasibility review for seagrass restoration, based on the parameters shown above, and international examples, the following broad steps could be followed when undertaking a regional search for a potential site (with the order, and need for some of the steps, to a large extent depending on the motivation for a given search):

- Identify areas where there are existing and historic sites in the region of interest (using the ‘Potential seagrass creation / restoration - historic sites’ datalayer created for this project);
- Consult local knowledge to help distinguish historic from existing sites, and create a list of most desirable sites (e.g. sites where there are abundant historic records, and where there are still some habitats nearby; sites where water quality is very good);
- Undertake a site characterisation process for short listed sites, investigating aspects such as: turbidity, water quality, water depth, predominant sediment, proximity to existing seagrass habitats, wave exposure; existing nature conservation designations; levels of recreational boating / mooring, fishing, bait digging, etc. (consult local knowledge if possible);
- Rank the sites according to suitability and project specific drivers; and
- Consult the landowner (likely to be The Crown Estate, with some local exceptions).

Once a site, or a set of sites have been identified as the preferred site(s), further studies would generally be required; these would likely involve the commissioning of specialist consultancies or academic institutions, and may include the development and assessment of preliminary designs, as well as numerical suitability modelling. Availability of suitable donor seeds or seedlings would also need to be investigated.

4.3. Biogenic reef datalayers

As shown in Table 2, two datalayers of point data were created in relation to biogenic reefs, showing current and historic presence of *S. alveolata* and native oysters respectively.

Methodology
The methodology for the creation of the datalayers was essentially identical, and will thus be discussed in tandem below:

- Potential *S. alveolata* restoration - historic and current sites; and
- Potential *O. edulis* restoration - historic and current sites.

The datalayers were produced by downloading English records for the two species from the OBIS (Ocean Biogeographic Information System) database in October 2018. The OBIS database (http://iobis.org/) 'is a global open-access data and information clearing-house on marine biodiversity for science, conservation and sustainable development'. It emanates from the Census of Marine Life (2000-2010) and was adopted as a project under the United Nations Educational, Scientific and Cultural Organization's (UNESCO) Intergovernmental Oceanographic Commission Data and Information programme in 2009, and contains over provided over 45 million observations (as of November 2018).

The original OBIS spreadsheets were refined by deleting empty and duplicate fields (see processing logs for details). A column was added detailing the age of the records, as agreed during the workshop on 25 October 2018, to help identify areas where the species may no longer be present. Records more than 30 years old were attributed as very old / historic for the purpose of this datalayer. As the data contained spatial coordinates, it was loaded into the ArcGIS software and a spatial (point) datalayer created.

Results and limitations
The following disclaimer is noted on the OBIS database website (OBIS, 2015), from which the data contained in the two datalayers has been derived:

'Appropriate caution is necessary in the interpretation of results derived from OBIS. Users must recognize that the analysis and interpretation of data require background knowledge and expertise about marine biodiversity (including ecosystems and taxonomy). Users should be aware of possible errors, including in the use of species names, geo-referencing, data handling, and mapping. They should crosscheck their results for possible errors, and qualify their interpretation of any results accordingly.'

In addition to these limitations, it should be noted that records do not necessarily indicate the presence of a biogenic reef, but may merely show records of individuals. The presentation of a record does also not indicate that all environmental conditions are (still) suitable for biogenic reef creation.

Furthermore, other databases should ideally also be consulted, notably that of the National Biodiversity Network.

Guidance for use

These datalayers are intended as an initial aid in a search for a potential restoration site; with the rationale being that suitable sites in the vicinity of the shown points would likely still/again be able to support new, or more extensive, biogenic reefs. This is particularly the case as larval recruitment is thought to be a key condition for the successful restoration or creation of a site, as outlined in Section 3.3 for both species.

As noted above in Section 3.3.2, to date, no full-scale *S. alveolata* restoration scheme has taken place in the UK, and native oyster restoration success can be variable. No guidance is available on *S. alveolata* restoration, but literature on native oyster is extensive. Of particular note is the 2005 Cefas guide on 'bivalve cultivation: criteria for selecting a site', which includes guidance on native oyster (Laing and Spencer, 2005), as well as the recent 'Berlin Oyster Recommendation on the future of Native Oyster Restoration in Europe' (Pogoda et al*.*, 2017). The latter's recommendations included that restoration should focus on two types of sites where;

- *O. edulis* was recorded before, but has disappeared ("reintroduction" sites); and
- *O. edulis* is still present, but in very low density ("reinforcement" sites).

Whilst it is beyond the scope of this project to provide detailed advice on how to undertake a site selection exercise and initial feasibility review for *S. alveolata* and Native Oyster reef restoration, based on the parameters outlined in Section 3.3.1, and the above-noted aspects, the following broad steps could be followed when undertaking a regional search for a potential site (with the order, and need for some of the steps, to a large extent depending on the motivation for a given search):

- Identify areas where there are existing and historic sites in the region of interest (using the 'historic and current sites' datalayers created for this project, as well as other relevant databases, including the National Biodiversity Network's);
- Consult local knowledge to help distinguish historic from existing sites, and create a long list of most desirable sites (e.g. low-density sites, where there are still some reefs, or areas with suitable clutch (for native oysters));
- Undertake a site characterisation process for short listed sites, investigating aspects such as: turbidity, water quality, water depth, predominant sediment, proximity to existing reefs, wave exposure, tidal current speeds; existing nature conservation designations; levels of recreational boating / mooring, fishing (particularly trawling, bait digging, etc. (consult local knowledge if possible);
- Rank the sites according to suitability and project specific drivers; and
- Consult the landowner (likely to be The Crown Estate, with local exceptions).

Once a site, or a set of sites have been identified as the preferred site(s), further studies would generally be required; these would likely involve the commissioning of specialist consultancies or academic institutions, and may include the development and assessment of preliminary designs, as well as numerical site suitability modelling. For native oyster restoration, the availability of a sufficient seed oyster supply would also need to be investigated, as should the introduction of suitable management measures to regulate fishing activity on any newly established beds.

5. 'Best' habitats for restoration / creation

As noted in Section 1, one task which formed part of this project, but did not involve the creation of datalayers, relates to the creation of a 'list of best types of habitat for use in creation', whilst detailing the trade-offs, limitations and considerations. In order to fulfil this task, the following methodology was applied by ABPmer (noting that this task was undertaken without the aid of the RSPB and AER, who were both only sub-contracted for the datalayer tasks).

- Step 1 – A mini-review of previous and ongoing initiatives / literature, and
- Step 2 – A workshop to discuss the opportunities and issues surrounding habitat equivalence with expert stakeholders. This workshop was held in the morning of 25 October 2018, and a summary of the discussions is also provided in this section

For the mini-review, habitat creation / restoration was set in the context of habitat offsetting and biodiversity and environment net gain, as these are topic areas where habitat accounting and issues of habitat equivalency would be encountered, and related literature may thus aid in identifying a methodology for creating a list of 'best habitats'. Section 5.1 outlines the policy context. Associated principles and guidance are summarised in Section 5.2. Section 5.3 relates to the marine environmental context, and Section 5.4 presents the resulting discussion on 'best' marine and coastal habitats for restoration and / or creation. Section 5.5 relates to the workshop.

5.1. Policy context

25 Year Environment Plan

Environmental Net Gain was proposed in the Government's 25 Year Environment Plan (HM Government, 2018) as a development to the increasingly established Biodiversity Net Gain. The Plan committed to embed Environmental Net Gain for development '*to deliver environmental improvements locally and nationally*'. It further specified that '*in future, we want to expand the net gain approaches used for biodiversity to include wider natural capital benefits, such as flood protection, recreation and improved water and air quality. Those approaches will sit alongside existing regulations that protect our most threatened or valuable habitats and species*'.

Marine commitments were also included in the 25 Year Environment Plan, though these did not mention Environmental Net Gain, but stated that the government would work towards:

- *"Reversing the loss of marine biodiversity and, where practicable, restoring it.*
- *Increasing the proportion of protected and well-managed seas, and better managing existing protected sites.*
- *Making sure populations of key species are sustainable with appropriate age structures.*
- *Ensuring seafloor habitats are productive and sufficiently extensive to support healthy, sustainable ecosystems."*

In December 2018, Defra launched a consultation on embedding an Environmental Net Gain approach in the planning system in England, including on whether or not (and how) Biodiversity Net Gain should be made a mandatory requirement[15]. While the consultation focused specifically on the terrestrial planning system, in relation to marine it was noted that:

'*While marine planning and licensing policy and nationally significant infrastructure projects are not in scope of this consultation, we are considering how to best support and mainstream the net gain approaches that many infrastructure and marine projects are already taking. For marine planning and licensing, we will evaluate the actions that projects are already taking to address their environmental impacts and consider how best to implement net gain in the marine context*'.

Biodiversity 2020

Biodiversity 2020, a government strategy for England's wildlife and ecosystem services published in 2011, includes a commitment ('outcome') to achieve *'more, bigger and less fragmented areas for wildlife, with no net loss of priority habitat and an increase in the overall extent of priority habitats by at least 200,000 ha'* (Defra, 2011). Priority habitats are more appropriately referred to as 'habitats of principal importance under Section 41 of the 2006 Natural Environment and Rural Communities (NERC) Act' (see Table 3 for a list of coastal and marine priority habitats).

Table 3: Marine and coastal habitats listed as being of principal importance in England

Category	Habitat Name
Wetland	Coastal and floodplain grazing marsh
Coastal	Coastal saltmarsh
	Coastal sand dunes
	Coastal vegetated shingle
	Intertidal mudflats
	Maritime cliff and slopes
	Saline lagoons
Marine	Blue mussel beds
	Estuarine rocky habitats
	Fragile sponge and anthozoan communities on subtidal rocky habitats
	Horse mussel beds
	Intertidal boulder communities
	Intertidal chalk
	Maërl beds
	Mud habitats in deep water
	Peat and clay exposures
	Sabellaria alveolata reefs
	Sabellaria spinulosa reefs
	Seagrass beds
	Sheltered muddy gravels

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[15] https://consult.defra.gov.uk/land-use/net-gain/ [last accessed in December 2018; consultation closed in early February 2019].

Category	Habitat Name
	Subtidal chalk
	Subtidal sands and gravels
	Tide-swept channels

Natural Environment White Paper 2011

The development of Biodiversity Net Gain in the UK essentially started with Defra's biodiversity offsetting pilots in 2012 (see Section 5.2 for more detail). These were in turn prompted by policy developments such as the Natural Environment White Paper 2011 (HM Government, 2011), which contained the following commitment: '*We want to improve the quality of our natural environment across England, moving to a net gain in the value of nature. We aim to arrest the decline in habitats and species and the degradation of landscapes. We will protect priority habitats and safeguard vulnerable non-renewable resources for future generations. We will support natural systems to function more effectively in town, in the country and at sea. We will achieve this through joined-up action at local and national levels to create an ecological network which is resilient to changing pressures.'*

UK Marine Policy Statement (2011) and Marine Plans

With regard to the marine context, the UK Marine Policy Statement is also worth noting; this stated as one of its high level marine objectives that biodiversity should be '*protected, conserved and where appropriate recovered and loss […] halted*'. It furthermore elaborates that:

"Marine plan authorities should be mindful that, consistent with the high level marine objectives, the UK aims to ensure:

- *A halting and, if possible, a reversal of biodiversity loss with species and habitats operating as a part of healthy, functioning ecosystems; and*
- *The general acceptance of biodiversity's essential role in enhancing the quality of life, with its conservation becoming a natural consideration in all relevant public, private and non-governmental decisions and policies."*

Where finalised, Marine Plans have started to take account of the commitments included in the Marine Policy Statement; for example, the South Marine Plans contain the following 3 policies relating to enhancing biodiversity, and ensuring no net loss:

- *"S-BIO-2 Proposals that incorporate features that enhance or facilitate natural habitat and species adaptation, migration and connectivity will be supported.*
- *S-BIO-3 Proposals that enhance coastal habitats where important in their own right and/or for ecosystem functioning and provision of goods and services will be supported (…).*
- *S-BIO-4 Proposals that enhance the distribution and net extent of priority habitats should be supported. Proposals must demonstrate that they will avoid reducing the distribution and net extent of priority habitats."*

Environmental legislation

In addition, there is a wide range of environmental legislation seeking to avoid, reduce and minimise environmental impacts; key pieces are listed in Table 4:

Table 4: Key pieces of UK legislation related to mitigation of environmental impacts

UK Legislation
Marine Works (EIA) Regulations 2007 (as amended) (re. marine licensing)
Water Environment (WFD) (England and Wales) Regulations 2017
Marine Strategy Regulations 2010
Conservation of Habitats and Species Regulations 2017 (the "Habitats Regulations")
The Wildlife and Countryside Act 1981 (as amended)

There is specific legislation in place for offsetting negative effects of plans or projects in Natura 2000 sites (Article 68 of the Habitats Regulations) and Marine Conservation Zones (MCZs) (S126 (subsection 7c) Marine and Coastal Access Act). With regard to the former, in the UK, this is interpreted as having to correspond precisely to the negative effects on the species or habitat concerned, and has in practice led to the adoption of a like-for-like principle, often with the application of project-specific multipliers to account for factors such as uncertainty and distance. Many managed realignment schemes have to date been motivated by the need to compensate for impacts to Natura 2000 sites, be they indirect anticipated damages related to coastal squeeze, or direct effects related to port developments. Examples include Stanford Marshes in the Thames Estuary and Medmerry near Selsey in West Sussex (ABPmer, 2017b).

With regard to MCZs, 'measures of equivalent environmental benefit to the damage' need to be undertaken if it is considered that the benefit to the public of proceeding with the act clearly outweighs the risk of damage to the environment that will be created by proceeding with it. MMO (2013) considered that 'types of compensatory measures that might be considered under the Habitats Regulations would also be appropriate to put forward here, although consideration will not be confined to those'.

Terrestrial planning

With regard to terrestrial planning, it is worth noting that the revised 2018 National Planning Policy Framework (NPPF) also contains a commitment related to Biodiversity Net Gain, stating that there should be a focus on '*minimising impacts on and providing net gains for biodiversity*'. Furthermore, in terrestrial planning, there exists a mechanism for securing planning gain under Section 106 of the Town and Country Planning Act 1990 (as amended), in the form of s106 agreements. These are often referred to as 'developer contributions'.

5.2. Principles and guidance

Business and Biodiversity Offsets Programme (BBOP) No Net Loss Principles

Numerous documents outlining principles of biodiversity offsetting, Biodiversity Net Gain and more recently Environmental Net Gain have been published. Most of these build on principles originally developed by the Business and Biodiversity Offsets Programme (BBOP) in 2012 on 'No Net Loss and Beyond'. These were as follows:

1. Adherence to the mitigation hierarchy;
2. Limits to what can be offset (irreplaceability/vulnerability);
3. Landscape context (full range of values of biodiversity, supporting an ecosystem approach);
4. No net loss (and preferably a net gain of biodiversity);
5. Additional conservation outcomes;
6. Stakeholder participation throughout;
7. Equity (in design and implementation);
8. Long-term outcomes (adaptive management, monitoring and evaluation, ideally in perpetuity);
9. Transparency (in design, implementation and communication); and
10. Informed by sound Science and traditional knowledge.

CIRIA/CIEEM/IEMA (2016) good practice principles for biodiversity net gain

The more recent CIRIA/CIEEM/IEMA (2016) ‘good practice principles’ related to Biodiversity Net Gain list the following principles:

- Apply mitigation hierarchy;
- Avoid losing biodiversity that cannot be offset by gains elsewhere;
- Be inclusive and equitable;
- Address risks;
- Make a measurable Net Gain contribution;
- Achieve the best outcomes for biodiversity;
- Be additional;
- Create a Net Gain legacy;
- Optimise sustainability;
- Be transparent.

Biodiversity versus environmental net gain

With regard to Biodiversity versus Environmental Net Gain, Biodiversity Net Gain has essentially developed into Environmental Net Gain; in the literature, there is a general agreement that Environmental Net Gain needs to be founded on Biodiversity Net Gain (e.g. Biodiversity Net Gain being ‘sacrosanct’ within Environmental Net Gain, whereby ‘*it must not be possible to ‘trade’ wider environmental or social components of an overall net gain for biodiversity gains. This is because biodiversity (nature) is the foundation of the environment that the Government is pledging to improve for future generations’* (CIWEM, 2018)).

Defra offsetting guidance (2012)

As noted above, the 2012 Defra biodiversity offsetting pilots in 2012 initiated the development of practical approaches to and application of Biodiversity Net Gain in England (Defra, 2012a). The methodology for these involved the development of matrices which took account of habitat type, habitat distinctiveness, habitat condition and delivery risks (the latter with the use of multipliers). These matrices are shown in Table 5 to Table 7 below. The matrices focused on priority habitats, i.e. ‘habitats of principal importance under Section 41 of the 2006 NERC Act’ (see Table 3).

Table 5: The Defra habitat distinctiveness / condition matrix (from Defra 2012a).

		Habitat distinctiveness		
		Low (2)	Medium (4)	High (6)
Condition	Good (3)	6	12	18
	Moderate (2)	4	8	12
	Poor (1)	2	4	6

Table 6: Proposed restoration multipliers (from Defra 2012a).

Difficulty of recreation/restoration	Multiplier
Very High	10
High	3
Medium	1.5
Low	1

Table 7: Defra guidance on distinctiveness (from Defra 2012a).

Habitat type band	Distinctiveness	Broad habitat type covered	Type of offset
High	High	Priority habitat, as defined in Section 41 of the NERC Act	Same band type, and ideally like for like
Medium	Medium	Semi natural	Within band type or trade up
Low	Low	E.g. Intensive agricultural- but may still form part of the ecological network in the area	Trade up

None of the Defra pilots which subsequently applied these matrices in practice included coastal or marine priority habitats. The matrices are currently undergoing an update, with consultation having taken place earlier in 2018, and publication anticipated in 2019 (CIWEM, 2018). Furthermore, in collaboration with the University of Oxford, Natural England is developing a new ‘Eco-Metric’ approach to growing natural capital, which aims to capture the non-monetary value of environmental goods and services from Biodiversity Net Gain (CIWEM, 2018).

The Crown Estate Marine Biodiversity Metric (2013)

With regard to coastal and marine environments, a project on behalf of The Crown Estate (TCE) developed a Marine Biodiversity Metric in the context of a scoping study (Dickie et al., 2013). This applied differing criteria for habitats and species, as follows:

- Habitats: Biodiversity lost = Abundance lost (Area lost x average abundance per unit area lost) x Mean macrofaunal species richness (No of species per unit area); and
- Species: Measures to offset impact to populations of species (see Figure 15 for a theoretical calculation applied for a hypothetical offshore windfarm).

Figure 15: Example species offset calculation (and image of hypothetical windfarm) (from Dickie., 2013).

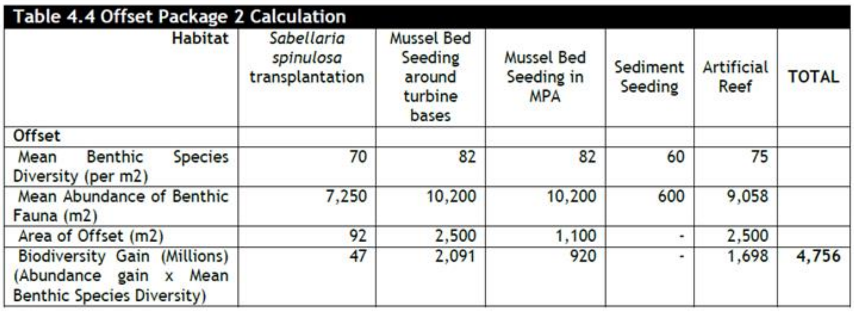
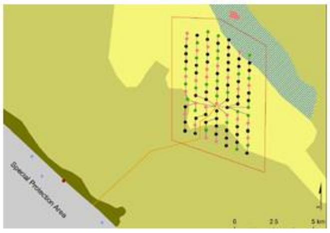


Table 4.4 Offset Package 2 Calculation

Habitat	*Sabellaria spinulosa* transplantation	Mussel Bed Seeding around turbine bases	Mussel Bed Seeding in MPA	Sediment Seeding	Artificial Reef	TOTAL
Offset						
Mean Benthic Species Diversity (per m2)	70	82	82	60	75	
Mean Abundance of Benthic Fauna (m2)	7,250	10,200	10,200	600	9,058	
Area of Offset (m2)	92	2,500	1,100	-	2,500	
Biodiversity Gain (Millions) (Abundance gain x Mean Benthic Species Diversity)	47	2,091	920	-	1,698	4,756



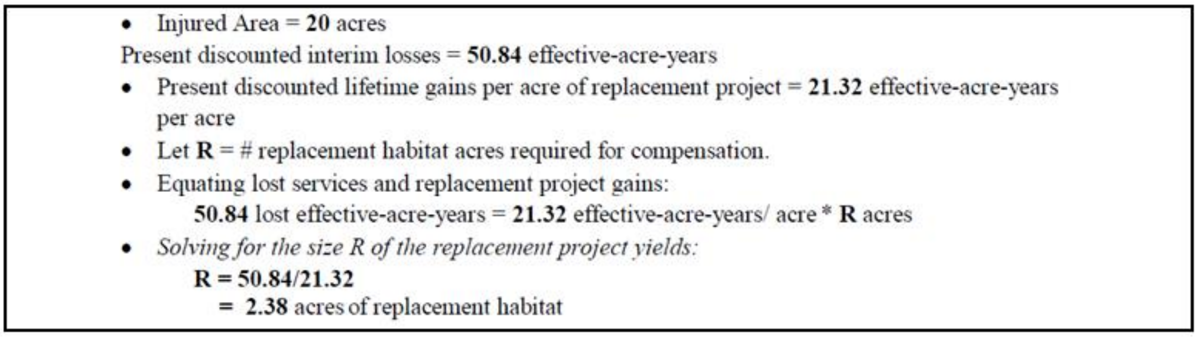
Habitat Equivalency Approaches

Offsetting examples, guidance and principles discussed above were pre-dated by ‘habitat equivalency’ approaches which emerged in the United States in the latter part of the last century, and are still in use there. For example, the National Oceanic and Atmospheric Administration’s (NOAA) Habitat Equivalence Analysis (NOAA, 1995/2000) was developed as a methodology used to determine compensation for ‘resource injuries’ from pathways such as discharges of oil, releases of hazardous substances, or physical injury due to such as vessel groundings. Habitat Equivalence Analysis would be applied by NOAA as a framework for scaling compensatory restoration, by calculating:

- The duration and extent of injury (from the time of injury until the resource recovers to baseline, or possibly to a maximum level below baseline);
- The services provided by the compensatory project, over the full life of the habitat;
- The size of the replacement project for which the total increase in services provided by the replacement project equals the total interim loss of services due to the injury; and
- The costs of the replacement project, or specify the performance standards in cases where the responsible party will be implementing the compensatory habitat project.

An example calculation for determining the size of a project to compensate for interim losses is shown in Figure 16 below.

Figure 16: Example habitat equivalency calculation (from NOAA, 1995/2000).



- Injured Area = **20** acres

Present discounted interim losses = **50.84** effective-acre-years

- Present discounted lifetime gains per acre of replacement project = **21.32** effective-acre-years per acre
- Let **R** = # replacement habitat acres required for compensation.
- Equating lost services and replacement project gains:
  50.84 lost effective-acre-years = **21.32** effective-acre-years/ acre * **R** acres
- *Solving for the size R of the replacement project yields:*
  R = 50.84/21.32
  = **2.38** acres of replacement habitat

5.3. The marine environment

When compared to terrestrial environments, it is worth highlighting that the marine environment is a more open system where habitats (and associated species) are largely controlled by physical factors (sediment type, salinity, water depth, temperature, exposure) and human pressures. Many specific challenges encountered here are not necessarily found in terrestrial environments. This includes, most notably, the dynamic and interconnected nature of the environment, the presence of numerous highly mobile and migratory species and higher uncertainty regarding the evidence base and the success of restoration and creation measures.

Given that, in England, most intertidal habitats, and a large percentage of coastal and marine habitats, are already designated under the Birds and Habitats Regulations, most of the existing marine habitat restoration and creation projects have to date been focussed on compensating for impacts on such international sites. There is less experience in England with offsetting for other purposes, noting that offsetting with regard to the Habitats Regulations tends to be interpreted as ‘like for like’ compensation, whereas offsetting related to Biodiversity Net Gain or Environmental Net Gain would tend to be interpreted in a much wider fashion, where theoretically losses could be offset using the creation of other habitats, averting risk or removing an anthropogenic pressure (e.g. Dickie et al., 2013).

In the rare cases where a development affects coastal or marine habitats which are not designated, then broader offsetting could be practiced. Equally, where a development takes place in a designated estuary, and a package of compensatory measures has been agreed with regard to the Habitats Regulations, there could be scope for additional measures to be undertaken to achieve Biodiversity Net Gain or even Environmental Net Gain. However, given that most compensatory measures undertaken for Habitats Regulations purposes already tend to include multipliers to account for uncertainties and distance (e.g. Morris et al., 2016), developers may not be motivated to pursue Biodiversity Net Gain without it being made mandatory for all development (which it currently is not, though this is being considered for terrestrial planning at present, as noted above).

Due to the prevalence of compensatory habitat creation in the marine context, there is consequently very limited discussion in the British and indeed European literature with regard to offsetting in the marine environment which is not related to 'like for like' creation, and little discourse on the types of habitats which could / should preferentially be created.

A notable exception is a review undertaken by Reach et al. (2015) related to the Swansea Tidal Lagoon Project on behalf of Natural Resources Wales (NRW). This report involves a review of measures to create, restore or enhance selected marine habitats. The habitats covered are: saltmarsh and intertidal mudflats, coastal saline lagoons, intertidal sheltered muddy gravels, Seagrass beds – *Zostera* species, *Sabellaria alveolata* reefs, Native Oyster *Ostrea edulis* beds, artificial reef habitat, non- migratory fish habitat, and Atlantic Herring spawning bed habitat. As part of the review, a method was developed for assessing levels of confidence that can be attributed to particular habitat creation measures. Table 8 presents the summary table shown in Reach et al. (2015).

Table 8: The overall confidence levels for each of the habitats, species and measures assessed in the Reach et al. (2015) review.

Habitat/species/Measure	Confidence score	Confidence level
Saltmarsh	5.0	High
Intertidal mudflats	4.2	Medium
Coastal saline lagoons	5.0	High
Intertidal sheltered muddy gravels	1.4	Very Low
Seagrass beds	3.6	Medium
Sabellaria alveolata reefs	2.5	Low
Ostrea edulis beds	2.4	Low
Artificial substrata habitat	3.6	Medium
Non-migratory fish habitat	2.8	Low
Atlantic Herring spawning habitat	2.0	Very Low
Marine invasive non-native species and biosecurity	3.2	Low

The marine biodiversity offsetting reports commissioned by The Crown Estate in 2013 (Dickie et al., 2013; Cook and Clay, 2013) also provide some insights. The latter considered five offsetting and habitat banking options, based on Dickie et al. (2013):

- Restoration – the manipulation of the physical, chemical, or biological characteristics of a degraded site, with the goal of enhancing natural functions or

species communities in an existing habitat. Habitat restoration options considered are biogenic reef, seagrass bed and kelp forest restoration and the restoration of historical aggregate extraction sites.
- Creation – the manipulation of the physical, chemical, or biological characteristics of a site to develop a habitat that did not previously exist. Habitat creation options considered are artificial island and artificial reef creation and sediment seeding.
- Averted risk – the protection of biodiversity that is at risk of loss or degradation. The eradication of invasive species and reduction in fishing pressures have been assessed as risk averting options.
- Preservation – an action to remove a threat to, or prevent the decline of, the condition of a habitat or species. Preservation options considered are seabird population protection measures and the provision of conservation officers and patrols.
- Research – Financial contributions to a research fund could be considered as an indirect offset for residual impacts where there is insufficient evidence to quantify the scale of impact and thus not possible to confidently apply metrics, or where metrics do not exist.

With regard to habitat restoration, the restoration of the following habitats / areas was discussed, and their feasibility reviewed: biogenic reef, seagrass beds, historic aggregate extraction areas and kelp forests. With regard to habitat creation, three potential habitat creation methods were discussed and assessed; artificial island creation, artificial reef creation and sediment seeding.

It should be noted however that, despite there being no offsetting examples *per se*, there have been several habitat creation initiatives by non-governmental organisations which have sought to re-create a variety of coastal and marine habitats and their functions. The RSPB's Wallasea Island Wild Coast Project constitutes a notable project, whereby various techniques have been employed to create a wide variety of intertidal and coastal habitats on Wallasea Island in Essex, incorporating numerous different niches and variations of habitats, and also taking account of climate change adaptation (RSPB, 2018c).

5.4. 'Best' habitats for restoration / creation

This section reviews what habitats could considered for restoration / creation in a marine context.

Rather than only presenting a list of a few habitats, it was decided to present a high-level matrix (see Table 9), related to restoration feasibility of all marine and coastal habitats of principal importance (under the 2006 NERC Act), in line with recommendations from the workshop participants, and also guidance related to the Defra Offsetting Matrices (Defra, 2012a, b). As shown in Table 9, all these habitats are considered to have a high 'distinctiveness', and would all have suffered from various levels of decline, loss and degradation in the past, for numerous reasons (see Section 3 for a discussion of mudflats, saltmarshes, seagrasses and biogenic reefs habitats this project focused on). It is considered outside the scope of this project to judge restoration/creation priorities, and rank habitats accordingly.

With regard to restorability, the following categories are evaluated in Table 9, each using a 3-point scale of high, medium and low, based on expert judgement (amongst others drawing on sources quoted in Section 3 above, and also other previous reviews such as Reach et al., 2015; Cook and Clay, 2013; Dickie et al., 2013):

- Feasibility – of restoration (theoretical rather than practical);
- Evidence – of successful restoration;
- Confidence – in the assessment provided in the matrix; and
- Distinctiveness – of the habitat, as identified for Defra (2012b).

Table 9: Coastal and marine habitats of principle importance – restorability.

NERC Habitat Name	Restorability	Evidence	Confidence	Distinctive-ness*
Coastal				
Coastal saltmarsh	High	High	High	High
Coastal sand dunes	High	Medium	Medium	High
Coastal vegetated shingle	High	Medium	Medium	High
Intertidal mudflats	High	Medium	Medium	High
Maritime cliff and slopes	Low	Low	Low	High
Saline lagoons	Medium-high	Medium	Medium	High
Marine				
Blue mussel beds	Medium	Low	Low	High
Estuarine rocky habitats	Medium	Low-medium	Low-medium	High
Fragile sponge and anthozoan communities on subtidal rocky habitats	Low	Low-medium	Low	High
Horse mussel beds	Medium	Low	Low	High
Intertidal boulder communities	Medium	Low-medium	Low-medium	High
Intertidal chalk	Medium	Low	Low	High
Maërl beds	Low	Low	Low	High
Mud habitats in deep water	Low	Low	Low	High
Peat and clay exposures	Low-medium	Low	Low	High
Sabellaria alveolata reefs	Medium	Low	Low-medium	High
Sabellaria spinulosa reefs	Low	Low	Low	High
Seagrass beds	Medium-high	Low-medium	Medium	High
Sheltered muddy gravels	Medium	Low	Low	High
Subtidal chalk	Medium	Low	Low	High
Subtidal sands and gravels	Medium	Low	Low	High
Tide-swept channels	Medium	Low	Low	High
* as defined by Defra (2012b)				

Table 9 reflects the fact that, with the exception of a few habitats, notably saltmarsh, the development of an effective range of measures to ameliorate the negative environmental impacts of development, or achieve Biodiversity Net Gain, in the marine environment, is still in its infancy. Furthermore, some habitats are very difficult to recreate or restore, notably features such as cliffs, and fragile, slow growing, habitats with very particular requirements such as Maërl beds. For some habitats, although there is a strong body of evidence to suggest that restoration measures should be possible (e.g. seagrass beds), restoration success in England has to date been limited. Some restoration / creation methods also rely upon the sourcing/harvesting of seed or brood stock (e.g. establishing *Zostera* spp. or *O. edulis* beds), and in many cases suitable sources may be scarce or themselves located within existing marine protected areas.

Even for those habitats where good evidence exists with regard to creation through physical interventions (notably intertidal habitats through managed realignment or RTE), as noted in Section 3.2., outcomes of such habitat creation schemes can sometimes be difficult to predict (e.g. with regard to use by a given bird species), and it can take up to several decades for habitat equivalency with adjacent habitats to be reached (though it can equally happened fairly quickly). Mudflat can quickly transition to saltmarsh in estuaries with high sediment loads. It is also noteworthy that saline lagoons created through RTE require fairly intensive management, and the 'in perpetuity' principle can be challenging in relation to habitats reliant on concrete and culvert structures. Indeed, 'in perpetuity' creation is challenging in the marine environment as a whole, given its dynamic nature, and also anticipated challenges in relation to climate change and accelerated sea level rise.

This analysis highlights the considerable uncertainties about the likely efficacy of possible creation / restoration measures. This was also noted by the workshop participants (see Section 5.3), who stressed the need for trials and consistent monitoring in those areas where considerable potential for restoration exists, but actual successful schemes are scarce, particularly in the UK. Difficulties around financing and overseeing such measures were acknowledged. It was also discussed that, rather than creating habitats, restoration of habitats through the removal of anthropogenic pressures such as pollution, mooring or fishing can be a highly efficient approach.

5.5. Summary of workshop discussions

In order to inform such a discussion, a workshop was convened with marine industry experts. This workshop was held in the morning of 25 October 2018, and attended by representatives from the following NGOs, government agencies and consultancies: ABPmer, AER, eftec, the Environment Agency, the MMO, the RSPB, The Biodiversity Consultancy, Treweek Environmental Consultants and The Wildlife Trusts.

In summary, with regard to identifying the 'best' types of habitats for creation or restoration, it was agreed that this should be related to aspects of restorability, demand, scale and uniqueness/value. Ecosystem services could potentially also be considered, though this was not supported by all participants, and viewed as an additional level of complexity. It was noted that there were some habitats which cannot be restored, or where there is no capacity to restore. Participants highlighted difficulties surrounding restoration, including for some species / habitats where one would expect greater success (e.g. due to the multitude of restoration efforts, e.g. for native oysters). Offsetting by reducing a pressure was also raised as a possibility, and participants discussed that it would be desirable to have regional strategies regarding creation / restoration, i.e. what habitats or species should be focussed on in a given region. Such net gain strategies should be set in a historic context. They could ensure that measures would be of a sufficient scale and /or in a beneficial location to improve chances of success. The possibility of initiating funds to facilitate the implementation of such regional strategies (e.g. pooled offsets) was discussed, and difficulties with such an approach noted by several participants.

6. Conclusions and recommendations

The main purpose of this study was to develop GIS datalayers which could be used / uploaded onto the MMO's MIS, and to help inform marine planning. Six datalayers were consequently created. Furthermore, a list of the 'best' habitats for use in (re)creation / restoration was also developed.

In order to facilitate the informed use of the datalayers created for this project, a brief literature review on the ecology of these habitats has also been presented in this report, focusing on the environmental conditions required for their restoration or creation. Furthermore, the status of the habitats and techniques which have been employed to create or restore them, have also been summarised.

The methodology for the creation of the six datalayers was confirmed and refined at a methodology workshop which was attended by the project's consultant team and members of the Defra group (namely the MMO, Natural England and the Environment Agency). The datalayers have been grouped in relation to the three habitats or habitat groups which formed the focus of the datalayer tasks (see also Table 2):

- Mudflats and saltmarshes:
    - Potential habitat creation sites within the current floodplain (applying the techniques known as 'managed realignment' or 'regulated tidal exchange');
    - Potential beneficial use (mud) – stretches which may benefit;
    - Potential beneficial use (mud) - potential material sources (maintenance dredge disposal sites);
- Biogenic reefs:
    - Potential honeycomb worm (*S. alveolata*) restoration - historic and current sites;
    - Potential European flat oyster *(O. edulis*) restoration - historic and current sites;
- Seagrass beds:
    - Potential seagrass creation / restoration – historic sites.

For each datalayer, a detailed description of the methodology for its creation has been summarised in the report, and broad guidance for its use provided, as well as limitations / caveats discussed. Each of the datalayers has been delivered to the

MMO as an Esri compatible GIS file (ArcGIS 10.2), accompanied by a detailed processing log, as well as MEDIN-compliant metadata.

These datalayers can all be used to varying degrees to aid searches for potential restoration or creation sites. They would generally be most useful during the initial stages of a search for potential sites, and further investigations and consultation of local knowledge would always be required to confirm whether or not a site is actually suitable for the restoration or creation of a given habitat. Where available, searches which may have previously been undertaken in a given region should also be consulted. This is particularly the case with regard to the ‘habitat creation in the current floodplain’ datalayer, which is related to the managed realignment technique, for which many searches have to date been undertaken by various stakeholders, covering most of the country. However, as all of these applied different criteria to undertake their searches, the datalayer produced for this project has been created to provide a consistent aid in future site searches. This datalayer should be viewed as providing a high level indication of where managed realignment or RTE may be feasible. Its limitations should be noted, especially that the larger sites identified are unlikely to be suitable for intertidal habitat creation across their entirety. Also, the inclusion of a site in the layer does not indicate whether or not land is actually available for intertidal habitat creation (e.g. landowners may well not be interested to sell), or how much of an impact undertaking managed realignment there may have on adjacent estuarine or coastal habitats. Further limitations of this, and the other datalayers, were highlighted throughout Section 4.

With regard to creating a list of the ‘best’ habitats for use in (re)creation / restoration, the report presents a high-level matrix which broadly assesses the restoration feasibility of all marine and coastal habitats which are considered to be of principal importance (under the 2006 Natural Environment and Rural Communities (NERC) Act). This is in line with recommendations from participants of a dedicated workshop, and available guidance and reviews.

The development of this matrix was supported by a ‘mini-review’ of topics related to habitat offsetting and biodiversity and environment net gain, as these are subject areas where habitat accounting and issues of habitat equivalency would be encountered, and related literature may thus aid in identifying a methodology for creating a list of ‘best’ habitats.

The review showed that, with the exception of a few habitats, notably saltmarsh, the development of an effective range of measures to ameliorate the negative environmental impacts of development, or achieve biodiversity net gain, in the marine environment, is still in its infancy. For many habitats, considerable uncertainties remain about the likely efficacy of possible marine habitat creation / restoration measures. Further trials, research and consistent monitoring are required to improve the evidence base and improve confidence in restoration / creation feasibly.

7. References

ABPmer (2006). Morecambe Phase VI Coastal Defence Development. Mitigation works – Final Technical Note. Southampton: ABPmer. p87.

ABPmer (2011). Case Study on the Wallasea (North) Managed Realignment Scheme (England). Southampton: ABPmer Ltd, White Paper. p9.

ABPmer (2017a). White Paper: Using Dredge Sediment for Habitat Creation and Restoration: A Cost Benefit Review, A summary of the techniques, costs and benefits associated with using fine dredge sediment to 'recharge' intertidal habitat. Southampton: ABPmer Ltd, White Paper. p43.

ABPmer (2017b). White Paper: UK Marine Habitat Creation Schemes. A summary of completed managed realignment and regulated tidal exchange projects. Southampton: ABPmer Ltd, White Paper. p25.

ABPmer (2018). Beneficial Use of Dredge Sediment in the Solent (BUDS), Phase 1 Project Scoping and Partnership Building, ABPmer Report No. R.2845. A report produced by ABPmer for Solent Forum, March 2018. Southampton: ABPmer Ltd, White Paper. p111.

ABPmer OMReg. Online Marine Registry – a database of completed coastal habitat creation schemes and other adaptation projects: www.omreg.net [last accessed February 2019].

Adam, P. (1990). Saltmarsh Ecology. Cambridge: Cambridge University Press. p461.

Adam, P. (2002). Saltmarshes in a time of change. Environmental Conservation, 29 (1), pp: 39-61.

Airoldi, L.B., Beck, M.W. (2007). Loss status and trends for coastal marine habitats of Europe. Oceanography and Marine Biology: An Annual Review, 45, pp: 345-405.

Andrews, J. (1979). Pelecypoda: Ostreidae. In: Giese, A. (Ed.). Reproduction of Marine Invertebrates, Volume 5: Molluscs: Pelecypods and lesser classes. New York, USA: Academic Press. pp: 293-341.

Ayata, S.D., Ellien, C., Dumas, F., Dubois, S. and Thiébaut, É. (2009). Modelling larval dispersal and settlement of the reef-building polychaete *Sabellaria alveolata*: Role of hydroclimatic processes on the sustainability of biogenic reefs. Continental Shelf Research, 29, pp: 1605-1623.

Baker, J.G. and Tate, G.R. (1867). A new flora of Northumberland and Durham with sketches of its climate and physical geography. London: Nabu Press. p336.

Baud, J.P., Gerard, A. and Graven, N.Y. (1997). Comparative growth and mortality of *Bonamia ostrea* resistant and wild flat oyster, *Ostrea edulis*, in an intensive system. Marine Biology, 130, pp: 71-79.

BBC (2018). Dornoch Firth: Extinct oyster reefs restoration starts. Online content available at: https://www.bbc.co.uk/news/uk-scotland-45829130 [last accessed November 2018].

Bertelli, C.M. and Unsworth, R.K.F. (2018). Light Stress Responses by the Eelgrass, *Zostera marina* (L). Frontiers in Environmental Science, 6 (39).

Bertelli, C.M., Robinson, M.T., Mendzil, A.F., Pratt, L.R. and Unsworth, R.K.F. (2017). Finding some seagrass optimism in Wales, the case of *Zostera noltii*. Marine Pollution Bulletin, 134, pp: 216-222.

Biofuelwatch (2011). Biomass Power Station at Blyth. Biofuelwatch briefing. Available online at http://www.biofuelwatch.org.uk/docs/Blyth-biomass-briefing.pdf [last accessed December 2018]. p6.

Black and Veatch (2012). EX 28.3 – Final Compensation Proposals Part 3: Development and Operation of the Intertidal Habitat Site. Revision 0. Redhill: Black and Veatch. p113.

Boorman, L. (2003). Saltmarsh Review: An overview of coastal saltmarshes, their dynamic and sensitivity characteristics for conservation and management. JNCC Report, No. 334. Peterborough: JNCC. p114.

Brodersen, K., Hammer, K.J., Schrameyer, V., Floytrup, A., Rasheed, M.A., Ralph, P.J., Kuehl, M. and Pedersen, O. (2017). Sediment resuspension and deposition on seagrass leaves impedes internal plant aeration and promotes phytotoxic H2S intrusion. Frontiers in Plant Science. 8 (657).

Brodersen, K., Lichtenberg, M., Paz, L.C. and Kuehl, M. (2015). Epiphyte-cover on seagrass (*Zostera marina* L.) leaves impedes plant performance and radial O_2 loss from the below-ground tissue. Frontiers in Marine Science. 2 (58).

Bruschetti, M., Luppi, T., Fanjul, E., Rosenthal, A. and Iribarne, O. (2008). Grazing effect of the invasive reef-forming polychaete *Ficopomatus enigmaticus* (Fauvel) on phytoplankton biomass in a SW Atlantic coastal lagoon. J Exp Mar Biol Ecol, 354, pp: 212-219.

Burkholder, J.M., Tomasko, D.A. and Touchette, B.W. (2007). Seagrasses and eutrophication. Journal of Experimental Marine Biology and Ecology, 350 (1-2), pp: 46-72.

Burton, P.J.K. (1961). The Brent Goose and its food supply in Essex. Wildfowl Trust Annual Report. 12. pp: 104-12.

Butcher, R.W. (1934). Report on the Present Condition of Eel Grass on the Coasts of England, based on a Survey during August to October, 1933. ICES Journal of Marine Science, 9, pp: 45-69.

Butcher, R.W. (1933). On the present distribution of *Zostera marina* in England. Journal du Conseil International pour l'Exploration de la Mer, 9, pp: 49-65.

Butcher, R.W. (1941). The distribution of *Zostera* (eelgrass, wigeon grass) and other seashore plants in relation to the migrations of wildfowl. International Wildfowl Inquirer. 1. pp: 29-49.

Cabaco, S., Santos, R. and Duarte, C.M. (2008). The impact of sediment burial and erosion on seagrasses: A review. Estuarine Coastal and Shelf Science. 79 (3), pp: 354-366.

Census of Marine Life (2000-2010) (COML): http://www.coml.org/ [last accessed February 2019].

Centre for Environment Fisheries and Aquaculture Science (Cefas) (2009). Beneficial use of fine-grained maintenance dredged material. Available online at www.cefas.defra.gov.uk/our-science/assessing-human-impacts/dredged-marine-sediments.aspx?RedirectMessage=true [last accessed February 2014].

Charpentier, A., Grillas, P., Lescuyer, F., Coulet, E. and Auby I. (2005). Spatio-temporal dynamics of a *Zostera noltii* dominated community over a period of fluctuating salinity in a shallow lagoon, Southern France. Estuarine, Coastal and Shelf Science, 64 (2–3), pp: 307-315.

Chartered Institution of Water and Environmental Management (CIWEM), 2018. Environmental Net Gain Measurement, Delivery and Application Findings of the CIWEM conference, October 2018. London: CIWEM. p24.

Clarke, R.T., Gray, A.J., Warman, E. A. and Moy, I.L. (1993). Niche modelling of saltmarsh plant species. Wareham: ETSU. p129.

Cleveland Naturalists' Field Club (1994). The Cleveland Coast Flora and History. Society Report. Cleveland Naturalists' Field Club Bulletin, p98.

Construction Industry Research and Information Association (CIRIA) (2010). Beach Management Manual. London: CIRIA. p989.

Construction Industry Research and Information Association (CIRIA), CIEEM (Chartered Institute of Ecology and Environmental Management) and Institute of Environmental Management and Assessment (IEMA) (2016). Biodiversity Net Gain Good practice principles for development. London: CIRIA. p9.

Cook, D. and Clay, N. (2013). Marine Biodiversity Offsetting and Habitat Banking Feasibility Study. London: Royal Haskoning. Produced for The Crown Estate. p124.

Cope, S.N., Bradbury, A.P. and Gorczynska, M. (2008). Solent Dynamic Coast Project: Main report (A tool for SMP2). Available online at www.newforest.gov.uk/nssmp/6652 [last accessed November 2018].

Dale, A.L., McAllen, R. and Whelan, P. (2007). Management considerations for subtidal *Zostera marina* beds in Ireland. Irish Wildlife Manuals, No. 28. National Parks and Wildlife Service, Department of Environment, Heritage and Local Government. Dublin, Ireland.

Davis, K.J., Binner, A., Bell, A., Day, B., Poate, T., Rees, S., Smith, G., Wilson, K. and Bateman, I. (2018). A generalisable integrated natural capital methodology for targeting investment in coastal defence, Journal of Environmental Economics and Policy. p18.

Davison, D.M. and Hughes, D.J. (1998). *Zostera* Biotopes (volume I). An overview of dynamics and sensitivity characteristics for conservation management of marine SACs, Scottish Association for Marine Science (UK Marine SACs Project). p95.

Defra (2003). Guidance Note on Managed Realignment: Land purchase, Compensation and Payment for Alternative Beneficial Land Use. London: Defra. p19.

Defra (2011). Biodiversity 2020: A strategy for England's wildlife and ecosystem services. London: Defra. p48.

Defra (2012a). Biodiversity Offsetting Pilots Technical Paper: the metric for the biodiversity offsetting pilot in England. London: Defra. p27. Available online at https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/69531/pb13745-bio-technical-paper.pdf [last accessed December 2018].

Defra (2012b). Appendix 1 - Distinctiveness Bands for the Biodiversity Offsetting Pilot. London: Defra. p13. Available at: https://webarchive.nationalarchives.gov.uk/20150510162309/https://www.gov.uk/biodiversity-offsetting [last accessed December 2018].

Defra and Environment Agency (2007). Saltmarsh management manual. R and D Technical Report SC030220. Almondsbury: Environment Agency. p129.

Dennison, W.C., Orth, R.J., Moore, K.A., Stevenson, J.C., Carter, V., Kollar, S., Bergstrom, P.W. and Batiuk, R.A. (1993). Assessing water quality with submersed aquatic vegetation. BioScience, 43 (2), pp: 86-94.

Dickie, I., McAleese, L., Pearce, B. and Treweek, J. (2013). Marine Biodiversity Offsetting – UK Scoping Study. Report to The Crown Estate. London: Economics for the Environment Consultancy. p130.

Dixon, M., Morris, R., Scott, C., Birchenough, A., and Colclough, S. (2008). Managed realignment - Lessons from Wallasea, UK. Proceedings of The Institution of Civil Engineers - Maritime Engineering 161 (2), pp: 61-71.

Dubois, S, Barille, L, and Cognie, B. (2009). Feeding response of the polychaete *Sabellaria alveolata* (Sabellariidae) to changes in seston concentration. J Exp Mar Biol Ecol, 376, pp: 94-101.

Dubois, S, Retiere, C. and Olivier, F. (2002). Biodiversity associated with *Sabellaria alveolata* (Polychaeta: Sabellariidae) reefs: effects of human disturbances. J Mar Biol Ass UK, 82, pp: 817-826.

Eagling, L. (2012). Reproductive Success of the Re-laid Native Oyster *Ostrea edulis* in Chichester harbour. Master's Thesis. Southampton: University of Southampton. p63.

Edwards, E. (1997). Molluscan Fisheries in Britain. National Oceanic and Atmospheric Administration and U.S. Department of Commerce, Washington. p129.

Egerton, S. (2014). Distribution mapping and health assessment of honeycomb worm, *Sabellaria alveolata*, reefs on Heysham Flat, Lancashire. Report to the North Western Inshore Fisheries and Conservation Authority.

Environment Agency (2009). Estuary Edges: Ecological Design Guidance. p55. Report produced for the Environment Agency by Biodiversity by Design Ltd, Salix River and Wetlands Services Ltd, Beckett Rankine Ltd and EcoSchemes Ltd. Available at: http://www.ecrr.org/Portals/27/Publications/Estuary%20Edges%20-%20design%20advice.pdf [last accessed February 2019].

Environment Agency (2011). The extent of saltmarsh in England and Wales: 2006–2009. Bristol: Environment Agency. p48.

Environment Agency (2015). Flood Map for Planning Risk. Available online at http://apps.environment-agency.gov.uk/wiyby/cy/151263.aspx [last accessed December 2018].

Envision Mapping, E. (2015). Poole Harbour SPA Seagrass Assessment. Report for Natural England, 2015. Newcastle: Envision Mapping. p47.

Erftemeijer, P.L.A. and Lewis, R.R.R. (2006) Environmental impacts of dredging on seagrasses: A review. Marine Pollution Bulletin 52(12), pp: 1553-1572.

Essex Wildlife Trust (2016) Native Oyster Restoration. Available online at www.essexwt.org.uk/news/2016/07/19/native-oyster-restoration [last accessed December 2018].

Fabi, G., Spagnolo, A., Bellan-Santini, D., Charbonnel, E., Çiçek, B.A., Goutayer García, J.J., Jensen, A.C., Kallianiotis, A. and Neves dos Santos, M., 2011. Overview on artificial reefs in Europe. Brazilian Journal of Oceanography, 59, pp: 155-166.

Firth, L.B., Browne, K.A., Knights, A.M., Hawkins, S.J., and Nash, R. (2016). Eco-engineered rock pools: a concrete solution to biodiversity loss and urban sprawl in the marine environment. Environmental Research Letters, Volume 11 (Number 9).

Firth, L.B., Mieszkowska, N., Grant, L.M., Bush, L.E., Davies, A.J., Frost, M.T., Moschella, P.S., Burrows, M.T., Cunningham, P.N., Dye, S.R. and Hawkins, S.J. (2015). Historical comparisons reveal multiple drivers of decadal change of an ecosystem engineer at the range edge. Evolution and Ecology, 5 (15), pp: 3210-3222.

French, P.W. (1997). Coastal and Estuary Management. Routledge, London. p272.

French, P.W. (2001). Coastal defences: processes, problems and solutions. Routledge, London. p366.

Future, N.S. (2008). A Biodiversity Action Plan for Teignbridge PART 2 - Action Plans T7 Estuaries Action Plan.

Garbutt, R.A., Reading, C.J., Wolters, M., Gray, A.J. and Rothery, P. (2006). Monitoring the development of intertidal habitats on former agricultural land after the managed realignment of coastal defences at Tollesbury, Essex, UK. Marine Pollution Bulletin, 53 (1-4), pp: 155-164.

Grant, J., Enright, C.T. and Griswold, A. (1990). Resuspension and growth of *Ostrea edulis*: a field experiment. Marine Biology, 104, pp: 51-59.

Gravestock, V., James, F., and Goulden, M. (2014). Solent native oyster (*Ostrea edulis*) Restoration: literature review and feasibility study. Lymington: MacAlister, Elliott and Partners, UK. p77.

Grizel, H. and Heral, H. (1991). Introduction into France of the Japanese oyster (*Crassostrea gigas*). Journal du Conseil, 47, pp: 399-403.

Gruet, Y. (1986). Spatio-temporal changes of Sabellarian reefs built by the sedentary polychaete *Sabellaria alveolata* (Linné). Marine Ecology, 7, pp: 303-319.

Gunther, R. (1897). The oyster culture of the ancient Romans. Journal of the Marine Biological Association of the United Kingdom, 4, pp: 360-365.

Harding, T. (1996). Reintroduction of the Native Oyster (*Ostrea edulis*) into the Firth of Forth, Scotland: a feasibility study. Master's thesis, Napier University, UK. p102.

Hastings, A., Byers, J.E., Crooks, J.A., Cuddington, K., Jones, C.G., Lambrinos, J.G., Talley, T.S. and Wilson, W.G. (2007). Ecosystem engineering in space and time. Ecology Letters, 10, pp: 153–164.

Hedgecock, D., Launey, S., Pudovkin, A.I., Naciri, Y., Lapègue, S., and Bonhomme, F. (2007). Small effective number of parents inferred for a naturally spawned cohort of juvenile European flat oysters *Ostrea edulis*. Marine Biology 150, 1173-1182.

Hirst, J.A. and Attrill, M.J. (2008). Small is beautiful: An inverted view of habitat fragmentation in seagrass beds. Estuarine Coastal And Shelf Science, 78 (4), pp: 811-818.

HM Government (2011). The Natural Choice: securing the value of nature. London: Defra. p84.

HM Government (2018). A Green Future: Our 25 Year Plan to Improve the Environment. London: Defra. p151.

Hocking, S. and Tompsett, P. (2002). The Location & Conservation of Seagrass Beds in Cornwall and the Isles of Scilly. Truro: Environment Records Centre for Cornwall and the Isles of Scilly, Cornwall Wildlife Trust. Project Series No. 5.

Holt, T.E.I, Rees, E.I, Hawkins, S.J., and Seed, R. (1998). Biogenic Reefs (volume IX): an overview of dynamic and sensitivity characteristics for conservation management of marine SACs, Oban, Scottish Association for Marine Science (UK Marine SACs Project).

Jackson, A. and Wilding, C. (2007). *Ostrea edulis*. Native oyster. Marine Life Information Network: Biology and Sensitivity Key Information Subprogramme. Plymouth: Marine Biological Association of the United Kingdom.

Jackson, E.L., C.A. Griffiths, and Durkin, O. (2013). A guide to assessing and managing anthropogenic impact on marine angiosperm habitat - Part 1: Literature review, Natural England Commissioned Reports.

Jackson, E.L., Cousens, S.L., Bridger, D.R., Nancollas S.J. and Sheehan, E.V. (2016). Conservation inaction in action for Essex seagrass meadows? Regional Studies in Marine Science, 8(1), pp: 141-150.

John, S. (2013). Target Setting and Delivery: three projects, 14 years on. Powerpoint presentation given by Sian John of Royal HaskoningDSV at the 2013 ABPmer intertidal habitat creation conference.

Joint Nature Conservation Committee (JNCC) (2008). UK Biodiversity Action Plan Priority Habitat Descriptions - *Sabellaria alveolata* Reefs. Peterborough: JNCC. p3.

Jones, B.L. and Unsworth, R.K.F. (2016). The perilous state of seagrass in the British Isles. Royal Society Open Science, 3 (150596) pp: 1-14.

Jones, B.L., Cullen-Unsworth, L.C., and Unsworth, R.K.F (2018). Tracking Nitrogen Source Using $\delta^{15}N$ Reveals Human and Agricultural Drivers of Seagrass Degradation across the British Isles. Frontiers in Plant Science, 9 (133) pp: 1-10.

Kamphausen, L., Jensen, A. and Hawkins, L. (2011). Unusually high proportion of males in a collapsing population of commercially fished oysters (*Ostrea edulis*) in the Solent, United Kingdom. Journal of Shellfish Research, 30, pp: 217-222.

Kennedy, R.J. and Roberts, D.A. (1999). Survey of the current status of the flat oyster *Ostrea edulis* in Strangford lough, Northern Ireland, with a view to the restoration of its oyster beds. Biology and Environment: Proceedings of the Royal Irish Academy, 99, pp: 79-88.

Kennedy, R.J. and Roberts, D. (2006). Commercial oyster stocks as a potential source of larvae in the regeneration of *Ostrea edulis* in Strangford Lough, Northern Ireland. Journal of the Marine Biological Association of the UK, 86, pp: 153–159.

Key, D. and Davidson, P.E. (1981). A Review of Development of the Solent Oyster Fishery, 1971-80. Laboratory Leaflet, No. 52. Lowestoft: Cefas. p40.

Koch, E.M. (2001). Beyond light: Physical, geological, and geochemical parameters as possible submersed aquatic vegetation habitat requirements. Estuaries, 24 (1), pp: 1-17.

Korringa, P. (1946). A revival of natural oyster beds? Nature, 158, pp: 586-587.

Korringa, P. (1952). Recent advances in oyster biology. Quarterly Review of Biology, 27, pp: 339-365.

Korringa, P. (1957). Water temperature and breeding throughout the geographical range of *Ostrea edulis*. L'Année Biologique, 33, pp: 109-116.

Laing, I. and Spencer, B.E. (2005). Bivalve cultivation: criteria for selecting a site. Lowestoft: Cefas. p34.

Laing, I., Walker, P. and Areal, F. (2005). A feasibility study of native oyster (*Ostrea edulis*) stock regeneration in the United Kingdom. Lowestoft: Cefas. p94.

Laing, I., Walker, P. and Areal, F. (2006). Return of the native - is European oyster (*Ostrea edulis*) stock restoration in the UK feasible? Aquatic Living Resources, 19, pp: 283-287.

Larsonneur, C. (1994). The Bay of Mont-Saint-Michel: A sedimentation model in a temperate macrotidal environment. Senckenbergiana maritima, 24, pp: 3-63.

Lee, K.S., Park, S.R., and Kim, Y.K. (2007). Effects of irradiance, temperature, and nutrients on growth dynamics of seagrasses: A review. Journal of Experimental Marine Biology and Ecology, 350 (1–2), pp: 144-175.

Leggett, D.J., Cooper, N. and Harvey, R. (2004). Coastal and Estuary managed realignment - design issues. London: Construction Industry Research and Information Association. p215.

Leuschner, C., Landwehr, S. and Mehlig, U. (1998). Limitation of carbon assimilation of intertidal *Zostera noltii* and *Z. marina* by desiccation at low tide. Aquatic Botany, 62 (3), pp: 171-176.

Lewis, J.G.E. (1962). The ecology, distribution and taxonomy of the centipedes found on the shore in the Plymouth area. Journal of the Marine Biological Association of the United Kingdom, 42 (3), pp: 655-664.

Lonsdale, J.A. (2012). The Potential Alternative Uses of Dredged Material in the Humber Estuary. MSc Thesis, University of Hull. 280p. Available at: https://hydra.hull.ac.uk/assets/hull:6050a/content [last accessed February 2019].

Loosanoff, V.L. (1962). Gametogenesis and Spawning of the European Oyster, *O. edulis*, in Waters of Maine. The Biological Bulletin, 112, pp: 86-94.

Mackenzie, C.L.J., Burrell, V.G., Rosenfield, A., and Hobart, W.L. (1997). The History, Present Condition, and Future of the Molluscan Fisheries of North America and Europe. National Oceanic and Atmospheric Administration and U.S. Department of Commerce, Washington: NOAA. p240.

Marine Management Organisation (MMO) (2013). MMO duties in relation to Marine conservation zones and marine licensing. Available at https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/410273/Marine_conservation_zones_and_marine_licensing.pdf, p9.

Marine Management Organisation (MMO) (2014). Use of beneficial dredged materials in the South Inshore and South Offshore Marine Plan Areas. A report produced by ABPmer for the MMO. Newcastle: MMO. p57.

Marsden, A.L. and Chesworth, J.C. (2015). Inventory of eelgrass beds in Hampshire and the Isle of Wight 2014, Section One: Report. Curdridge: Hampshire and Isle of Wight Wildlife Trust. p224.

Mettam, C., Conneely, M.E., and White, S.J. (1994). Benthic macrofauna and sediments in the Severn Estuary. Biological Journal of the Linnean Society, 51, 71-81.

Morris, R.K.A., Harley, M., Cottle, R., Banks, B., Doody, J.P., Brown, A.E., Weston, A., Hart, R. and Prince, S. (2016). Review of the Effectiveness of Natura 2000 Sites Compensation Measures in England. London: Natural England, p122.

National Oceanic and Atmospheric Administration (NOAA) (2000). Habitat Equivalency Analysis: An Overview. Silver Spring (United States): NOAA. 23p. Available at: https://darrp.noaa.gov/economics/habitat-equivalency-analysis [last accessed February 2019].

Natural England (2016) Healthy Estuaries 2020: Towards Addressing Coastal Squeeze in Estuaries (IPENS002): http://publications.naturalengland.org.uk/publication/4734703644966912 [last accessed February 2019].

Naylor, L.A. and Viles, H.A. (2000). A temperate reef builder: an evaluation of the growth, morphology and composition of *Sabellaria alveolata* (L.) colonies on carbonate platforms in South Wales. London : Geological Society, Special Publications, 178, pp: 9-19.

Naylor, L.A., Kippen, H, Coombes, M.A., Horton, B., MacArthur, M. and Jackson, N. (2017). Greening the Grey: a framework for integrated green grey infrastructure (IGGI) - Appendix 4: Coastal. Glasgow: University of Glasgow, p56. Available at: http://eprints.gla.ac.uk/150672/ [last accessed February 2019].

Naylor, L.A., Venn, O., Coombes, M.A., Jackson, J. Thompson, R.C. (2011). Including Ecological Enhancements in the Planning, Design and Construction of Hard Coastal Structures: A process guide. Report to the Environment Agency (PID 110461). University of Exeter, p66.

Newkirk, G.F. and Haley, L.E. (1982). Phenotypic analysis of the European oyster, *Ostrea edulis* L.: relationship between length of larval period and post setting growth rate. Journal of Experimental Marine Biology and Ecology, 59, pp: 177-184.

Nottage, A. and Robertson, P. (2005). The saltmarsh creation handbook: a project manager’s guide to the creation of saltmarsh and intertidal mudflat. Sandy: Royal Society for the Protection of Birds. p128.

Ocean Biogeographic Information System (OBIS) (2015). Guidelines on the sharing and use of data in OBIS. Available online at: http://iobis.org/data/policy/ [last accessed December 2018].

Ocean Biogeographic Information System (OBIS) database: http://iobis.org/ [last accessed February 2019].

Olsen, O.T. (1883). The piscatorial atlas of the North Sea, English Channel, and St. George's Channels: illustrating the fishing ports, boats, gear, species of fish (how, where, and when caught), and other information concerning fish and fisheries. O.T. Olsen, Grimsby, 50 maps.

Orth, R.J., Carruthers, T.J.B., Dennison, W.C., Duarte, C.M., Fourqurean, J.W., Heck, K.L, Randall Hughes, A., Kendrick, G.A., Kenworthy, W.J., Olyarnik, S., Short, F.T., Waycott, M. and Williams, S.L. (2006). A global crisis for seagrass ecosystems. BioScience, 56 (12), pp: 987-996.

Orton, J.H. (1927). Observations and experiments on sex-change in the European oyster (*O. edulis*) Part I. The change from female to male. Journal of the Marine Biological Association of the United Kingdom, 14, pp: 967-1046.

OSPAR (2009). Background document for *Ostrea edulis* and *Ostrea edulis* beds. OSPAR Biodiversity: Series 428/2009. OSPAR, UK. 22p. (cited 22/10/12). Available online at www.ospar.org/site/assets/files/1505/p00428_ ostrea_edulis_and_beds.pdf [last accessed December 2018].

Pérez-Lloréns, J.L. and F.X. Niell (1993). Temperature and emergence effects on the net photosynthesis of two *Zostera noltii* Hornem. morphotypes. Hydrobiologia, 254 (1), pp: 53-64.

Pethick, J.A. (1984). An introduction to coastal geomorphology. London: Edward Arnolds. p272.

Phillip, G.L. (1936). An Enalid Plant Association in the Humber Estuary. Journal of Ecology, 24, pp: 205-219.

Permanent International Association of Navigation Congresses / International Navigation Association (PIANC) (2009). PIANC Report no 104-2009 Dredged Material as a Resource: Options and Constraints. Brussels (Belgium): PIANC Secretariat. p147.

Pogoda, B., Brown, J., Hancock, B. and von Nordheim, H. (eds.) (2017). Berlin Oyster Recommendation on the Future of Native Oyster Restoration in Europe. The Native Oyster Restoration Alliance (NORA). 6p. Available online at https://noraeurope.eu/wp-content/uploads/2018/Berlin-Oyster-Recommendation-Part-1.pdf [last accessed December 2018].

Pontee, N. (2011). Reappraising coastal squeeze: a case study from north-west England. Maritime Engineering, 164 (MA3), pp: 127-138.

Pye, K. and French, P.W. (1993). Erosion and Accretion Processes on British Saltmarshes. Volumes 1-5. Cambridge: Cambridge Environmental Research Consultants.

Reach I.S., Crabtree R., Morris R.K.A., Cottle R.A., Blyth-Skyrme R.E., Bray S., Latto P. and Lloyd Jones D. (2015). Advice on Adaptive Management and Marine

Biodiversity Enhancement Measures for Coastal Lagoon Developments. A report for Cyfoeth Naturiol Cymru/Natural Resources Wales. Southampton: MarineSpace, p146.

Royal Society for the Protection of Birds (RSPB) (2013). Pagham Harbour Local Nature Reserve Management Plan. Pagham: RSPB, p101.

Royal Society for the Protection of Birds (RSPB) (2017). A productive 1st year for the Oysterbeds Tern Raft. RSPB blog available at: https://community.rspb.org.uk/placestovisit/langstoneharbour/b/weblog [last accessed December 2018].

Royal Society for the Protection of Birds (RSPB) (2018a). Wave after wave of fledglings. Conservation success for the West Hayling Common Tern colony. Blog post 14 December available at: https://community.rspb.org.uk/placestovisit/langstoneharbour/b/weblog/posts/wave-after-wave-of-fledglings [last accessed January 2019]

Royal Society for the Protection of Birds (RSPB) (2018b) Sustainable Shores Technical Report. Sandy: RSPB, p118. Available at: https://www.rspb.org.uk/our-work/conservation/projects/sustainable-shores/ [last accessed December 2018].

Royal Society for the Protection of Birds (RSPB) (2018c). About Wallasea Island Wild Coast project. Website content available at: https://www.rspb.org.uk/reserves-and-events/reserves-a-z/wallasea-island-wild-coast-project/ [last accessed December 2018].

Royal Society for the Protection of Birds (RSPB) - Precipitating a SEA Change in the Beneficial Use of Dredged Sediment - Technical report (2018): https://www.rspb.org.uk/globalassets/downloads/documents/conservation-projects/seabuds-report.pdf [last accessed December 2018].

Salo, T., Pedersen, M.F. and Boström, C. (2014). Population specific salinity tolerance in eelgrass (*Zostera marina*). Journal of Experimental Marine Biology and Ecology, 461, pp: 425-429.

Sanderson, W.G. (1996). Rarity of marine benthic species in Great Britain: development and application of assessment criteria. Aquatic Conservation: Marine and Freshwater Ecosystems, 6, pp: 245-256.

SEASEARCH (2005). Seasearch Dartmouth Dive survey 2005 Short Report. Available at http://www.seasearch.org.uk/downloads/Dartmouth%202005.pdf p2.

Short, F.T. and Wyllie-Echeverria, S. (1996). Natural and human-induced disturbance of seagrasses. Environmental Conservation, 23 (1), pp: 17-27.

Short, F.T., Davis, R.C., Kopp, B.S., Short, C.A., and Burdick, D.M. (2002). Site-selection model for optimal transplantation of eelgrass *Zostera marina* in the Northeastern US. Marine Ecology-Progress Series, 227, pp: 253–267.

Smyth, D., Roberts, D., and Browne, L. (2009). Impacts of unregulated harvesting on a recovering stock of native oysters (*Ostrea edulis*). Marine Pollution Bulletin 58, pp: 916-922.

Sobolewska, H. and Beaumont, A.R. (2005). Genetic variation at microsatellite loci in northern populations of the European flat oyster (*Ostrea edulis*). Journal of the Marine Biological Association of the United Kingdom, 8, pp: 955–960.

Solent Forum (2018) Solent News – article on ‘Oyster Restoration Project Update’ (p.13). Newsletter. Winchester: Solent Forum. p16.

The Solent Forum - Beneficial Use of Dredging in the Solent Project (BUDS): http://www.solentforum.org/services/Current_Projects/buds/ [last accessed December 2018].

Toft, A.R., Pethick, J.S., Burd, F., Gray, A.J., Doody, J.P., Penning-Rowsell, E. and Maddrell, R.J. (1995). A guide to understanding and management of saltmarshes, NRA, R&D Note 324. p213.

Trenhaile, A.S. (1997). Coastal dynamics and landforms. Oxford: Clarendon Press. p382.

Tubbs, C. (1999). The Ecology, Conservation and History of the Solent. Chichester: Packard Publishing Limited. p179.

UK Marine policy statement (2011). https://www.gov.uk/government/publications/uk-marine-policy-statement. [Last accessed February 2018].

UNEP-WCMC and Short, F.T. (2018). Global distribution of seagrasses (version 6.0). Sixth update to the data layer used in Green and Short (2003). Cambridge (UK): UN Environment World Conservation Monitoring Centre. Available online at http://data.unep-wcmc.org/datasets/7 [last accessed December 2018].

Unsworth, R.K.F., Williams, B., Jones, B.L. and Cullen-Unsworth, L.C. (2017). Rocking the Boat: Damage to Eelgrass by Swinging Boat Moorings. Frontiers in Plant Science, 8 (1309) pp: 1-11.

Utting, S.D. and Spencer, B.E. (1992). Introductions of bivalve molluscs into the United Kingdom for commercial culture-case histories. ICES Marine Science Symposium, 194, pp: 84-91.

van Katwijk, M.M., Bos, A.R., de Jonge, V.N., Hanssen, L.S.A.M., Hermus, D.C.R. and de Jong, D.J. (2009). Guidelines for seagrass restoration: Importance of habitat selection and donor population, spreading of risks, and ecosystem engineering effects. Marine Pollution Bulletin, 58 (2), pp: 179-188.

van Katwijk, M.M., Thorhaug, A., Marba, N., Orth, R.J., Duarte, C.M., Kendrick, G.A., Althuizen, I.H.J., Balestri, E., Bernard, G., Cambridge, M.L., Cunha, A., Durance, C., Giesen, W., Han, Q.Y., Hosokawa, S., Kiswara, W., Komatsu, T., Lardicci, C., Lee, K.S., Meinesz, A., Nakaoka, M., O'Brien, K.R., Paling, E.I., Pickerell, C., Ransijn, A.M.A. and Verduin, J.J. (2016). Global analysis of seagrass restoration: the importance of large-scale planting. Journal of Applied Ecology, 53 (2), pp: 567-578.

Vause, B. (2010). Chichester Harbour Oyster Initiative. Shellfish News, 30, pp: 5-6.

Walne, P.R. (1974). Culture of Bivalve Molluscs: 50 years' experience at Conwy. Oxford: Fishing News Books. p190.

Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, W.W., Heck, K.L., Randall Hughes, A., Kendrick, G.A., Judson Kenworthy, W., Short, F.T. and Williams, S.L. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proceedings of the National Academy of Sciences of the United States of America, 106 (30), pp: 12377-12381.

Wightlink (2018). Wightlink’s vertipools amaze marine biologists. Website content available at: https://www.wightlink.co.uk/pressrelease/wightlinks-vertipools-amaze-marine-biologists/ [last accessed December 2018].

Wilson, D.P. (1976). *Sabellaria alveolata* (L.) at Duckpool, North Cornwall, (1975). Journal of the Marine Biological Association of the United Kingdom, 56, pp: 305-310.

Wilson, J. and Simons, J. (1985). Gametogenesis and breeding of *Ostrea edulis* on the west coast of Ireland. Aquaculture, 46, pp: 307-321.

Woodward, I.D., Calbrade, N.A. and Holt, C.A. (2014). Humber Estuary Bird Decline Investigation. BTO Research Report No. 668. Report of work carried out by The British Trust for Ornithology under contract to Natural England, 2015.

Yonge, C.M. (1966). Oysters. London: Collins. p209.

Yorkshire Post (2016). Letting nature take its course: Why they’re no longer defending Spurn Point. Online content available at: https://www.yorkshirepost.co.uk/news/letting-nature-take-its-course-why-they-re-no-longer-defending-spurn-point-1-8198934 [last accessed February 2019].

8. Abbreviations and acronyms

AER	Aquatic Environmental Research
BBC	British Broadcasting Corporation
BBOP	Business and Biodiversity Offsets Programme
BTO	British Trust for Ornithology
BUDS	Beneficial Use of Dredge Sediment in the Solent
Cefas	Centre for Environment, Fisheries and Aquaculture Science
CIEEM	Chartered Institute of Ecology and Environmental Management
CIRIA	Construction Industry Research and Information Association
CIWEM	Chartered Institution of Water and Environmental Management
CO_2	Carbon dioxide
DAS	Disposal At Sea
Defra	Department of Environment, Food and Rural Affairs
EC	European Commission
EEC	European Economic Community
EIA	Environmental Impact Assessment
ETSU	Energy Technology Support Unit
EU	European Union
GIS	Geographical Information System
ha	hectare(s)
HAT	Highest Astronomical Tides
HM	Her Majesty’s
ICES	International Council for the Exploration of the Sea
ID	Identity
IEMA	Institute of Environmental Management and Assessment
IFCA	Inshore Fisheries and Conservation Authority
JNCC	Joint Nature Conservation Committee.
LiDAR	Light Detection And Ranging
MCZ	Marine Conservation Zone
MEDIN	Marine Environmental Data Information Network
MHWN	Mean High Water Neaps
MHWS	Mean High Water Springs
MIS	Marine Information System
MLWS	Mean Low Water Springs
MMO	Marine Management Organisation

MSFD	Marine Strategy Framework Directive
NERC	Natural Environment and Rural Communities
NGO	Non-Governmental Organisation
NOAA	National Oceanic and Atmospheric Administration
NPPF	National Planning Policy Framework
NRW	Natural Resources Wales
OBIS	Ocean Biogeographic Information System
ODN	Ordnance Datum Newlyn
OMReg	Online Marine Registry
OS	Ordnance Survey
OSPAR	Convention for the Protection of the Marine Environment of the North-East Atlantic
PIANC	Permanent International Association of Navigation Congresses / International Navigation Association
RSPB	Royal Society for the Protection of Birds
RTE	Regulated Tidal Exchange
SAC	Special Areas of Conservation
SEABUDS	SEA Change in the Beneficial Use of Dredged Sediment
SEASEARCH	Volunteer Scuba Divers Mapping of coasts of Britain and Ireland
SI	Surface Irradiance
SMP	Shoreline Management Plan
SPA	Special Protection Area
SSSI	Sites of Special Scientific Interest
TCE	The Crown Estate
UK	United Kingdom
UN	United Nations
UNEP-WCMC	United Nations Environment Programme - World Conservation Monitoring Centre
UNESCO	United Nations Educational, Scientific and Cultural Organization
US	United States
USA	United States of America
WFD	Water Framework Directive

Annex A. LiDAR validation sites for the datalayer 'Potential habitat creation sites within the current floodplain'

In order to show what intertidal habitats might initially be anticipated at a given site, as noted in Section 3.1.1, a LiDAR validation exercise was undertaken for seven representative sites. The sites for which this was undertaken are shown in Figure A.17 below, and the results of the validation are illustrated in Figure A.18 to Figure A.24 (as shown in relation to elevations in Ordnance Datum Newlyn (ODN)).

This validation exercise involved downloading LiDAR data for the seven sites from the Survey Open Data website (https://environment.data.gov.uk/ds/survey/#/survey), and then defining the anticipated habitats in relation to tidal height (with tide levels of the nearest port obtained from the Admiralty TotalTide software).

As noted in Section 3.1, based on available guidance and past experience[16], for preliminary site assessments, the intertidal habitat extents that are likely to occur within proposed manged realignment sites can be predicted and plotted using data on tidal elevations in relation to the proposed site's existing ground levels. This is based on the fact that habitats typically develop at the following elevations:

- mudflat between the levels of Mean Low Water Springs (MLWS) and Mean High Water Neaps (MHWN);
- saltmarsh between MHWN and Mean High Water Springs (MHWS)[17];
- upper saltmarsh between MHWS and Highest Astronomical Tides (HAT);
- transitional grassland between HAT and (approx.) one metre above HAT (HAT+1)[18]; and
- grassland and other terrestrial habitats at elevations over one metre above HAT.

Intertidal habitat is therefore generally understood to extend to the levels of HAT. These definitions of intertidal habitats are in agreement with government practice (specifically Defra, Natural England and the Environment Agency). It is recognised that this 'habitat prediction' process using land elevations involves an inherent simplification of the likely formation of intertidal habitats as the formation of vegetation will be dependent on a number of site specific influences (especially the drainage patterns). However, it does provide a valuable indication of the potential suitability of a site to deliver these different habitat types. In practice, detailed design work will need to be undertaken to create conditions within the site that are best suited to the creation of targeted habitats (e.g. mudflat).

The validation exercise confirmed that, generally, using of the Floodplain 3 datalayer would lead to the identification of sites which could support mostly intertidal habitat if they were breached, though actual habitats would depend on the regional and site conditions. The highest lying site among the validation sites was at 'Ross' in

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[16] E.g. Nottage and Robertson, 2005; Dixon *et al*., 2008.

[17] Noting however that establishment of comprehensive saltmarsh cover in areas of suitable elevation would typically take up to 5 years; thus, during the first one or 2 years post breach, until pioneer species start colonising, most of the site will appear bare (once the terrestrial vegetation has died off).

[18] The HAT+1m 'rule of thumb' would generally not be recommended for application in micro-tidal estuaries, where it may be appropriate to reduce this to 0.5 m or less, depending on local conditions, and ideally referring to insights from local saltmarsh habitat surveys.

Northumberland (see Figure A.24), where only around 20% of the site would develop into saltmarsh if it were to be breached. The lowest lying site was the ‘Walton’ site (Figure A.22), where around 85 % of the site would likely initially be mudflat. This is in line with ABPmer’s knowledge of sites in Essex and the Thames Estuary, which tend to be fairly low lying. Humber and Severn Estuary sites tend to be higher lying, which was confirmed during this exercise, as shown in Figure A.18 and Figure A.23. Solent sites are often higher lying, with the Hayling site shown in Figure A.20 being slightly lower than most nearby sites.

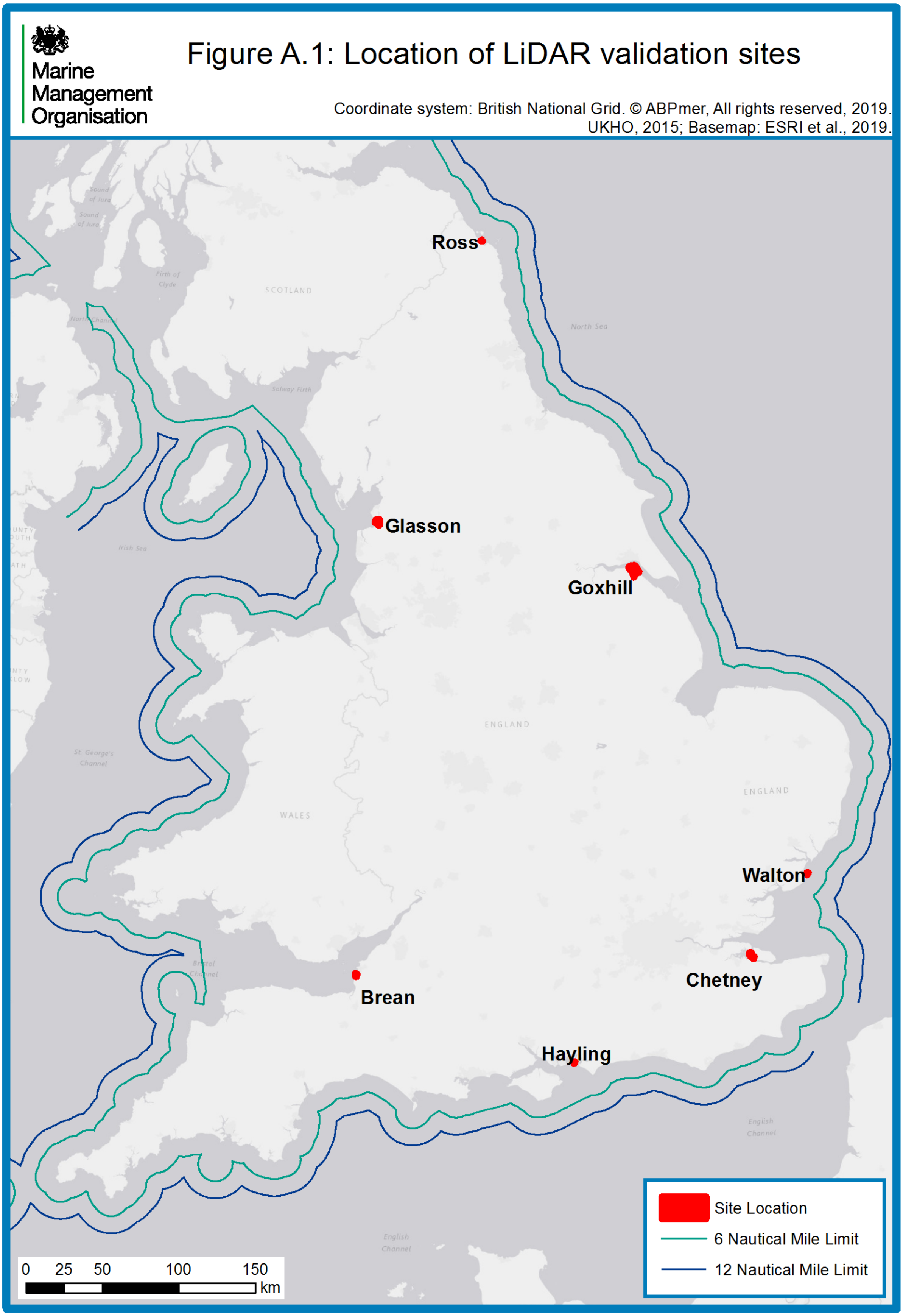

Figure A.17: Location of LiDAR validation sites.

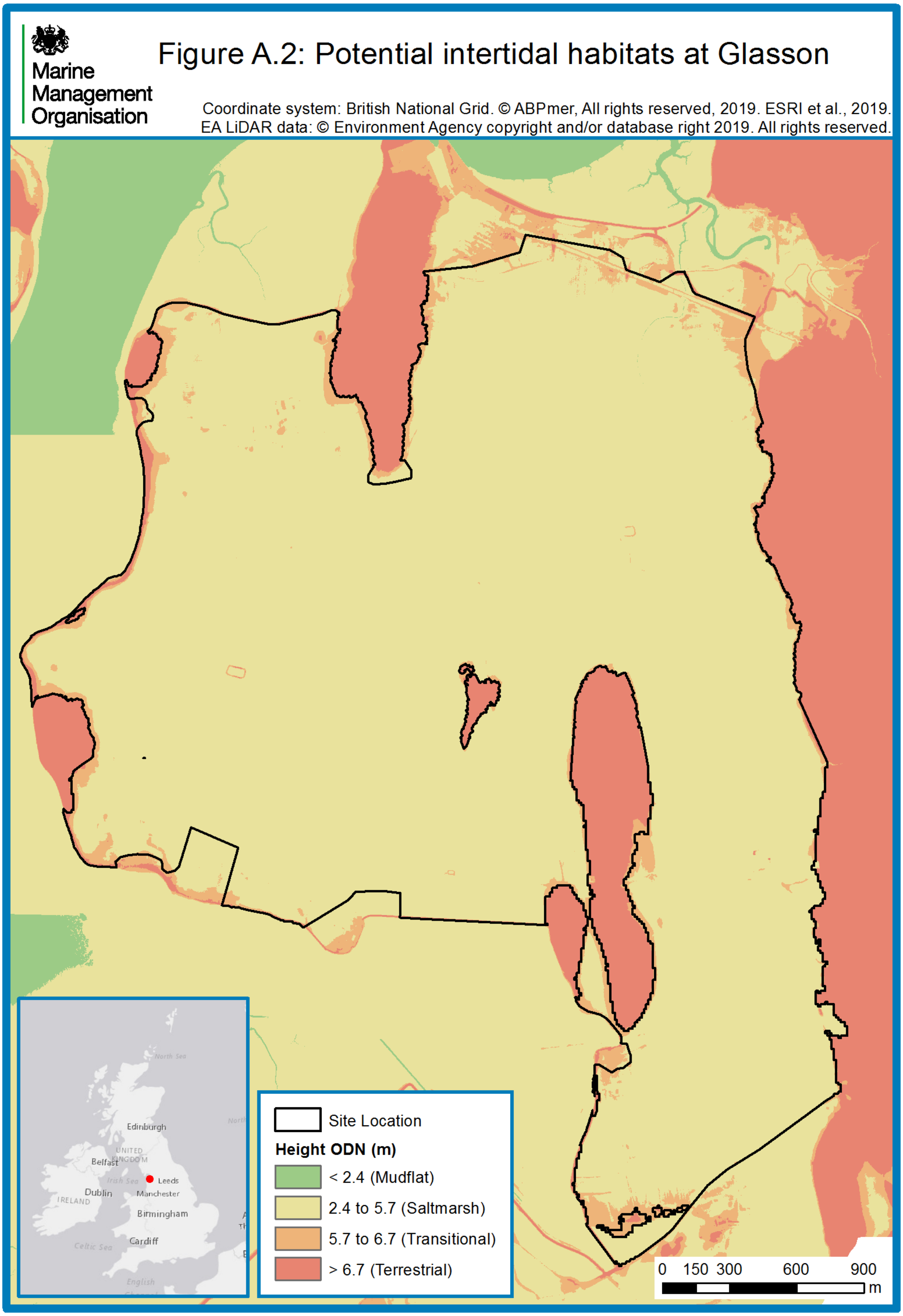

Figure A.18: Potential intertidal habitats at Glasson.

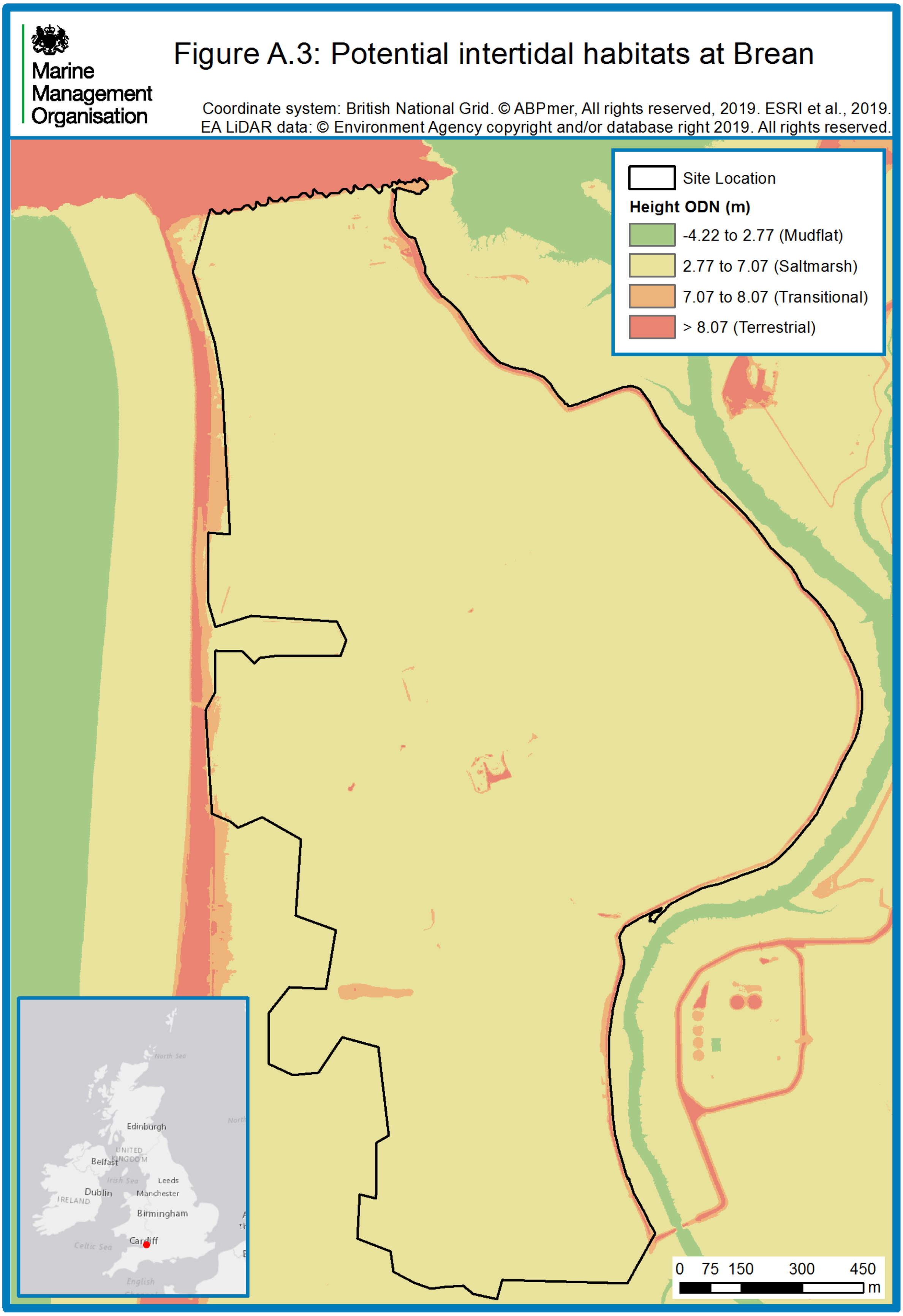

Figure A.19: Potential intertidal habitats at Brean.

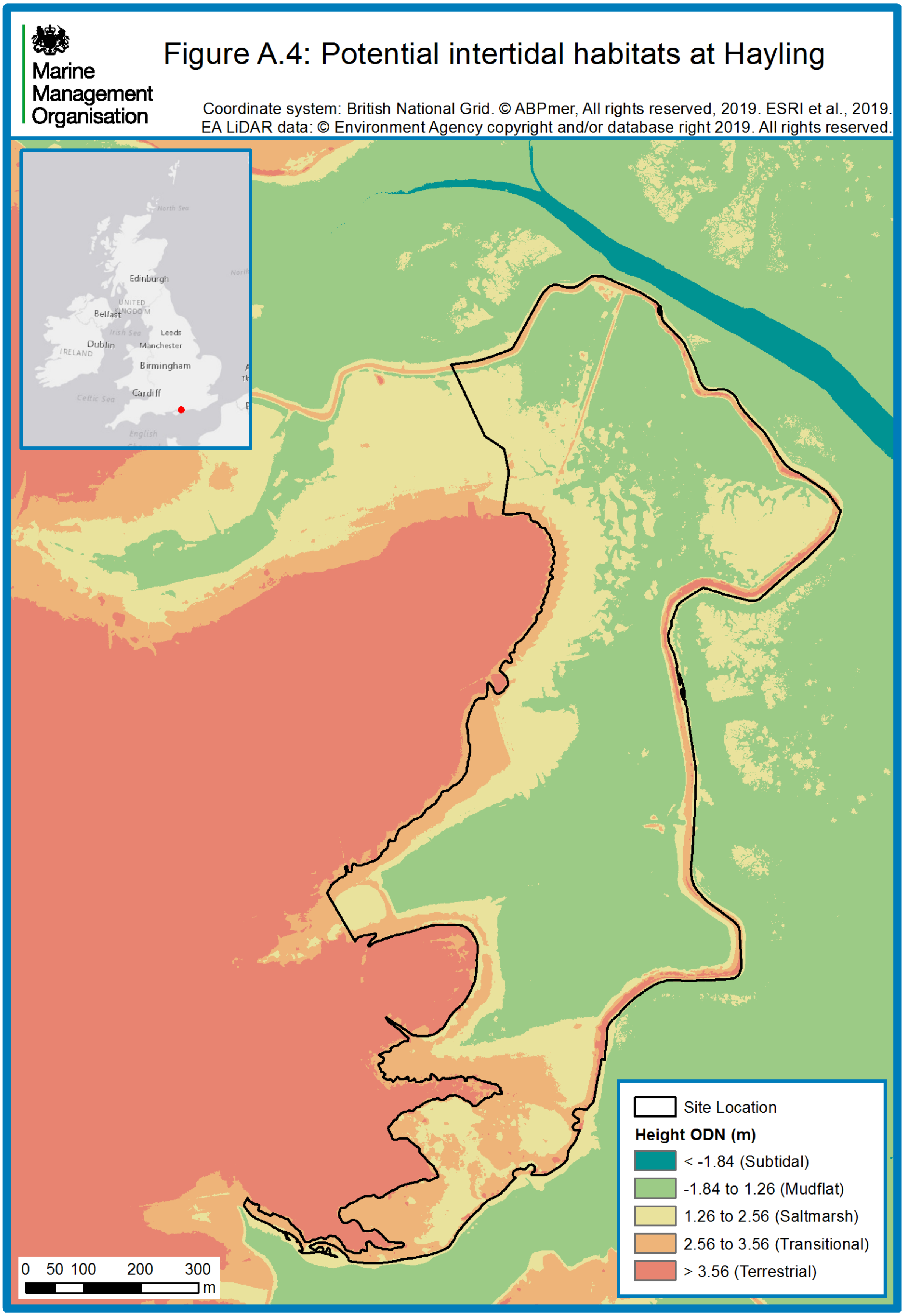

Figure A.20: Potential intertidal habitats at Hayling.

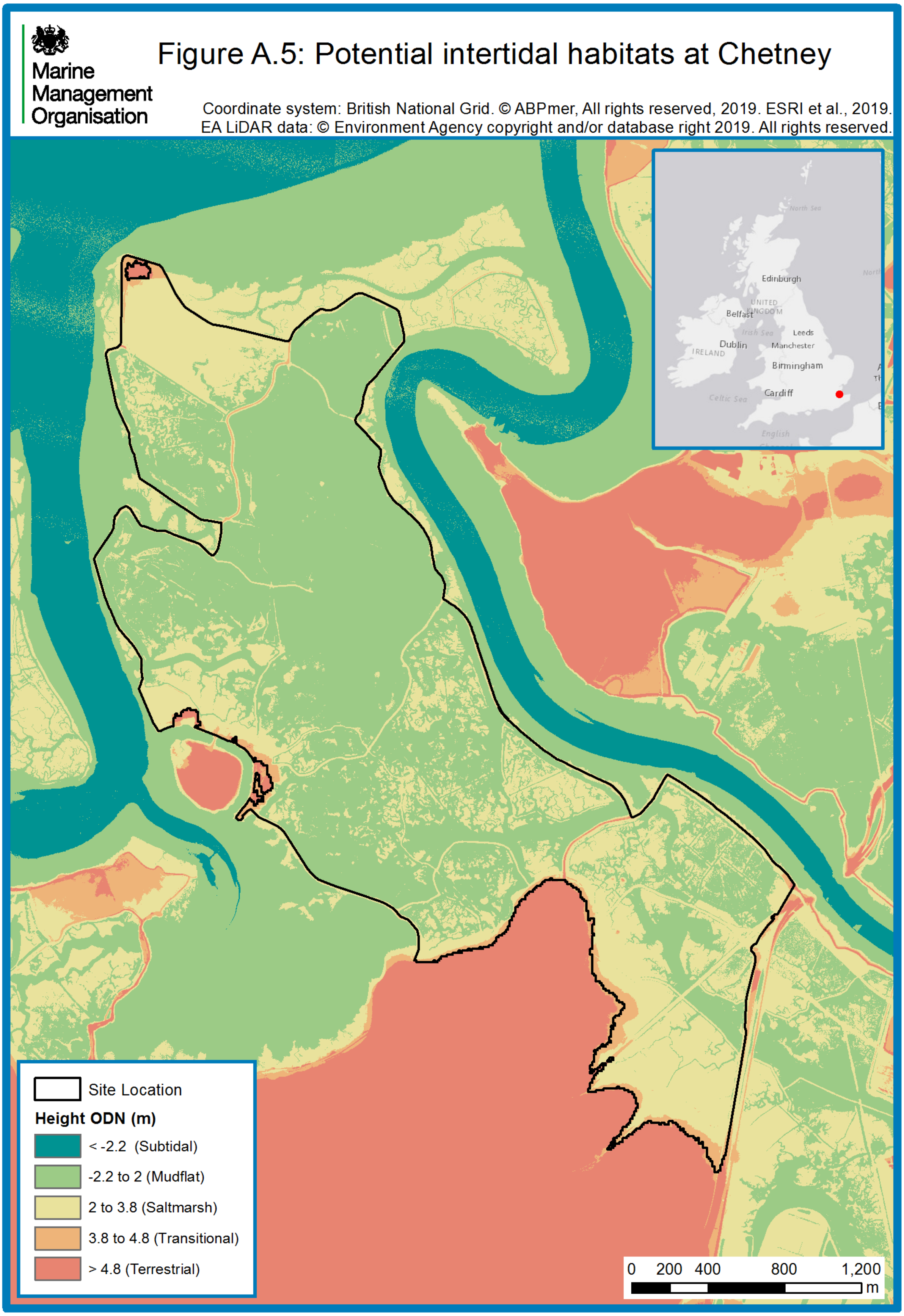

Figure A.21: Potential intertidal habitats at Chetney.

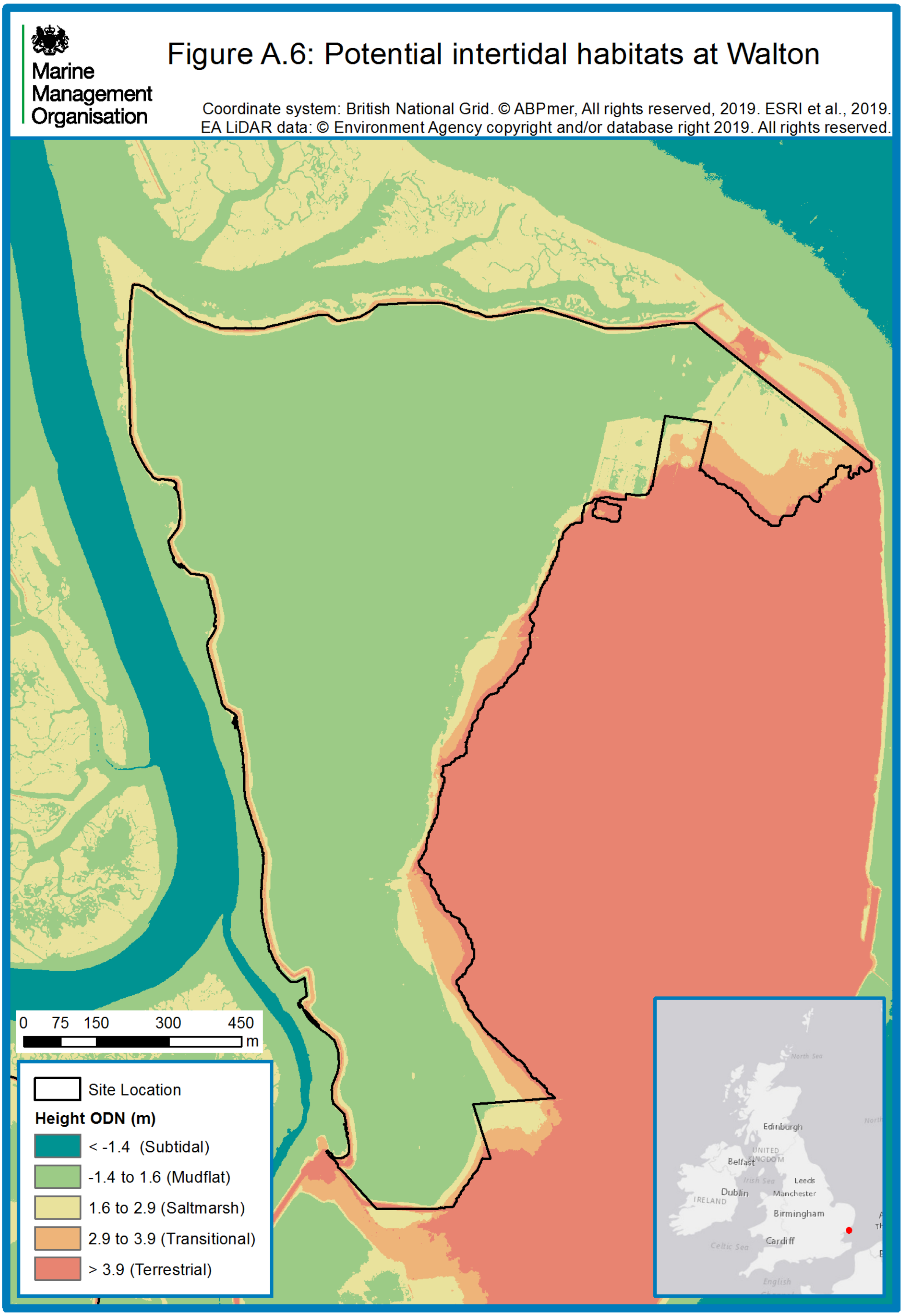

Figure A.22: Potential intertidal habitats at Walton.

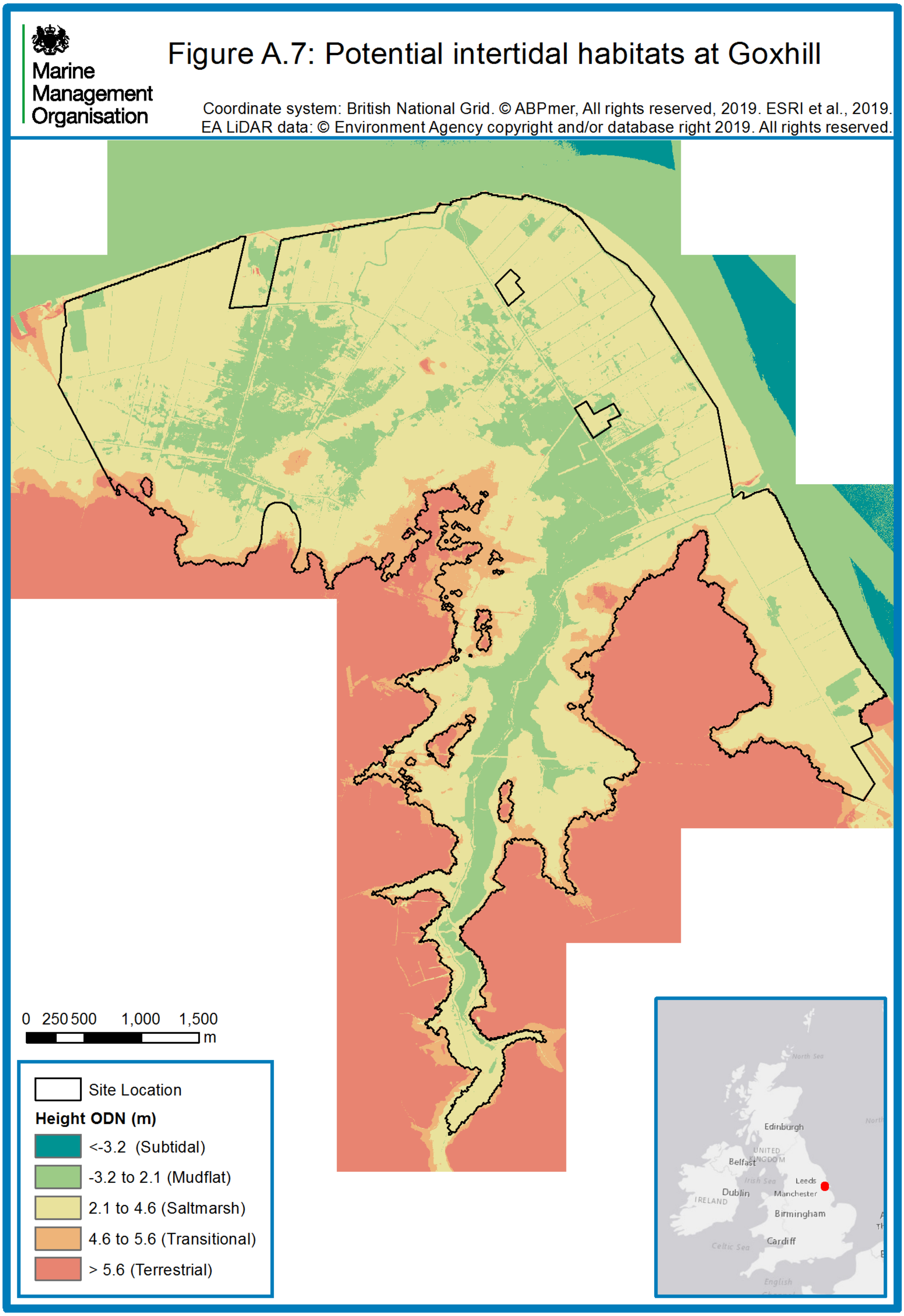

Figure A.23: Potential intertidal habitats at Goxhill.

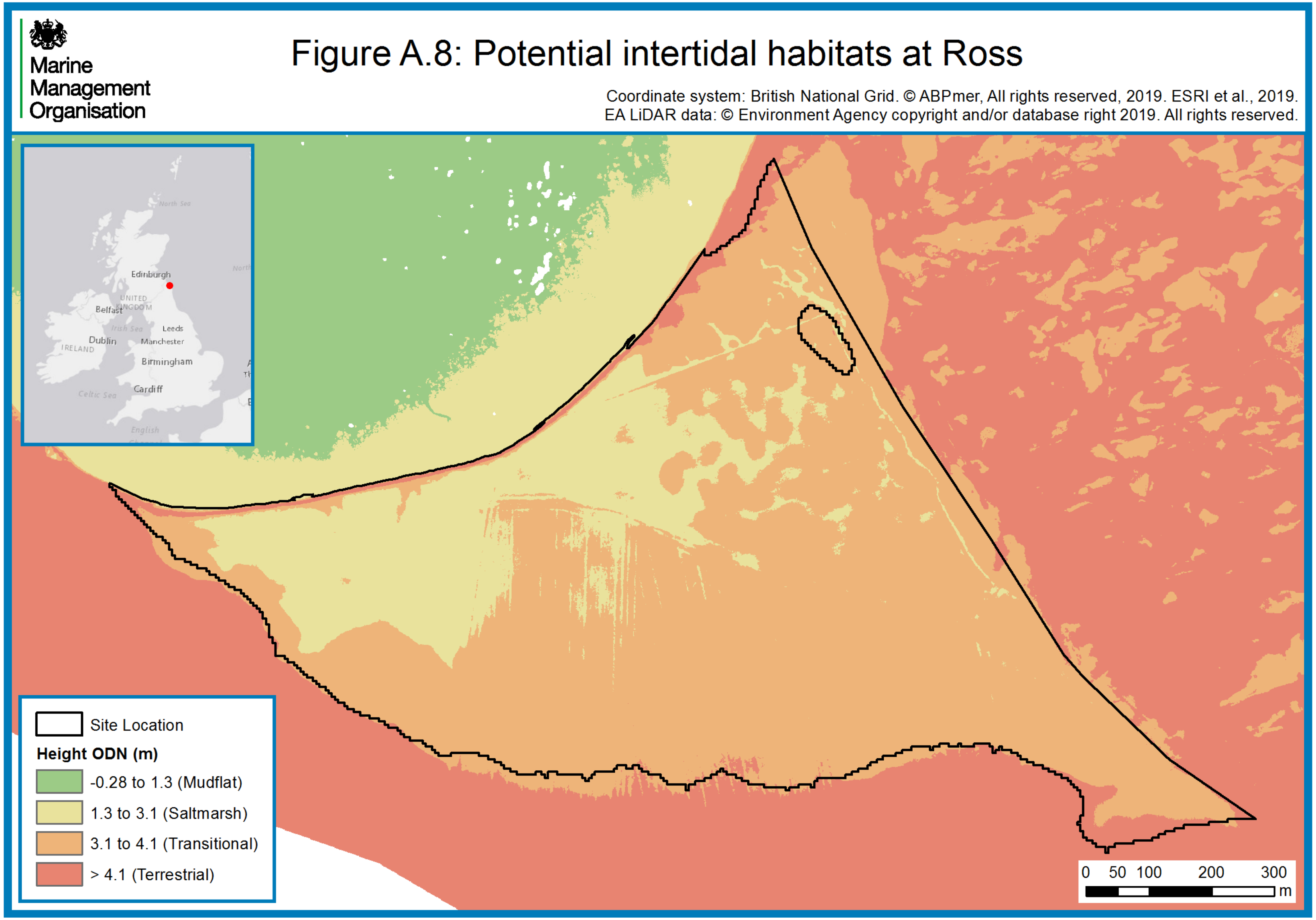

Figure A.24: Potential intertidal habitats at Ross.

Annex B. Potential future seagrass restoration sites

This annex has been prepared by AER to provide background information on the potential seagrass restoration areas which have been mapped in the datalayer 'potential seagrass creation / restoration – historic sites', as outlined in Section 4.2 of the main report. It is structured according to the English counties the potential sites are located in:

- B.1. Cornwall;
- B.2. Devon;
- B.3. Dorset;
- B.4. Hampshire;
- B.5. Kent and Sussex;
- B.6. Essex;
- B.7. Lincolnshire;
- B.8. Yorkshire, County Durham and Tyne & Wear; and
- B.9. Lancashire.

For all of these sites, water quality and conflict with boating would need to be considered prior to restoration efforts being initiated, to ensure the viability of seagrass restoration.

B.1. Cornwall

St Ives and Padstow

There are anecdotal observations of seagrass in St Ives Bay and at Padstow in the Camel River. This information is present within the UN's World Conservation Monitoring Centre's (WCMC) global seagrass database (UNEP-WCMC and Short, 2018). Although there is no direct evidence of there ever having been an abundance of seagrass in either of these locations, environmental conditions appear suitable (sufficient shelter, suitable depth ranges, correct substrate).

Key potential sites for seagrass restoration are:

- St Ives Bay – intertidal and subtidal *Zostera marina*; and
- Padstow - intertidal and subtidal *Z. marina*.

Helford River

The report by Hocking and Tompsett (2002) details all the survey evidence of the loss of all intertidal seagrass meadows (13 discrete meadows) in the Helford River since the 1980s and 1970s. This is possibly due to the combined influences of disturbance from fishing and boating activities, and the problems of eutrophication. The loss of Oysters due to over fishing may also have been an impact upon the seagrass. At present the subtidal *Z. marina* beds remain. There is also some evidence of damage to subtidal seagrass by boat moorings at Durgan, indicating mooring removal and environmentally sustainable replacement as a potential means of allowing for the targeted restoration of seagrass. Mooring related loss has been reported by Unsworth et al. (2017).

Key potential sites for seagrass restoration are:

- Helford and Traeth (numerous possible locations) – *Z. marina* and *Z. noltii* ;
- Helford Passage – intertidal *Z. marina*;
- Lower Calamansack – intertidal *Z. noltii* ;
- Flushing and St Anthony (numerous possible locations) – intertidal *Z. marina*;
- Frenchmen’s creek – intertidal *Z. noltii*;
- Flushing and St Anthony (numerous possible locations) – subtidal *Z. marina*; and
- Durgan Bay – subtidal *Z. marina* (sustainable mooring replacement).

River Fal and Falmouth Harbour

The Fal Estuary has lost at least one intertidal seagrass bed of *Z. marina (*Hocking and Tompsett, (2002) although subtidal *Z. marina* beds are still reported to be present as is one of the *Z. noltii* meadows. This is based on extensive survey and observational evidence. There is some evidence of damage to subtidal *Z. marina* by boat moorings at St Mawes (and other sites) indicating mooring removal and environmentally sustainable replacement as a potential means of allowing for the targeted restoration of seagrass. Mooring related loss has been reported by Unsworth et al*.* (2017).

Key potential sites for seagrass restoration are:

- Cellars Cove, St Mawes – intertidal *Z. marina*; and
- Amsterdam Point, St Mawes – subtidal *Z. marina* (sustainable mooring replacement).

River Fowey and River Looe

Hocking and Tompsett (2002) report that *Z. noltii* has disappeared from its one previous known site in the Fowey at St Winnow. Loss in the Fowey is based on extensive survey and observational evidence. This river estuary is a potential locality for future seagrass, provided water quality is suitable. There is no evidence of loss of seagrass in the River Looe however suitable habitats exist for intertidal restoration efforts. There is some evidence of damage to seagrass by boat moorings in both the River Looe and the River Fowey indicating mooring removal and environmentally sustainable replacement as a potential means of allowing for the targeted restoration of seagrass. Mooring related loss has been reported by Unsworth et al*.* (2017).

Key potential sites for seagrass restoration are:

- St Winnow – intertidal *Z. noltii*;
- Fowey – subtidal *Z. marina*;
- Poluran – subtidal *Z. marina* (sustainable mooring replacement); and
- Looe – subtidal *Z. marina* (sustainable mooring replacement).

B.2 Devon

North Devon - River Torridge

A record of *Z. marina* exists close to the town of Appledore in the Torridge River (UNEP-WCMC and Short, 2018). Whilst no more information is available about seagrass at this site it confirms the potential viability of seagrass in this estuary. Based on high level knowledge of the site, the locations appear to be sufficiently sheltered, the depth ranges are suitable, and the substrate correct to support future seagrass

Key potential sites for seagrass restoration are:

- Appledore - *Z. marina*; and
- Instow – *Z. marina*.

Rivers Tamar, Lynher and Yealm – Devon/Cornwall
Hocking and Tompsett (2002) report that meadows in the Tamar have become smaller and more patchy, this is based upon extensive survey and observational evidence. They report that intertidal *Z. noltii* and *Z. marina* were known to be far more extensive up until the 1970s. They also report that the extent of subtidal *Z. marina* beds has decreased and that they have also become more fragmented (Dale et al*.*, 2007). There may also be extensive potential (based on expert opinion) habitat for *Z. marina* and *Z. noltii* in the Tamar River and the Lynher River. Historic use of Plymouth sound as a major maritime base, large land reclamation and extensive agriculture may have contributed to seagrass loss over time. With improvements to water quality over the last 2 decades, conditions may again be suitable for seagrass; however, this would need to be assessed to ensure viable conditions for restoration.

There may be extensive potential habitat for *Z. noltii* in the Yealm River. Specific localities of the mud banks near to Conflate creek offer great potential as future seagrass sites. There is also some evidence of damage to seagrass in the Yealm by boat moorings, indicating mooring removal and environmentally sustainable replacement as a potential means of allowing for the targeted restoration of seagrass. Mooring related loss has been reported by Unsworth et al*.* (2017).

Key potential sites for seagrass restoration are:

- St Johns Bay – intertidal *Z. noltii* ;
- Millbrook Bay – intertidal *Z. noltii*;
- Yealm River near Conflate Creek – intertidal *Z. noltii*;
- Yealm River - subtidal *Z. marina* (sustainable mooring replacement);
- Cawsand Bay – subtidal *Z. marina*; and
- Drakes Island – subtidal *Z. marina*.

Kingsbridge Estuary
There may be extensive potential habitat for *Z. noltii* in the Kingsbridge Estuary. Specific localities of the mud banks near to West Charleton and Lincombe offer great potential as future seagrass sites. There is some evidence that the intertidal and subtidal *Z. marina* near Salcombe, on either side of the estuary, has decreased in area. There is some evidence of damage to seagrass by boat moorings (Unsworth et al*.,* 2017), indicating mooring removal and environmentally sustainable replacement as a potential means of allowing for the targeted restoration of seagrass. There are scattered recent records of seagrass in the River Dart (UNEP-WCMC and Short, 2018), and a survey in 2005 indicates evidence of loss or decline in populations near to Greenway Quay (SEASEARCH, 2005). Although there is no hard evidence of there ever being an abundance of seagrass in either of these locations, environmental conditions appear suitable (sufficient shelter, suitable depth ranges, correct substrate).

Key potential sites for seagrass restoration are:

- West Charleton and Lincombe –*Z. marina* and *Z. noltii*;
- Salcombe - *Z. noltii* and *Z. marina* (sustainable mooring replacement); and
- Dartmouth - *Z. marina*.

Torbay

There are numerous small meadows throughout Torbay, many of which are fragmented (Hirst and Attrill, 2008), however there is only one explicit example of seagrass loss in the Bay. In 2006 a scallop dredger removed a large area of *Z. marina* from Fishcombe Cove in the middle of the bed, which has struggled to recover ever since. This loss provides a potential means of allowing for the targeted restoration of seagrass at the site. Given the continued presence of seagrass at the site (in other areas), water quality is not expected to be a problem.

Key potential sites for seagrass restoration:

- Fishcombe Cove – subtidal *Z. marina*.

Teign River

According to a report by the local Teign council 'Nature's Future: A Biodiversity Action Plan for Teignbridge' (Future, 2008) there are anecdotal observations of intertidal *Zostera* sp. in the Teign Estuary prior to the construction of a pipeline (early 1990s). These were believed to be in between Buckland sewage treatment works and Coombe cellars. It is not clear whether the species was *Z. noltii* or *Z. marina*. This is no longer present. There may be extensive potential habitat for subtidal *Z. marina* and intertidal *Z. noltii* and/or *marina* (based on expert opinion) in the Teign River. It would be very unusual for any estuary in the south of England not to have had seagrass at one time in the past.

Key potential sites for intertidal to subtidal seagrass restoration are:

- Between Buckland sewage treatment works and Coombe cellars – intertidal *Z. noltii* and/or *marina*; and
- Teignmouth – subtidal *Z. marina*.

B.3 Dorset

Portland Harbour

There may be extensive potential (based on expert opinion) habitat for subtidal *Z. marina* in the Portland harbour beyond present distribution (UNEP-WCMC and Short, 2018). There is some evidence of damage to seagrass by boat moorings indicating mooring removal and environmentally sustainable replacement as a potential means of allowing for the targeted restoration of seagrass. Mooring related loss has been reported by Unsworth et al. (2017). Given the man-made nature of this environment and its status as an active port there might be key boating related concerns related to using this site for restoration.

The key potential site for seagrass restoration is:

- Portland Harbour - subtidal *Z. marina*.

Poole Harbour

There may be extensive potential (based on expert opinion) habitat for subtidal *Z. marina* in the Poole harbour beyond present distribution (Envision Mapping, 2015). It appears that there has been long-term loss, but limited data to determine this. There is some evidence of damage to seagrass by boat moorings indicating mooring removal and environmentally sustainable replacement as a potential means of allowing for the targeted restoration of seagrass. Mooring related loss has been reported by Unsworth et al*.* (2017).

The key potential sites for seagrass restoration are:

- Sandbanks and Brownsea Island – intertidal and subtidal *Z. marina* and noltii (sustainable mooring replacement).

B.4 Hampshire and the Isle of Wight

The Solent and the Isle of Wight

Extensive loss of seagrass has been documented historically in Southampton Water. Butcher (1934) wrote that before the outbreak of 'wasting disease', in Southampton Water, eelgrass was formerly very large and abundant from Southampton up the River Hamble to Bursledon; along the opposite side from Eling south-eastwards and the Beaulieu and Lymington Rivers. Anecdotal evidence collected by Tubbs (1999) suggests prolific subtidal and intertidal *Zostera* beds in Portsmouth, Langstone and Chichester Harbours. The situation was similar on the Isle of Wight, with significant meadows recorded on the north east and north-west coasts. Whilst some areas subsequently recovered, today, seagrass remains absent in Lymington River, and beds present in Southampton Water are still vastly reduced and thought to now only occur only around Chilling. The Hamble was extensively surveyed in 2011 and no seagrass was recorded. The beds in Langstone and Chichester Harbours continue to exist but may have been impacted by further outbreaks of wasting disease. Other significant populations occur at Portsmouth Harbour, Beaulieu and Calshot off Hampshire, Totland, Yarmouth and Wootton on the north-west coast of the Isle of Wight and large beds extending down the north east coast of the Isle of Wight from Cowes to Bembridge (Marsden and Chesworth, 2015).

Key potential sites for seagrass restoration are:

- Beaulieu Estuary – intertidal *Z. noltii* or *marina*;
- Lymington River – intertidal *Z. noltii* or *marina*;
- Medina and Yar Rivers – intertidal *Z. noltii* or *marina*;
- Keyhaven Marshes - intertidal *Z. noltii* or *marina*, and subtidal *Z. marina*;
- Portsmouth Harbour - intertidal *Z. noltii* or *marina*, and subtidal *Z. marina*; and
- Langstone Harbour - intertidal *Z. noltii* or *marina*, and subtidal *Z. marina*.

B.5 Kent and Sussex

Chichester Harbour
Chichester Harbour is the most westerly site of seagrass in the area of Kent and Suffolk, containing a series of meadows mapped since 2006 (Marsden and Chesworth, 2015). This harbour no longer contains any subtidal distributions of *Z. marina* and intertidal meadows of *Z. noltii* and *Z. marina* are not as extensive as once recorded. Seagrass is extensive at the mouth of the Pagham River (UNEP-WCMC and Short, 2018), however the extensive sand and mud flats create a range of potential sites for restoration within this waterway.

Key potential sites for seagrass restoration are:

- Chichester Langstone harbour - intertidal *Z. noltii* or *marina*, and subtidal *Z. marina*; and
- Pagham River- intertidal *Z. noltii* or *marina*.

Thames and Medway
Within the southern Thames, Zostera distribution in the Medway remains unclear however extensive historical information about its presence can be found. This area presents a range of opportunities for restoration of seagrass as there are extensive sheltered mudflats available for seagrass colonisation.

Key potential sites for seagrass restoration are:

- Whitstable (Grassy Sand) - intertidal to subtidal *Z. marina*; and
- The Medway - intertidal *Z. noltii* or *marina*.

B.6 Essex

Deben River
Seagrass in the Deben is clearly mentioned in the 1933 review by Butcher, with extensive distribution of seagrass between Bawdsey Ferry up to Woodbridge suggesting an almost estuary wide distribution (Butcher, 1933). These areas present a range of opportunities for restoration of seagrass as there are extensive sheltered mudflats available for seagrass colonisation.

Key potential sites for seagrass restoration are:

- Bawdsey - intertidal *Z. noltii* or *marina*; and
- Waldringfield - intertidal *Z. noltii* or *marina*.

Rivers Stour and Orwell
A series of historic reports document seagrass loss in the River Stour (Butcher, 1933, Butcher, 1941, Burton, 1961) with indications that there is now very little if any left (Jackson et al., 2016). Previous descriptions refer to extensive seagrass between Manningtree and Wrabness, as well as seagrass at Lower Holbrook. Locations of seagrass loss are less clearly defined within the Orwell. Given the slow decline of the seagrass in these sites due to water quality caution is advised as to whether restoration is worth pursuing.

Key potential sites for seagrass restoration are:

- Manningtree to Wrabness - intertidal *Z. noltii* or *marina*;
- Lower Holbrook - intertidal *Z. noltii* or *marina*;
- Levringham Creek - intertidal *Z. noltii* or *marina*;
- Woolverstone - intertidal *Z. noltii* or *marina*; and
- Pin Mill - intertidal *Z. noltii* or *marina*.

Blackwater Estuary
Jackson et al*.* (2016) provides information about the previous presence of *Z. marina* at Osea Island, this site was referred to in the 1934 review of seagrass by Butcher (1933). In addition Jackson et al*.* (2016) also refer to seagrass no longer being present at Goldhanger and St Lawrence. These areas present a range of opportunities for restoration of seagrass as there are extensive sheltered mudflats available for seagrass colonisation.

Key potential sites for seagrass restoration are:

- Osea Island - intertidal *Z. noltii* or *marina*;
- Goldhanger - intertidal *Z. noltii* or *marina*; and
- St Lawrence - intertidal *Z. noltii* or *marina*.

River Colne
Jackson et al*.* (2016) provides information about the potential current and previous presence of *Z. marina* at Point Clear. In addition, an historic record of seagrass at West Mersea Beach is also described in Lewis (1962). These two areas present a range of opportunities for restoration of seagrass as there are extensive sheltered mudflats available for seagrass colonisation.

Key potential sites for seagrass restoration:

- Point Clear - intertidal *Z. noltii* or *marina*; and
- Mersea Island - intertidal *Z. noltii* or *marina*.

B.7 Lincolnshire

Although there are extensive records of seagrass in The Wash and indications of its loss, it is not explicitly clear where such loss occurred. No recommendations for restoration sites are thus provided.

B.8 Yorkshire, County Durham and Tyne & Wear

Humber Estuary
Numerous authors and reports over time have concluded that seagrass has reduced in area and density at a number of sites towards the mouth of the Humber Estuary (Woodward et al*.*, 2014; Phillip, 1936). Particular loss of seagrass has occurred on the north side of the estuary at Spurn Head and on the southern side towards Cleethorpes and Horseshoe Point. Sites at Spurn Head and Horseshoe Point provide locations for future seagrass restoration, however for this to be viable suitable water quality will need to be extensively considered (noting that the recent

erosion at Spurn Head might well preclude such restoration if a chosen location were to be exposed to too much wave energy[19]).

Key potential sites for seagrass restoration are:

- Spurn Head –*Z. marina* and *Z. noltii*; and
- Horseshoe Point and Cleethorpes –*Z. marina* and *Z. noltii*.

Tees Estuary and Hartlepool

There is evidence of the previous presence of intertidal *Zostera* sp. (Cleveland Naturalists' Field Club, 1994). In the Tees Estuary although this is limited and anecdotal, and it is likely that observations at Eston were prior to land reclamation. The physical environment of the Tees suggests the potential for future seagrass along the sheltered coast of the Teesmouth Nature Reserve and adjacent to the Conoco plant. The shelter created by the bay at Hartlepool also creates potential future seagrass habitat as historic records are available for its presence at this site. There are no historic records of seagrass from the Tyne River, however these are all from areas now reclaimed from the sea. The Tyne is unlikely now to provide much substrate useful for restoration.

Key potential sites for seagrass restoration are:

- Teesmouth Nature Reserve - subtidal *Z. marina*;
- Off from the Seaton on Tees Channel - intertidal *Z. noltii* or *marina*;
- Seaton Carew - subtidal *Z. marina*; and
- Hartlepool - subtidal *Z. marina*.

Blyth River

No seagrass is now known from the Blyth River (Yorkshire) but historic observations are known from Baker and Tate (1867) and more recently in an environmental report by Biofuelwatch (2011). The protection of this site suggests its potential for future seagrass viability

The key potential site for seagrass restoration is:

- Blyth River port - intertidal *Z. noltii* or *marina*.

B.9 Lancashire

Walney

Seagrass records in Lancashire are mostly limited to the Barrow/Walney area although the meadows are small (UNEP-WCMC and Short, 2018). Given the prevailing exposure of the Lancashire coast to South Westerly winds there are only limited locations for potential future seagrass with the Walney area likely to be the most plausible locality.

The key potential site for seagrass restoration is:

- Walney - intertidal *Z. noltii* or *marina*.

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[19] See, for example, Yorkshire Post (2016).