Blue Carbon
and Marine Carbon Sequestration in Irish Waters and Coastal Habitats

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TABLE OF CONTENTS

1 SUMMARY

Atmospheric CO_2 is rising globally. Opportunities for reducing this trend include energy sector adjustments and management of both land and ocean resources. Improved management of coastal and oceanic ecosystems is therefore poised to contribute to, and enhance, climate mitigation and adaptation. This report outlines the emergence of blue carbon as a concept for the integration of coastal carbon dynamics into policy and management frameworks and defines blue carbon ecosystems. It also emphasises the importance of marine carbon sequestration and highlights its potential role in climate adaptation. Ireland is estimated to store at least 9.2 Mt of carbon in its saltmarsh and seagrass habitats, which cover an estimated minimum area of 162 km². Estimates of carbon stocks in potential blue carbon ecosystems such as macroalgae beds are hampered by lack of data on extent, productivity and actual contribution. Irish coastal blue carbon ecosystems and their carbon sequestration capacity are currently threatened by anthropogenic factors such as land reclamation and poor water quality. The possibility of including saltmarsh and seagrass habitats in Ireland's National Inventory Report on GHG emissions to the United Nations Framework Convention on Climate Change (UNFCCC) and including Ireland's potential blue carbon ecosystems in Ireland's Nationally Determined Contributions is highlighted. The critical knowledge gaps and future research priorities are outlined, so that Ireland can advance the pace of scientific discovery whilst harnessing the climate change potential of its coastal and marine environment.

2 GLOSSARY OF TERMS

Allochthonous Carbon - Carbon produced in one location and deposited in another. In the context of blue carbon systems, this type of carbon results from the hydrodynamic environment in which it is found, where sediments and associated carbon are transported from neighbouring ecosystems (offshore and terrestrial).

Autochthonous Carbon - Carbon produced and deposited in the same location. In the context of blue carbon systems, this type of carbon results from vegetation uptake of CO_2 from the ocean and/or atmosphere that is converted for use by plant tissue and decomposes into the surrounding soil.

Carbon pool - Carbon reservoirs such as soil, vegetation, water and the atmosphere that contain a measurable mass of carbon. Carbon can be imported (gained) or exported (lost) from one carbon pool to another.

Carbon stock - The total amount of organic carbon stored in a blue carbon ecosystem of a known size. A carbon stock is the sum of one or more carbon pools.

Emission factor - A categorised model for estimating GHG flux rate changes from a predefined area due to change in land coverage and use

LULUCF - Land-Use, Land-Use Change and Forestry

POC - Particulate Organic Carbon

DOC - Dissolved Organic Carbon

PIC - Particulate Inorganic Carbon

GWP – Greenhouse Warming Potential

Soil Organic Carbon – The carbon component of organic matter in soils

Soil Inorganic Carbon - The carbon component of inorganic matter in soils. In blue carbon ecosystems it is dominated by carbonates (e.g. calcium carbonate) in the form of shells or coral fragments and carbonate precipitates.

3 EMERGENCE OF BLUE CARBON

Oceans and coastal marine systems play a significant role in the global carbon cycle. They represent the largest long-term sink for carbon, whilst also storing and redistributing CO_2. Some 93% of the earth's CO_2 (40 Tt) is stored and cycled through the oceans (Nelleman et al., 2009). Blue carbon emerged as a concept in 2009, stemming back to reports by the United Nations Environment Programme (UNEP) and the International Union for Conservation of Nature (IUCN) to describe the carbon that is captured and stored by the oceans and, in particular, the carbon stored by vegetated coastal habitats (Nelleman et al., 2009; Laffoley & Grimsditch, 2009). The concept developed as part of the 'colour framework' to distinguish carbon fluxes that have different causes and implications, including green carbon of forests and black carbon of atmospheric particulates. Although occupying less than 2% of ocean area (<5 % global land area), vegetated coastal habitats are estimated to account for up to 50% of carbon burial in marine sediments (Duarte, 2005), representing a large transfer of carbon from atmosphere and surface waters to long-term sediment storage. The realisation that vegetated coastal habitats support globally relevant rates of organic carbon burial has led to the development of strategies to mitigate climate change through the conservation and restoration of saltmarsh, mangrove and seagrass habitats, termed 'blue carbon strategies' (Nelleman et al., 2009; Mcleod et al., 2011; Duarte et al., 2013). Furthermore, inherent in the blue carbon concept is the recognition that the destruction of blue carbon ecosystems results in the fluxes of historical carbon pools back to the atmosphere as carbon dioxide (Pendleton et al., 2012). Ultimately, through management, blue carbon strategies aim to protect the sink of carbon and ensure a source of carbon to the atmosphere is not created.

Ireland has a vast marine territory of approximately 880,000 km². It is, therefore, critical that Ireland can leverage the carbon sequestration potential of its marine resource, ensuring it is appropriately managed to continue functioning as a carbon sink and that no additional carbon sources are created through mismanagement.
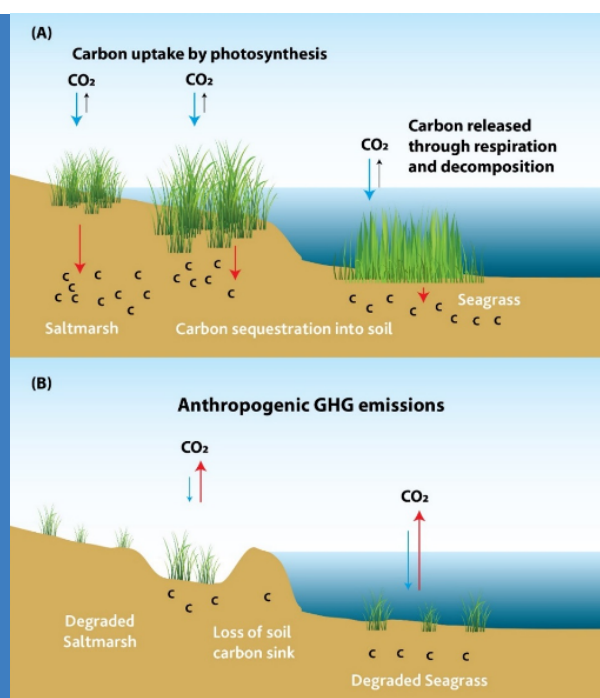

Figure 1. In intact coastal wetlands (saltmarsh and seagrass habitats), carbon is taken up via photosynthesis (blue arrows) where it is sequestered long term into biomass and soil organic matter (red arrows) or respired (black arrows). (b) When coastal wetlands are degraded (e.g. drained of water), the carbon stored in the soils is consumed by microorganisms, which respire and release CO_2 as a metabolic waste product. Human activity that degrades coastal blue carbon ecosystems or converts them to other land uses (e.g. drainage or impoundment for agriculture) results in a reduction in CO_2 uptake due to the loss of vegetation (blue arrows) and the release of globally important GHG emissions (red arrows). Adapted from Howard et al. (2017).

4 DEFINING BLUE CARBON ECOSYSTEMS AND MARINE CARBON ECOSYSTEMS

In the context of climate policy frameworks, blue carbon has been defined as the carbon accumulating in vegetated, tidally influenced coastal ecosystems such as tidal marshes, tidal forests (including mangroves), and seagrass meadows (International Blue Carbon Science Working Group, 2015). Climate-relevant blue carbon pools include the carbon stored in sediments and soils, waters, living biomass and non-living biomass. In addition, climate relevant fluxes include fluxes of carbon between the pools, as well as GHG exchanges with the atmosphere.

4.1 Blue carbon ecosystems that can connect management to climate policy frameworks

The defining of blue carbon ecosystems is underpinned by the fact that the carbon is sequestered *in situ*, i.e. there is a fixed pool of carbon and, in addition, there is clear ownership of this resource and this resource can be managed. Specifically, for inclusion as a blue carbon ecosystem in policy frameworks there is a set of 6 criteria (Table 1).

Table 1.
Inclusion criteria for recognition as a blue carbon ecosystem (Crooks et al., 2019)

1	Rates of carbon sequestration and/ or prevention of emissions of GHGs by the ecosystem is at sufficient scales to influence climate
2	Major carbon stocks and/or, changes in stocks and fluxes of GHGs can be quantified spatially and temporally
3	Anthropogenic drivers are impacting carbon storage, stock change, or GHG emissions
4	Management of the ecosystem to improve sequestration or emission reductions is possible and practicable
5	Interventions can be achieved without causing environmental or social harm
6	Management actions can be aligned with existing or developing policy and national commitments to address climate change

Four major habitat types recognised as blue carbon ecosystems (Crooks et al., 2019) fall into categories based primarily on their vegetation. Two of these, namely tidal marsh and seagrass, are relevant for Ireland.

1. SALTMARSHES

Saltmarshes are coastal habitats that are regularly flooded by the tide, dominated by halophilic grasses, grass-like rushes, and annual and perennial flowering plants (e.g. Salicornia spp.) and shrubs. The plant composition of tidal marshes is influenced by salinity and flooding frequency. Saltmarshes are being lost at a rate of 1-2% per year globally (Pendelton et al., 2012). They cover roughly 140 million hectares of Earth's surface, but more than 50% of their historical global coverage has been lost. Ireland has approximately 100 km^2 of saltmarsh habitat (Mcowen et al., 2017).

2. SEAGRASS MEADOWS

Seagrasses are flowering plants (i.e. angiosperms) that grow in shallow marine environments. They cover less than 0.2% of the ocean floor, but store about 10% of the carbon buried in the oceans each year. Seagrasses are being lost at a rate of 1.5 -2.5% per year and have lost approximately 30-50% of historical global coverage (Waycott et al., 2009; Pendelton et al., 2012). It is estimated that there are at least 62 km^2 of seagrass habitat in Ireland but the actual extent remains to be determined (Beca-Carretero et al., 2019, 2020).

3. MANGROVE FORESTS

A mangrove is a tree or shrub that grows in the intertidal zone of marine and estuarine environments in predominantly tropical climates. Globally about 35% of the area of mangrove coverage has disappeared since 1980, with a current global areal rate of loss of between 0.7 and 3% per year (Pendelton et al., 2012). These ecosystems are not found in Ireland.

4. TIDAL FRESHWATER FORESTS

Tidal freshwater forests are communities of trees and associated plants in low-lying coastal settings and at the head of estuaries where freshwater rises and falls under the influence of the tide. Aside from some small fragments of willow-nettle woodland that are subject to freshwater tidal inundation (Cross & Collins, 2017) Ireland does not have any significant areas of tidal freshwater forests. However, there are more substantial areas of reed swamp which are located at the heads of estuaries or in adjacent coastal floodplains.
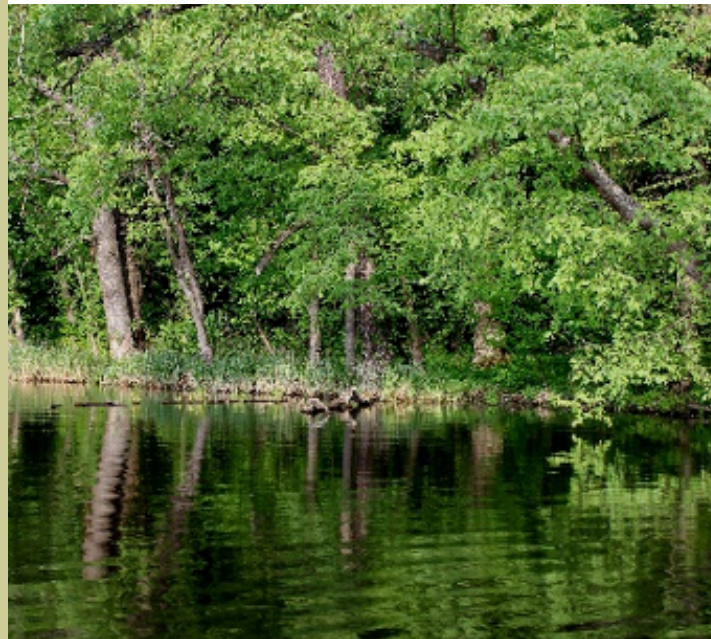
4.2 Potential Blue Carbon Ecosystems

Other marine ecosystems meet some, but not all, of the above criteria and are therefore not fully recognised as blue carbon ecosystems in a climate policy context. They are primarily located in open-water, fully marine systems and comprise macroalgae (including maërl), phytoplankton and cold-water corals. These ecosystems are an essential part of the biosphere's process of capturing, fixing and transferring carbon dioxide to long-term storage in sediments and deep ocean water bodies (Smale et al., 2018). However, the complexity of measuring and accounting for this carbon sequestration remains a barrier to managing these carbon pools for the purpose of greenhouse gas mitigation. Furthermore, from a management perspective it is likely that this carbon sequestration is cross-jurisdictional, presenting another challenge in integrating these ecosystems into policy frameworks. However, the challenges represent an opportunity for scientists and policy makers to work together to optimise the climate-related potential of these systems. Further details on these ecosystems are provided in the following sections.

5 STATE OF SCIENCE IN BLUE CARBON ECOSYSTEMS

5.1 Current global estimates of carbon sequestration and sinks

Although seagrass meadows and saltmarshes represent a much smaller area than terrestrial forests (Table 2), their total contribution to long-term carbon sequestration is comparable to carbon sinks in terrestrial ecosystem types. The total global carbon burial is estimated at 5-87 Tg C yr^{-1}, and 41-112 Tg C yr^{-1} for saltmarshes and seagrasses, respectively (Table 2, Mcleod et al., 2011). These global carbon burial rates are comparable to those of terrestrial forest types (Table 2). The relatively low above-ground biomass and areal coverage of vegetated coastal ecosystems is compensated by high rates of carbon sequestration in soils resulting in very high rates of organic carbon sequestration on a per area basis (Figure 2). As such blue carbon ecosystems have the potential to contribute substantially to long-term carbon sequestration (Table 2). However, global losses of vegetated coastal ecosystems threaten their ability to function as long-term carbon sinks.

Table 2.
Carbon storage potential of coastal vegetated ecosystems (Source global aerial extent: Mcowen et al., 2017; Jayathilake & Costello, 2018; Davidson & Finlayson, 2019; Source carbon values: Mcleod et al. 2011, and references therein; Source conversion rates: Pendelton et al., 2012; Waycott et al. 2009).

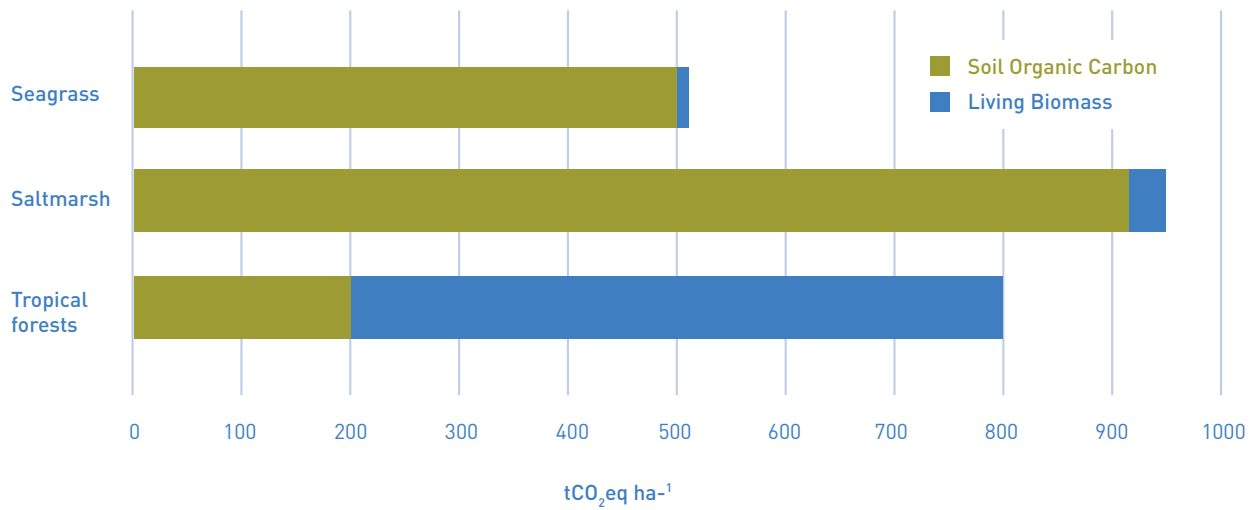|	Global Aerial Extent	Total carbon sequestered annually	Mean global estimate of carbon stock	Anthropogenic conversion rate	Potential emissions due to anthropogenic conversion
	km^2	Tg C yr^{-1}	Tg C	% yr^{-1}	Tg CO$_2$
Saltmarsh	55,000	4.8 - 87.2	570 - 10,360	1.0 - 2.0	20.9 -760.4
Seagrass	788,000 - 1,646,788	41.4 - 82.8	4,260 - 8,520	1.5 – 2.5	62.5 - 813
Tropical forests	19,622,846	78.5	–	–	–

Figure 2.
Global averages of carbon pools on a per area basis in blue carbon habitats; seagrass beds and saltmarshes in comparison to tropical forests. Adapted from Murray et al. (2010).

5.2 Physical and biological regulation of carbon sequestration in saltmarshes

The rate of carbon sequestration in tidal marshes is regulated by complex feedbacks between biological and physical factors including biomass production, tidal amplitude and the concentration of suspended sediments. Of these, carbon sequestration is most affected by biomass productivity and tidal amplitude (Morris & Callaway, 2019). Above-ground plant growth and stem density affect elevation gain through surface accretion, as it slows down water movements and allows for the flocculation and deposition of sediments (Figure 3). In addition, root growth enables elevation gain through the addition of organic carbon and subsequent sub-surface expansion. Tidal inundation maintains anoxic conditions, which enables the preservation of carbon over long-term timescales. Furthermore, in contrast to terrestrial habitats, tidal wetland carbon stocks grow volumetrically and are thus far less prone to carbon saturation (Morris et al., 2012), a condition in which the amount of carbon a soil can sequester reaches a limit.

5.3 Greenhouse gas fluxes in saltmarshes

Methane (CH_4) and nitrous oxide (N_2O) are potent greenhouse gases. Methane has a global warming potential (GWP) 32 times that of CO_2 (Neubauer & Megonigal, 2015), and N_2O has a GWP of 298 (Myhre et al., 2013). Even relatively small fluxes of these gases from blue carbon ecosystems can offset the uptake of CO_2 and the subsequent storage of carbon. The production and consumption of methane and nitrous oxide are primarily regulated by microbial activities and rates are affected by environmental factors including water level, plant traits and the amount of organic matter present in the soil.
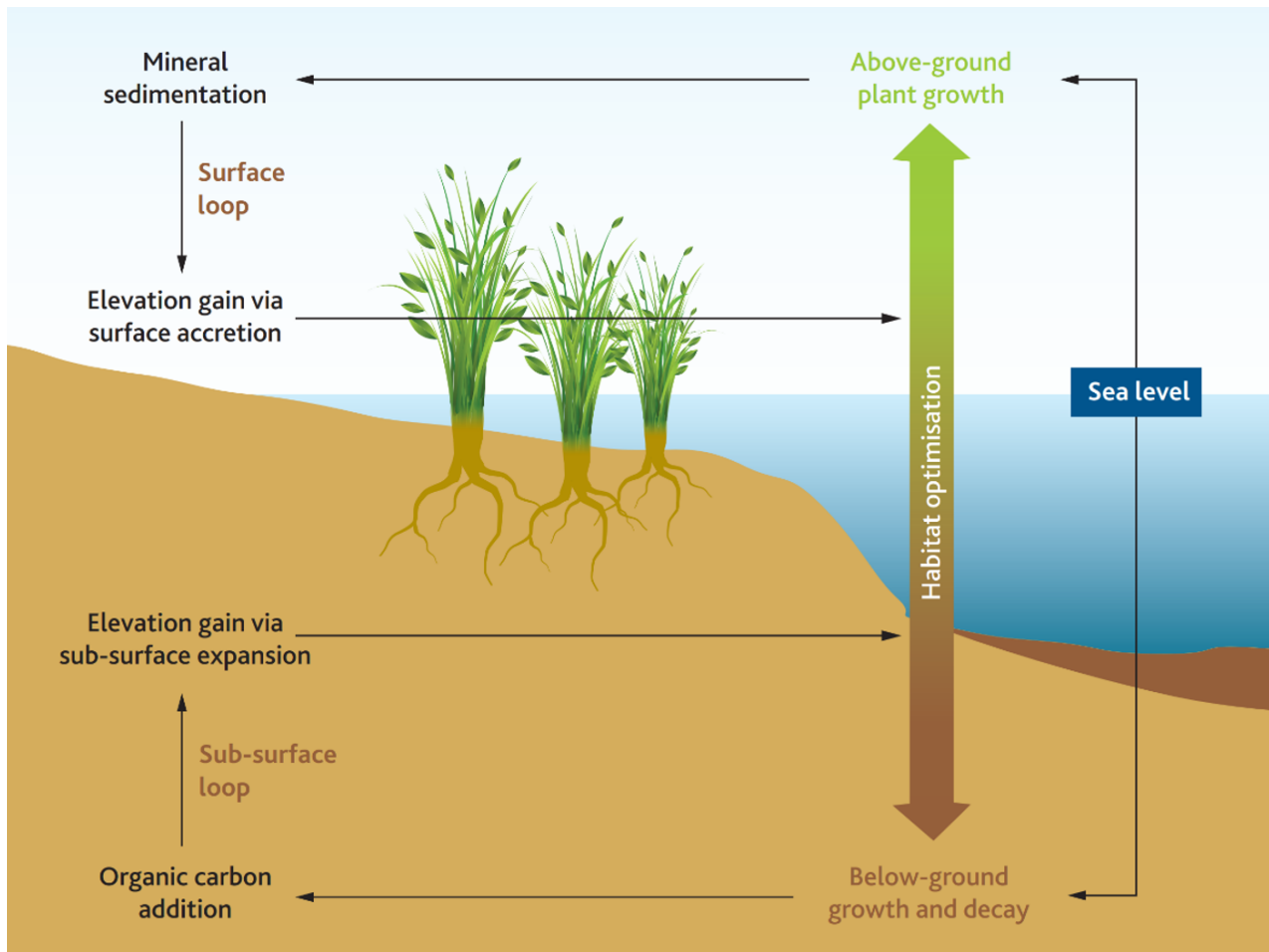
Figure 3.
Complex feedbacks between plant growth and tidal inundation in saltmarshes.
Adapted from Kirwan & Megonigal (2013).

Methane is the product of the decomposition of organic matter under anaerobic conditions, such as in waterlogged wetland sediments, where very little oxygen is present. The net flux of methane to the atmosphere is determined by the net flux of methane produced (methanogenesis) minus the amount of methane that is either consumed (methanotrophy) or chemically oxidized before reaching the atmosphere. Plant-mediated transport can be an important flux of methane, whereby the plant essentially acts as a conduit facilitating the transport of methane from the soil to the atmosphere. A key consideration regarding methane fluxes in blue carbon ecosystems is the interaction with seawater. Seawater is high in sulfate and sulfate-reducing bacteria compete with methanogenic bacteria ultimately reducing the production of methane. This sulfate suppression of methane is what makes blue carbon ecosystems more appealing as carbon sinks than their freshwater counterparts such as peatbogs (Keller, 2019). Salinity has been used as proxy for sulfate availability (Poffenbarger et al., 2011), and it has been demonstrated that marshes above a salinity of 18 ppt have minimal methane emissions. Global methane flux from saltmarshes has been estimated at 0.071 ± 0.027 Tmol CH_4 year^{-1} and from seagrass 0.031 ± 0.006 Tmol CH_4 year^{-1} (Al-Haj & Fulweiler, 2020).

Tidal marsh N_2O emissions are predominantly derived from the microbial processes of nitrification and denitrification. In coastal wetlands the majority of in situ N_2O production occurs as a result of sediment denitrification, although the water column contributes N_2O through nitrification in suspended particles (Murray et al., 2015). The most important factors controlling N_2O fluxes are inputs of inorganic nitrogen and oxygen availability, which in turn are affected by tidal cycles, density of vegetation and external inputs of nitrate-enriched surface water and groundwater. Global estimates of N_2O emissions from saltmarshes are in the range of -0.01 -0.87 Tg N_2O year^{-1} and for seagrass habitats range from 0.0 to 0.77 Tg N_2O year^{-1} (Murray et al., 2015).

5.4 Physical and biological regulation in seagrass ecosystems

Seagrass ecosystems are important global carbon sinks due to their capacity to uptake inorganic CO_2 through photosynthesis and primary production (carbon sequestration) and to store it as organic carbon both in living tissues (above- and below-ground) and also as detrital organic carbon (non-living organic matter) in sediments. In fact, the largest seagrass organic carbon pools are found in the sediments as non-living organic material, which represents between 50 to 90 % (Duarte et al., 2005) of the total organic carbon stock. Worldwide seagrass sediments have an average organic carbon content of 2.5% and annual carbon sequestration rates of 138 g organic C m^{-2} yr^{-1} (Fourqurean et al., 2012).

A number of contributing factors allow seagrass systems to act as valuable global carbon sinks, including; (i) their high primary productivity and biomass production; (ii) their capacity to slow down water movements, thereby enhancing sedimentation and burial rates; (iii) the fact that the biomass of some seagrass species is recalcitrant and contains high levels of lignin favouring low organic carbon decomposition rates; and (vi) the suboxic or anoxic nature of their sediments which reduces the remineralization rate of organic matter and favours its long-term storage (Duarte et al., 2013; Fourqurean et al., 2012; Mazarrasa et al., 2018). Seagrass carbon cycles (carbon sequestration and retention) are modulated by diverse environmental, geomorphological and biological factors such as climatic conditions (e.g., temperature, acidification, irradiance), hydrodynamic conditions (e.g., wave exposure, depth), sediment characteristics (e.g., grain size, sedimentation rates), water quality (e.g. nitrogen and phosphorus pollution) and species-specific seagrass biological characteristics (e.g., shoot density, ecological strategies) (Duarte et al., 2005; Mazarrasa et al., 2018). In addition, seagrass ecosystems may act as donors of carbon, as unattached seagrass leaves are partially exported from littoral zones to adjacent ecosystems such as terrestrial systems or deep canyons. This organic carbon can be assimilated by other organisms, therefore having a relevant role as a blue carbon service (Francois et al., 2018).

5.5 Calcium carbonate cycle in seagrass ecosystems

Calcium carbonate ($CaCO_3$) sediments include two main mineral forms, calcite and aragonite, mostly produced by biogenic precipitation (Smith, 2013). In seagrass meadows, the calcium carbonate ($CaCO_3$) associated with marine calcifying organisms present as epiphytes and benthic invertebrates on leaves or within sediments (Gacia et al., 2003) represents a major particulate inorganic carbon (PIC) fraction in seagrass beds, and plays a highly relevant role in the seagrass carbon cycle. Recent studies reported high rates of carbonate accumulation in the seagrass systems (Mazarrasa et al., 2015). PIC stocks of seagrass ecosystems depend on diverse factors such as species' growth strategies and functional roles, with larger and slower-growing seagrass species accumulating more PIC than smaller and faster-growing species, as large genera can sustain a higher amount of calcareous epiphytes. Additionally, differences in geomorphology, water composition, surrounding species (e.g. presence of corals), water depth, irradiance or water pH may affect PIC in seagrass sediments (Gacia et al., 2003; Hendriks et al., 2014). Despite the fact that these high rates of carbonate accumulation imply CO_2 emissions from precipitation, seagrass meadows are still strong CO_2 sinks worldwide (Mazarrasa et al., 2015).

6 STATE OF SCIENCE IN POTENTIAL BLUE CARBON ECOSYSTEMS

6.1 Macroalgae

To date, blue carbon strategies have largely focused on angiosperm-dominated coastal habitats. However there is increasing recognition of the potential role that marine macroalgae (seaweeds), both wild and cultivated, can play in climate change mitigation and adaptation (Krause-Jensen & Duarte, 2016; Duarte et al., 2017; Krause-Jensen et al., 2018; Lovelock & Duarte, 2019). Their current exclusion in blue carbon markets is largely due the fact that they colonise hard substrates and they lack roots which limits carbon burial occurring *in situ*, unlike plants in coastal wetland habitats which are deeply rooted in sediments and the majority of carbon accumulates within the habitat. There are three fundamental requirements for macroalgal beds to be considered a significant carbon donor to receiving systems: (1) high rates of carbon sequestration and biomass production; (2) effective transfer of biomass to receiver habitats; and (3) the donor carbon must undergo efficient burial within the receiver habitats; the latter in turn is a function of several factors, including: the intrinsic stability of the carbon, physical mechanisms of protection, and environmental factors that affect microbial activity/decay (Hill et al., 2015). When assessing the potential of macroalgae for carbon storage, key components thus include the longevity of the species (ranging from days to centuries) and fate of decaying algal biomass.

Macroalgae represent the largest global area of any vegetated coastal ecosystem (Duarte et al., 2017), with substantial biomass naturally occurring across both intertidal and subtidal habitats; however, specifically for Ireland, data on both distribution of individual species and communities, as well as biomass, are currently non-existent. Recent studies have shown that some macroalgae can contribute substantially to carbon sequestration at distant locations through organic carbon exports to the open ocean (Figure 4, Krause-Jensen & Duarte (2016) and references therein). It is estimated that 0.4% of global macroalgal net primary production is sequestered *in situ*, 44 % is exported as POC (particulate organic carbon) and DOC (dissolved organic carbon) and 11% is sequestered in deep sea sediments (Figure 5). It is important to note however that these estimates are derived from just 11 studies. When scaled to the global distribution of macroalgae it is possible that 173 Tg C year^{-1} may be stored long-term. Such estimates are comparable to the carbon sequestered by all other blue carbon habitats combined (Duarte et al., 2013). The magnitude of this carbon sequestration potential may therefore be of significance, with fate of the sequestered carbon identified as critical.

Macroalgae are an extremely diverse group of organisms that span three phyla; Rhodophyta, Ochrophyta (Phaeophyceae) and Chlorophyta. This diversity comprises a wide variety of morphologies and sizes, and life and carbon accumulation strategies that carry functional consequences affecting the fate of macroalgal carbon (Krause-Jensen et al., 2018; Ní Longphuirt et al., 2013). As an example, short-lived green *Ulva* spp. are less likely to have any impact on broader budgets than long-lived undisturbed intertidal *Ascophyllum* or subtidal maërl beds with individuals surviving for several hundreds of years (deduced from annual growth rates of <1mm/year; e.g. Bosence & Wilson 2003).

A recent study assessing the likely contributions of macroalgae to blue carbon sequestration highlighted that some macroalgae contain refractory compounds supporting long-term carbon storage, but that there is a much larger variation in tissue stability among macroalgal taxa relative to vascular plants, consistent with the larger diversity among macroalgae of cell wall structure and composition (Trevathan-Tackett et al., 2015).
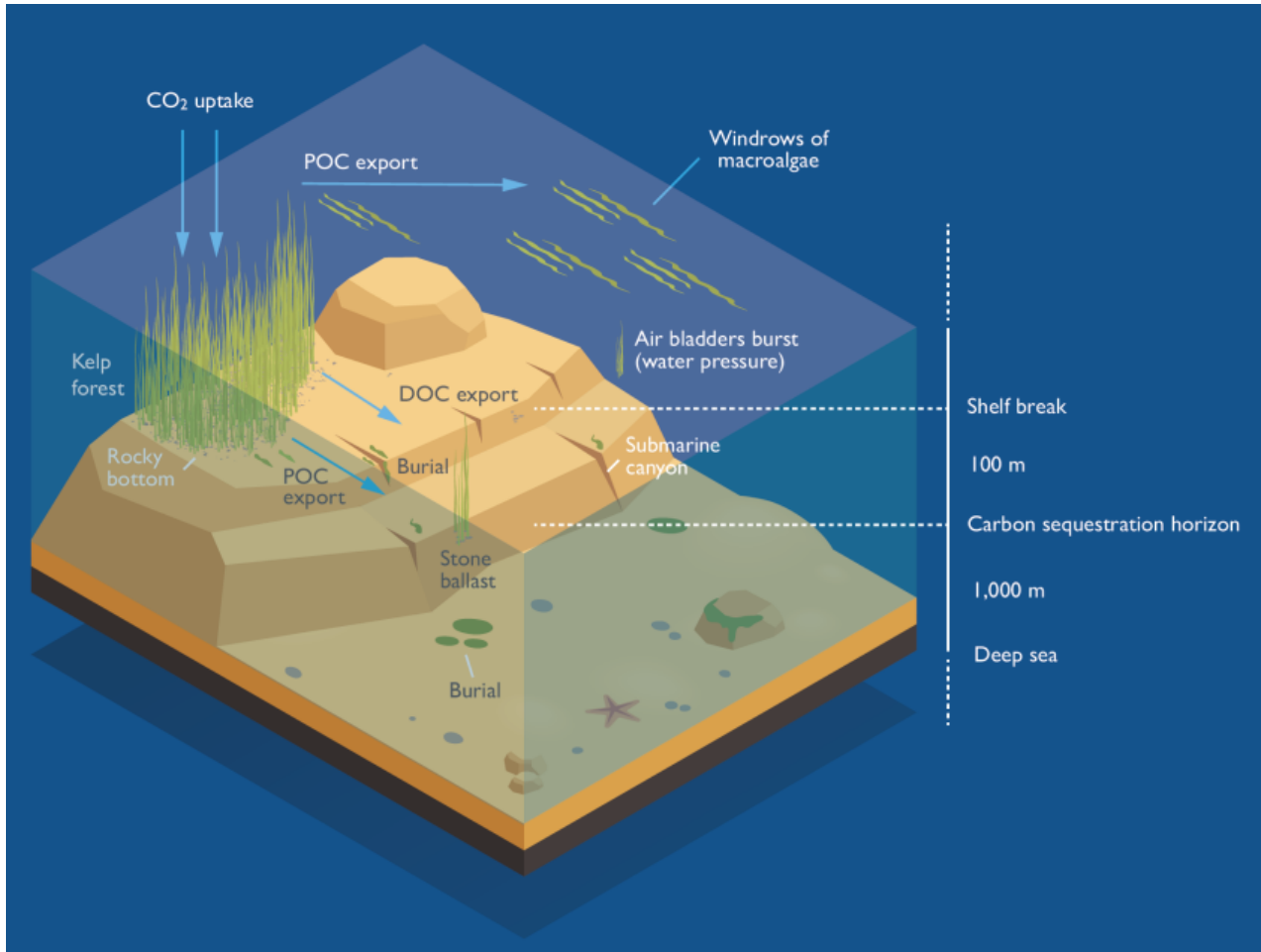

Figure 4.
Macroalgae as carbon donors - conceptual diagram of the pathways for export and sequestration of macroalgae-derived carbon. Adapted from Krause-Jensen & Duarte (2016).
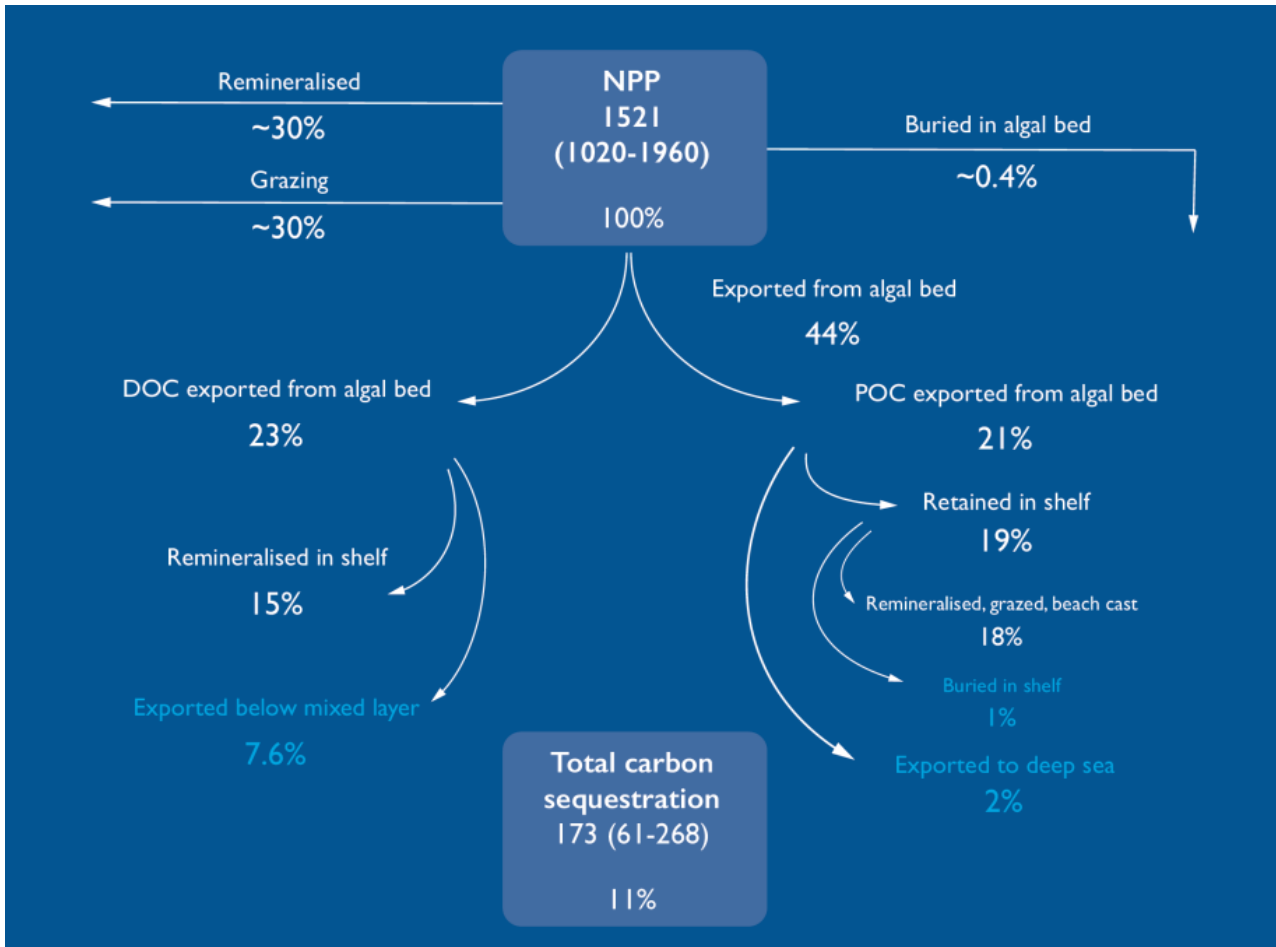
Figure 5.
Estimation of the percentage of NPP (net primary productivity in Tg C year^{-1}) that is processed through the ocean and ultimately sequestered. Adapted from Krause-Jensen & Duarte (2016), with data derived from a compilation of 11 studies.

In addition to extent, production and recalcitrance being key to carbon sequestration, an important factor is the transport of macroalgal carbon to areas where long-term storage is possible (Snelgrove et al., 2018) - this depends on species and conditions. Some large brown algae, including some species of the Fucales and Laminariales (though not the Laminariales species in Ireland), have buoyancy mechanisms such as pneumatocysts that allow the thalli to float and can facilitate long-range export. In principle, where this occurs and when algal materials degrade and sink to deep-sea sediments, algal carbon can be sequestered and stored long-term (Figure 4). Given the absence of floating devices in Irish kelp species, and the nature of Irish coastal environments that more likely are receivers of decaying biomass, the long-term deep-water or sediment sequestration of macroalgal carbon is possibly limited. The potential for long-term carbon sequestration and potential integration into blue carbon strategies is therefore most likely restricted to undisturbed beds of long-lived Fucales (namely *A. nodosum*) that are widely distributed in sheltered coastal areas in the west of Ireland and may contain refractory carbon.

For cultivated algae to be of value in a blue carbon context, the carbon budget (including full life-cycle analysis) involved in their production and downstream processing needs to be accounted for. Recent data suggest that 130 ha (1.3 km²) of Irish coastlines are licenced for seaweed cultivation, with a potential biomass of 6 tonnes of wet weight per hectare (Millard, 2019). The stated aim of these enterprises is to produce predominantly kelps (*Saccharina latissima* and *Alaria esculenta*) immersed in the water (and thus assimilating carbon) for periods shorter than 6 months per crop. Currently only a fraction of this areal potential is realised, and in comparison with Ireland's extensive natural seaweed beds, management of seaweed cultivation is unlikely to be a significant carbon mitigation opportunity in the near future.

On the other hand, previous research has demonstrated the potential for a locally significant role of the extensive natural macroalgal beds in regulating marine and coastal cycles of the halogen element iodine (Huang et al 2013; Nitschke et al. 2015) and their detectable contribution to regional atmospheric chemistry (Huang et al. 2010). This implies that significant and large-scale changes in algal biomass (whether by removal harvesting or via increased halocarbon emission from large-scale cultivation) have the potential to induce measurable, regional changes in atmospheric composition. The potential contribution of halocarbon emitting, commercially cultivated seaweeds such as *Asparagposis* spp. but also some brown algae including kelps (Keng et al., 2020) may further counteract the blue carbon potential of such species. Thus, as with CH_4 and N_2O, the impacts of management on emissions of other undesirable greenhouse gases and other algae-derived volatile compounds of interest in a climate context (e.g.H_2S) should thus be quantified and incorporated into potential climate mitigation protocols.

Climate mitigation under policy frameworks (UNFCCC) requires stringent standards (guidance from the Intergovernmental Panel on Climate Change / IPCC) to account for emissions and removals and the integration of macroalgae into these strategies is still some years away. However, for now a pathway of integration lies in climate change adaptation benefits, also incorporating coastal protection and environmental co-benefits such as alleviation of eutrophication, hypoxia and acidification (Lovelock & Duarte, 2019), though recommended measures need to be adapted to local species and environmental demands.

6.2 Maërl

Although classified as macroalgae, for the purpose of this report maërl is treated separately. Maërl is a collective term for several species of calcified red marine macroalgae within the Corrallinaceae, including *Phymatolithon calcareum, Lithothamnion glaciale*, *Lithothamnion corallioides* and *Lithophyllum dentatum* and *Lithophyllum fasciculatum* which live unattached on marine sediments at various depths. These species can form extensive beds, with typically 30% cover or more, mostly in coarse gravels or clean sands or muddy mixed sediments (Hall-Spencer et al., 2008). These beds are built up over millennia to create carbonate-rich deposits that form isolated habitats of high benthic biodiversity (Hall-Spencer, 1998; Grall & Hall-Spencer, 2003).

In the short term, calcifying coralline algae can act as a CO_2 sink via photosynthesis and calcium carbonate ($CaCO_3$) dissolution and act as a CO_2 source during respiration and $CaCO_3$ production. Indeed, $CaCO_3$ production results in the release of 0.6 mol of CO_2 per mol of $CaCO_3$ precipitated (Macreadie et al., 2017). Calcium carbonate cycling makes an important contribution to global carbon budgets over geological timescales, yet the magnitude and direction of its influence on CO_2 sinks within blue carbon habitats is unclear and currently unaccounted for (Macreadie et al., 2017). Van der Heijden and Kamenos (2015) have estimated total potential global carbon sink of maërl to be in the region of 1.6 × 109 Mg C per year, which would be comparable to saltmarsh and seagrass habitats. However, large uncertainties remain, including understanding the fate of the CO_2 produced by calcification. In addition, there is little knowledge on the dissolution rates of calcium carbonate in maërl beds, and the extent to which the calcium carbonate pool is made up of geogenic calcium carbonate (fossil) versus biogenic calcium carbonate (recent) (Zamanian et al.,

2016).

It is likely that the net effect of carbonate production and dissolution in both maërl and seagrass habitats does not alter the general interpretation that they are net sinks of atmospheric CO_2. However, as stated by Macreadie et al. (2017), if the balance between calcium carbonate production and dissolution is a source of CO_2, it should be subtracted from organic matter sequestration and the net carbon sequestration evaluated and accounted for.

6.3 Phytoplankton

Phytoplankton comprise diverse algal communities of largely single-celled, mostly autotrophic organisms present in the water column, most of which take up carbon through photosynthesis and release carbon through respiration, playing a significant role in the ocean's chemical cycles. The global total biomass of phytoplankton is estimated to be between 0.5 and 2.4 billion Mg C and they fix up to 35^{-50} Gt C yr^{-1}, representing a significant component of the natural carbon cycle (Buitenhuis et al., 2013). The majority of phytoplankton are consumed by higher-trophic level organisms where some of the carbon becomes integrated into marine fauna biomass. A small but important percentage (0.1% or 0.5-2.4 million Mg C yr^{-1}) of phytoplankton biomass is estimated to be sequestered long-term in seafloor sediments (Falkowski, 2012). Despite representing a globally relevant carbon sink, phytoplankton are considered less well suited for climate mitigation policies due to gradients in productivity, lack of practical accounting measures and issues with cross-jurisdiction (Howard et al., 2017).

In addition, the only current management strategy to increase phytoplankton productivity above the baseline involves artificially increasing nutrients (iron, nitrogen, phosphorus) in large expanses of the ocean. However, concerns have been expressed regarding the potential negative effects of such geoengineering projects on ocean ecosystems (Schiermeier, 2007; Howard et al., 2017).

6.4 Cold-water coral

Cold-water corals (CWCs) are sessile, filter-feeding, Cnidarians (Roberts et al., 2006) which most commonly occur in oceanic water temperatures between 4 °C and 12 °C (Freiwald et al., 2004). They are found on the continental shelf, and also in deep-sea areas with topographic elevations, such as seamounts. Species such as Madrepora oculata and Lophelia pertusa, build complex frameworks and carbonate mound habitats and are regarded as deep-sea biodiversity hotspots (Boolukos et al., 2019). Cold-water corals function in a similar way to maërl in that they act as a CO_2 sink via photosynthesis and calcium carbonate ($CaCO_3$) dissolution, as a CO_2 source during respiration and $CaCO_3$ production and they have a slow growth rate (Howard et al., 2017). However, a lack of practical accounting measures, issues of jurisdiction on the high seas and solutions for their effective management remain a challenge for the inclusion of cold-water corals into climate policies.

6.5. Bivalve Reefs

The degree to which bivalve (namely oyster and mussel species) reefs act as a source or sink of carbon is highly nuanced and can be dependent on factors such as the location and age of the reef (Fodrie et al., 2017). Given their nature as primary consumers, bivalves are net generators of CO_2. The biocalcification of CO_2 in the shell is not enough to compensate for the release of CO_2 generated during the respiration of organic matter (Lejart et al., 2012; Hily et al., 2013; Filgueira et al., 2019). Conversely, through a process known as biodeposition, living reefs can enhance particulate sedimentation enabling the build-up of organic and inorganic carbon stores (Lee et al., 2020). The overarching argument, however, is that regardless of how reefs function as carbon sources and sinks, disturbance of bivalve reefs would result in increased CO_2 emissions. Therefore, reefs should be protected to avoid further releases of carbon into the atmosphere (Lindenbaum et al., 2009; Fodrie et al., 2017).

7 STATE OF KNOWLEDGE IN IRELAND

7.1 Saltmarshes

In Ireland saltmarsh habitats are classified according to their morphology and nature of substratum (Curtis & Sheehy Skeffington, 1998), with 5 basic types identified: estuary, bay, sandflat, lagoon and fringe. The estuary type occurs at the mouths of medium to large rivers, and is well represented in counties Dublin, Cork, Limerick and Clare. Bay-type saltmarshes form in sheltered bays where freshwater input is minimal and occur in areas on the west coast principally around Donegal, Clew Bay and Galway Bay. Both estuary and bay types form on silt and clay substrata. Sandflat saltmarshes typically form in association with dune systems and can develop as extensive seaward extensions of machair (coastal grassy plains) in the west of Ireland. The lagoon type, the rarest of the saltmarsh types, forms behind shingle or sand barriers and more rarely on peat. Peat saltmarshes, also referred to as ombrogenic Atlantic saltmarshes (Cott et al., 2012), overlie peats formed under freshwater conditions and are found fringing sheltered rocky bays on the west coast.

Although ubiquitously distributed along the coastline of Ireland (Figure 6), few saltmarshes are extensive in area. The total national area is approximately 100 km^2 (Mcowen et al., 2017). However, preliminary studies point towards extremely efficient carbon sinks on a per-area basis. A Science Foundation Ireland project on blue carbon in Irish saltmarshes currently underway at University College Dublin (UCD) (Cott, *unpubl. data*), is testing the hypothesis that despite the small area, Irish saltmarshes are hotspots of carbon storage due to a number of factors. Ireland's mild, wet climate produces one of the longest growing seasons in the world for grasslands. Therefore, it is likely that Irish coastal wetlands which are dominated

by grass, rush and sedge species, also have an exceptionally long growing season translating into high productivity rates and subsequent carbon sequestration (Cott et al., 2013). In comparison to raised and blanket bogs, coastal wetlands emit low levels of methane gas due to inhibition of methanogens by sulphate in seawater. Research has shown that the higher the salinity of tidal wetlands, the lower the methane emissions (Poffenbarger et al., 2011). Since the majority of saltmarshes in Ireland are inundated by full seawater (35 ppt salinity), it is likely they emit low amounts of methane gas, further highlighting their potential role as efficient carbon sinks. Finally, a significant proportion of Irish saltmarsh soils are peat in origin which contain high amounts of organic carbon enhancing the carbon stocks of Irish coastal wetlands.

Data from North Bull Island saltmarsh, Co. Dublin, reveal the average carbon stock of these wetland soils is 883 ± 225 Mg C ha^{-1} (Burke et al., *unpubl. data*) which is over three times the global average per hectare for saltmarsh soils (and 3 x the IPCC default emission factor (IPCC, 2013)). The carbon density values of North

Figure 6. Distribution of saltmarsh habitats in Ireland. Adapted from Cott (2021).

Bull Island saltmarsh are comparable to carbon densities of low-lying blanket bogs (Wellock et al., 2011). Including above and belowground vegetation, the total carbon stock of this wetland is 106,574 ± 729 Mg C. If this wetland were to be degraded through drainage, it would result in the release of 391,126 ± 2,675 Mg CO_2 into the atmosphere. This is the equivalent of burning 900,000 barrels of oil (US EPA). By protecting and conserving this wetland it is clear that the avoided emissions are climate-relevant and these ecosystems can play an important role in climate mitigation.

Using the average carbon stock (soils and vegetation) from North Bull Island (888 Mg C ha^{-1}) and scaling it up, it is estimated that Irish saltmarshes could store up to **8.8 Mt carbon**. This site is an estuarine-type saltmarsh comprised of mud substrates that have a mid-range organic matter content of Irish saltmarsh types (Cott et al., 2013). Using the North Bull Island data therefore generates a reasonable estimate. Using carbon sequestration estimates based on the literature (Chmura et al., 2003) approximately **21,000 tonnes** of carbon could be sequestered every year.

$$100 \text{ km}^2 \times 888 \text{ Mg C ha}^{-1} = \textbf{8.8 Mt carbon}.$$

Accurate datasets will be published in the coming years by the team at UCD working on blue carbon. They are currently carrying out a national carbon stock inventory of saltmarshes, examining rates of carbon sequestration and assessing the fluxes of GHGs including methane and N_2O. As part of this project a management framework for relevant government agencies will be drawn up to ensure these ecosystems can be managed effectively for carbon benefits.
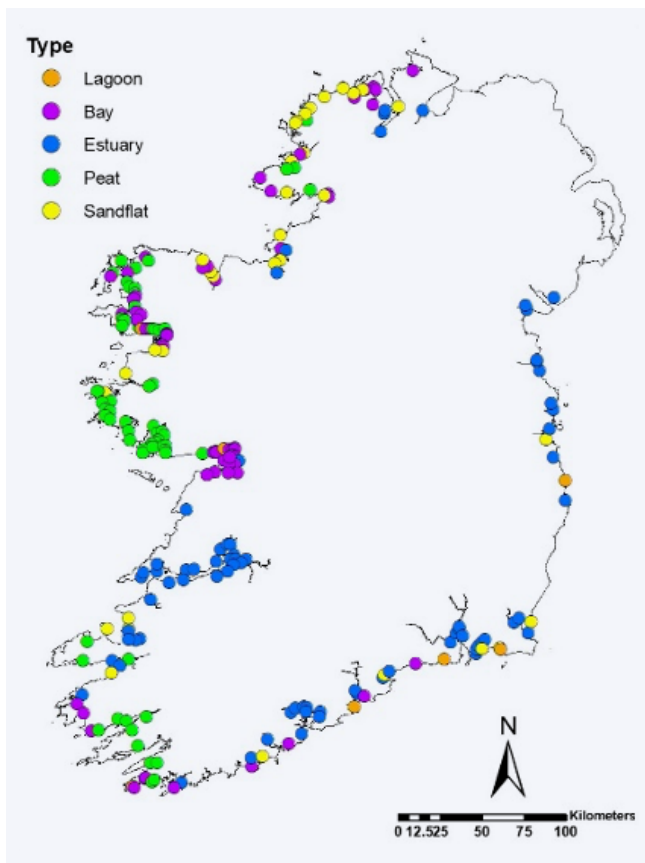
7.2 Seagrasses

Positioned in the middle of their latitudinal distribution (51°–55°N), *Z. marina* and *Z. noltii* are the most dominant seagrass species in Ireland that colonize soft-bottom sediments in intertidal areas to maximum depths of 10m (NPWS, 2014; MERC, 2005). Irish seagrass meadows are found all around Ireland but are particularly abundant along the west coast due to the presence of bays, estuaries, inlets and sheltered areas which create optimal environmental conditions; however, large extensions of intertidal meadows are also present on eastern coasts.

To date, the distribution along the coast of Ireland has been estimated as 45 km^2 (i.e. MERC, 2005; Wilkes et al., 2017), however, their distribution is insufficiently documented, owing to (i) a lack of previous research efforts; (ii) the logistical difficulty of mapping subtidal meadows; and (iii) the inaccessibility of remote sites. Recent studies mapped a significant wider expansion of 60-62 km^2 (Beca-Carretero et al., 2020; Beca-Carretero et al., unpubl. data), and it is estimated that seagrass ecosystems may cover an actual area of 165-300 km^2 (Beca-Carretero et al., 2020; Hastings et al., 2020).

Recent studies (Beca Carretero et al. 2019, Beca Carretero et al., *unpubl. data* and Villamayor et al. *unpubl. data*) reported that Irish *Z. marina* and *Z. noltii* meadows accumulate 207.5 ± 31 and 252.1 ± 75 kg km^{-2} of dry weight (DW) in the living below-ground tissue (rhizomes and roots) respectively, and 114 ± 37 and 73.7 ± 16 kg km^{-2} of dry weight (DW) in the above-ground biomass (leaves and sheaths). These values, extrapolated to the current seagrass mapped area in Ireland of 62 km^2, represent 20,797 kg DW in the living seagrass tissues (Wilkes et al., 2017; Beca-Carretero et al., 2019, 2020) (Table 3). These conservative estimates are based on living tissues (leaves, sheaths, rhizomes and roots) only, with additional carbon storage occurring within, this far, unquantified dead materials within sediments.
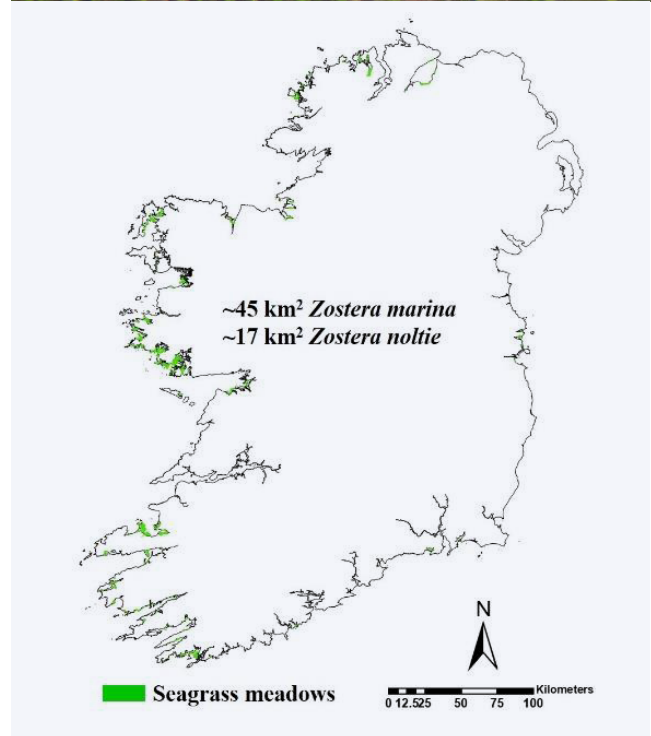

Figure 7. Map of the seagrass distribution (green color) along the coast of Ireland (Wilkes et al., 2017; Beca-Carretero et al., 2020; Beca Carretero et al., *unpubl. data*).

Based on estimates in the literature for the global average seagrass carbon density (108 Mg C ha^{-1}; Fourqurean et al., 2012) and a national extent of 62 km^2, the potential carbon stocks of Irish seagrass beds are estimated to be 0.6 Mt carbon:

62 km^2 x 108 Mg C ha^{-1} = **0.6 Mt carbon**.

7.3 Coastal reed swamps

Swamps are stands of emergent herbaceous vegetation that generally occupy a zone at the transition from open water to terrestrial habitats and can occur at the heads of estuaries and adjacent coastal floodplains (Fossitt, 2000). They can be associated with freshwater or brackish systems and experience tidal fluctuations. Most reed swamps are overwhelmingly dominated by *Phragmites australis* and soils can be high in organic matter (Cott, *unpubl. data*). At present there are no data on the extent of coastal reed swamps in Ireland which hampers our ability to estimate the carbon stock potential of these habitats. In addition, it may be that due to lower soil salinity levels they have the potential for higher methane emissions than saltmarsh habitats.

7.4 Key macroalgae in Ireland

Of the several large macroalgal species that dominate Irish coasts, based on their abundance and likely biomass, the macroalgae listed below may stand out from a blue carbon perspective. However at this point, no data on distribution, biomass or blue carbon potential are available for any species. Building upon recent pilot-scale research undertaken in association with INFOMAR (Rossiter et al., 2020), which mapped intertidal seaweed beds in western Ireland, it is anticipated that upscaled methodologies will ultimately provide more accurate data on relevant habitats and associated species in around Irish coasts. Such methodologies, and derived data, will be critical in order to assess the carbon stored in the living biomass and estimates of carbon sequestered over time in Ireland's marine and coastal environments.

Intertidal *Ascophyllum nodosum* beds

The brown macroalga *Ascophyllum nodosum* (Fucales) is a key foundation species as it plays a strong role in structuring coastal communities (Schmidt et al., 2011) that occur along sheltered intertidal rocky shores of North Atlantic coasts. *A. nodosum* represents the main biomass of Irish intertidal seaweeds, though commonly mixed with other fucoids, and is particularly abundant on the west coast given the high proportion of sheltered intertidal rocky shores. It is a long-lived species with unbroken individual fronds commonly persisting for over a decade (e.g. Stengel & Dring, 1997); basal clumps (holdfasts) from which shoots arise likely survive multiple decades, making these systems amongst the most stable, and based on extent and biomass, productive in Irish waters.

At present, the absence of data on extent and biomass of this species hamper our ability to estimate its potential contribution to carbon sequestration. Recent data suggest carbon acquisition rates of 5.25 ± 1.27 µmol C g^{-1} DW h^{-1}; these likely increase with increasing CO_2 concentrations and are (as with all primary producers) significantly temperature dependent (Ní Longphuirt et al., 2013). Although these rates are lower than those of other intertidal algae, the potential of this species is associated with its extensive (assumed) distribution, biomass, longevity and resistance to grazing, and thus anticipated low carbon exports compared to shorter-lived species.

Sargassum muticum

The invasive Pacific brown macroalga *Sargassum muticum* has successfully invaded lower intertidal to subtidal regions of Irish coastal areas since it was first recorded in 2001 (Loughnane & Stengel, 2002). Seasonally it exhibits much higher growth rates than *A. nodosum* (Hall-Spencer et al., 2008) which are likely to be exacerbated by global change (Ní Longphuirt et al., 2013). As the species is only perennial in part, the annual bulk biomass produced is short-lived: fronds are shed in late summer to early autumn (Baer & Stengel, 2010), with large quantities of decaying biomass washed up ashore or re-entering the sea. Thus, despite its local and seasonal abundance in biomass, it is likely that this species has a very low blue carbon potential. Additionally, there are reports that it negatively impacts on highly biodiversity ecosystems with high blue carbon storage capacity, namely seagrass and maërl beds (Scally et al., 2020).

Kelps

There are five species of kelp native to Ireland; *Laminaria digitata, Laminaria hyperborea, Saccharina latissima, Alaria esculenta* and *Saccorhiza polyschides*. They differ in various aspects, such as habitat preference, morphology, ecophysiology and longevity, and display distinct life strategies and distribution patterns (Kelly, 2005). *Saccorhiza polyschides* is an opportunistic fast-growing species and although it can reach a significant size of up to 3 metres in length (Norton & Burrows, 1969) it is an annual species that dies off in winter. *Laminaria digitata* and *L. hyperborea* are the only Irish kelp species that currently form extended monospecific beds. In less exposed areas of the coast, where the substratum is suitable, large, dense beds of *L. digitata* and *L. hyberborea* occur and are locally common. *Laminaria hyperborea* can form forests extending to a depth of 30 m in Ireland, whereas *L. digitata* and *Saccharina latissima* inhabit shallower sublittoral zones and some intertidal zones. The longest-lived species is *L. hyperborea* with individuals reaching an average age of 15 years, whereas other laminarian species of the upper sublittoral zone generally live approximately 3-4 years (Lüning, 1990).

Although Hession et al. (1998) predicted that up to 56% of the west coast of Ireland is occupied by kelp forests, these estimates are not verified, and where sand and mud are predominant on the substratum and in most estuarine regions, kelps are scarce or absent. Additionally, such areal estimates do not take into account habitat preferences or influences of hydrodynamics on recruitment and biomass accumulation. In highly wave-exposed locations, kelp biomass and that of associated species is generally reduced (Edwards, 1980), and global warming further likely limits both kelp extent and productivity (Smale et al., 2013).

The current data gap prevents scale-up estimates of carbon stocks and subsequent carbon sequestration. Recent and current mapping initiatives by INFOMAR provide data on potentially suitable habitats for some shore sections but do not provide data on actual kelp distribution nor biomass, nor associated carbon fluxes within and across these. Based on pilot estimates from Strangford Lough and Galway Bay, Kelly (2005) proposed primary production in Irish kelp forests at 843-4,800 g C m^{-2} per year. However, the majority of this carbon (82%, Krumhansl & Scheibling, 2012; Krause-Jensen & Duarte, 2016) is likely exported as detritus each year as the blades, or entire thalli for short-lived species, decay locally or are incorporated into intertidal food chains once washed ashore.

7.5 Maërl

Maërl beds in Ireland are highly biodiverse and species-rich habitats but are predominantly characterised by the rhodophyte *Lithophyllum dentatum* (De Grave et al., 2000). They are widely distributed in the mid-west and the south-west (Hall-Spencer et al., 2008), with the majority of the maërl beds located between 0 to 20 m depth with a vertical extent of 0.1-3 m (De Grave & Whitaker, 1999). The extent of sediments bearing maërl in Ireland is estimated at 60 km^2, however the entire coastline has not been surveyed (Hall-Spencer, 2008). Based on figures by De Grave et al. (2000) and Burrows et al. (2014) - the Scottish blue carbon report, we can estimate that the quantity of carbon in Irish maërl deposits is **1,143,120 t carbon**. However, modern mapping methods should verify the actual extent of maërl habitat in relation to depth which, in turn, will influence net productivity. Bed thickness and carbon storage potential should further be quantified as Scottish conversion rates are not necessarily transferable to Irish waters.

Estimated maërl bearing area: 57 x 10^6 m^2 (minimum area -De Grave et al., 2000)
Estimated average thickness of maërl: 2 m (range: 0.1–3m) (De Grave et al., 2000)
Estimate volume of maërl bearing sediment: 114 x 10^6 m^3 (De Grave et al., 2000)
Of which 10% is maërl: 11 x 10^6 m^3 (De Grave et al., 2000)
Weight of maërl: 866.7 kg m^{-3} (Burrows et al., 2014)
Carbon content of maërl: 12% (Burrows et al., 2014)
Carbon stock estimates = maërl volume x maërl weight x 12%
Estimated carbon in Irish maërl deposits: 1.1 Mt carbon

7.6 Phytoplankton

Ireland's territorial waters and Exclusive Economic Zone (EEZ) extends out across the North Atlantic Ocean, includes parts of the Irish and Celtic Seas and covers an area of 880,000 km^2 (Marine Institute, 2020). Given the extent of Ireland's marine environment, the phytoplankton resource is likely to be significant in terms of a carbon sink. Based on conservative estimates of phytoplankton production of 81 g C m^{-2}·year^{-1} (Burrows et al., 2014), which takes into account consumption of phytoplankton by pelagic micro-heterotrophs, net productivity in Irish waters is estimated at 71.3 Mt C year^{-1}:

880,000km^2 x 81 g C m^{-2}·year^{-1} **= 71.3 Mt C year^{-1}**

However, there is a gradient in phytoplankton productivity between coastal and offshore, with higher values typically recorded in coastal and estuarine environments due to increased levels of nutrients. Cloern et al. (2014) estimated an average of 252 g C m^{-2} yr^{-1} for coastal waters. In addition, studies have shown that there is a high level of natural variability associated with phytoplankton assemblages in Irish waters with phytoplankton biomass largely affected by the stabilization and de-stabilization of the water column and horizontal transport processes (O'Boyle & Silke, 2010). This is, therefore, a conservative estimate.

The proportion of carbon that is removed from surface waters to deep ocean and sediments is estimated to be 10% of annual net productivity (Burrows et al., 2014; Lee et al., 2002):

Carbon sequestered to deep waters/sediments **= 7.13 Mt C year^{-1}**

It is important to note that this estimate is based on complex models simulating production and sedimentation in the North Sea (Lee *et al.,* 2002) and its application to Irish waters is therefore associated with a high degree of uncertainty. Exports to deep waters may not remain in a steady state as organic and inorganic carbon can be brought back to the surface in upwelling regions. Therefore, although a potentially significant sink of carbon, inclusion of phytoplankton in climate policies is challenging due to gradients in productivity and high seasonal variation in conjunction with a lack of practical accounting measures.

7.7 Cold-water coral

Ireland has extensive reefs of cold-water corals in its North East Atlantic waters, but as of yet no estimate of total area. It is likely to be a significant carbon store but, as of now, there is no formal pathway for the inclusion of cold-water corals into climate mitigation policy frameworks. However, their conservation and protection is essential to maintaining healthy biodiverse ecosystems.

7.8 Bivalve reefs

Reef forming species such as the native oyster *Ostrea edulis* and blue mussel *Mytilus edulis* are widespread around the coast of Ireland (Tully & Clarke, 2012; Lynch et al., 2014). However, accurate data on extent are lacking and this hampers our ability to estimate the carbon stock of bivalve reefs, in addition to potential local differences in the degree to which reefs act as a source or sink of carbon. Oyster populations were at their highest in Ireland in the 19th century and have declined considerably since then (Went, 1962). Six main areas known for oyster production were surveyed between 2010 - 2012 and the total extent was estimated at 17 km^2, with highest densities reported for Tralee Bay (Tully & Clarke, 2012). Indeed, Tralee bay hosts a

considerable amount of saltmarsh and seagrass habitat. Evidence suggests that proximity to depositional habitats such as saltmarsh and seagrass facilitates carbon storage in oyster reefs (Fodrie et al., 2017). Furthermore, restoration of oyster reefs can have co-benefits for increases in saltmarsh extent (Fodrie et al., 2017). This coupling of carbon storage services is particularly important in a restoration context for site selection to optimise reef services.

7.9 Summary of estimated contribution

Table 4 summarises the carbon stocks and sequestration rates of all Irish blue carbon and potential blue carbon ecosystems. Given the extensive national marine territory, phytoplankton have the highest carbon sequestration potential. As indicated above there is a high degree of uncertainty with this estimate but it is likely that phytoplankton productivity could be higher in coastal areas than estimated here which would lead to higher sequestration rates. However, on a per-area basis these estimates indicate that saltmarshes are 25 times more productive than phytoplankton.

Table 4.

Estimates of extent, carbon stocks, carbon sequestration rates and avoided emissions in Ireland's blue carbon ecosystems (BCEs); saltmarsh and seagrasses and potential blue carbon (pBCES). Data for seagrasses are conservative estimates. Estimates for maërl and phytoplankton have a high degree of uncertainty.

	Habitat	Extent	Carbon Standing Stock	Carbon sequestration rates	Avoided Emissions if Conserved
		km^2	Mt C	Mt C year^{-1}	Mt CO$_2$
BCE	Saltmarsh	100*	8.8[a]	0.02[d]	32.3
BCE	Seagrass	›62[†]	0.6[b]	0.01[e]	2.2
BCE	Coastal reed swamp	-	-	-	-
pBCE	Macroalgae	NA	NA	-	-
pBCE	Maërl	57[‡]	1.1[c]	-	4
pBCE	Cold-water corals	-	-	-	-
pBCE	Phytoplankton	880,000	-	7.1[f]	-
pBCE	Bivalve Reefs	-	-	-	-

*Mcowen et al. (2016)

[†] Wilkes et al. (2017); Beca-Carretero et al. (2020)

[‡] estimated area of maërl bearing sediment

[a]calculated using data from North Bull Island saltmarsh, Co. Dublin, which represents a substratum type of mid-range organic matter content. This is assumed to generate a reasonable estimate.

[b]calculated based on Fourqurean et al. (2012)

[c]calculated based on Burrows et al. (2014) and may not be transferable to Irish waters

[d]calculated based on Mcleod et al. (2011)

[e]calculated based on Mcleod et al. (2011)

[a]calculated based on Burrows et al. (2014) and Lee et al. (2002), this estimate has the highest degree of uncertainty

8 THREATS TO BLUE CARBON ECOSYSTEMS

8.1 Current threats

Any agent that degrades a blue carbon ecosystem has the capacity to reduce its ability to store carbon. A fully functioning healthy ecosystem is one in which can optimally store carbon. When a habitat becomes degraded its capacity to act as a carbon sink is reduced or lost and it can quickly become a source of CO_2 emissions to the atmosphere (see Figure 1). The scale of this source is dependent on the extent of degradation. Current global annual loss rate is at 1–2% for tidal marshes and 1.5 – 2.5% for seagrass beds (Waycott et al., 2009; Pendleton et al., 2012). It is estimated that the amount of CO_2 released annually from degraded or lost wetlands is equivalent to the annual emissions of the United Kingdom (Pendleton et al., 2012). Losing these systems also means high risks of severe flooding and coastal erosion, thus increasing the vulnerability of millions of people living along the world's coastline and adding to eutrophication (Narayan et al., 2016).

In Ireland, the conservation status of coastal wetlands and marine ecosystems is based on Annex I of the European Habitats Directive (92/43/EEC). Saltmarshes are divided into categories based on their vegetation associations, whereas Annex I marine habitats correspond directly to geographical features (e.g. *Large shallow inlets and bays, Estuaries, Mudflats and sandflats not covered by seawater at low tide*) as opposed to communities of species. In addition to the Habitats Directive, saltmarshes and seagrass beds have been used as a Biological Quality Element (BQE) under the Water Framework Directive (WFD) (Foden & Brazier, 2007).

The recent habitat assessment report of Irish saltmarshes produced by the National Parks and Wildlife Service (NPWS) found that the overall conservation status of the two main saltmarsh plant community types, termed Atlantic and Mediterranean saltmeadows, was assessed as unfavourable/inadequate (Brophy et al., 2019). Area loss was recorded for Atlantic saltmeadows mostly in the form infilling/reclamation for a range of uses, while the structure and function of this habitat was primarily negatively impacted by livestock grazing, most commonly cattle - an impact the report noted was likely to continue into the future. Habitat loss was also recorded for Mediterranean saltmeadows attributable mainly to infilling and reclamation. Some 87% of the Annex I saltmarsh habitat mapped by the NPWS was located in candidate Special Areas of Conservation (cSAC). Within these cSACs, however, much of the land is in private ownership, which means that despite the designation, damage by infilling, overgrazing, and reclamation is still occurring (McCorry & Ryle, 2009).

In its marine habitat assessment report the NPWS assessed the conservation status of Estuaries and *Mudflats and sandflats not covered by sea water at low tide* as unfavourable/inadequate and that of *Large shallow inlets and bays* was assessed as unfavourable/bad (Scally et al., 2020). The latter represents a decline in status from the previous reporting period (2009). It is considered likely that agriculture, in combination with changing land use, urbanisation, and discharge of inadequately treated waste-water played a significant role in the negative assessment for estuary sites. Similarly, the major pressures and threats on the *Large shallow inlets and bays* have been identified as agriculture, forestry, aquaculture, fisheries and waste-water treatment and disposal. The main explanation for the failure of this habitat to achieve favourable conservation status is the significant change recorded in the area and structure and function of keystone communities *Zostera marina, Z. noltii* and maërl. In addition, significant increases in the

invasive alien species *Sargassum muticum* have been found to impact *Z. marina* communities and have the potential to lead to gross habitat change and significant ecosystem-wide impacts in the future (Scally et al., 2020). Its establishment in *Zostera* beds leads to light attenuation with consequent impacts on growth rates and the ability of eelgrass beds to regenerate. In specific sites such as Roaringwater Bay SAC, aquaculture has been identified as the cause of direct smothering of vulnerable maërl species.

In summary, it is evident that despite EU protection under the Habitats Directive, coastal wetlands and marine habitats in Ireland are being negatively impacted with regard to extent and structure, and function. This ultimately affects the capacity of these systems to sequester and store carbon, and could result in the emissions of considerable amounts of CO_2 into the atmosphere.

8.2 Future threats

Global warming, sea-level rise and ocean acidification

In tidal marshes, plant productivity is likely to increase with warming and concomitantly rates of decomposition are likely to increase. Models suggest that the net effect of warming on tidal marsh carbon sequestration is likely to be small (Megonigal et al., 2019). Warming may eventually exceed the adaptive thermal capacity of seagrass species worldwide causing the migration or disappearance of their ecological niche. However, temperature rises may eventually favour seagrass growth and habitat expansion of species currently living under suboptimal temperatures. A recent study demonstrated that for Irish seagrass populations living at current maximum summer seawater temperatures of 15-17 °C, a 2–3 °C increase projected for the end of this century (IPCC, 2014) may favour growth and production and significantly (8-12%) increase capacity to accumulate organic carbon in their rhizomes (Beca-Carretero et al. *unpubl. data*).

The effect of sea-level rise on tidal marshes is dependent on the rate of increase. At modest rates, sea-level rise can enhance carbon sequestration through organic matter burial in anaerobic soils, but at very high rates it can cause plants to grow slowly and eventually die, effectively turning the marsh over to open water (Kirwan & Megonigal, 2013). Irradiance is the most relevant environmental parameter affecting photosynthetic activity and productivity and therefore, vertical distribution of seagrasses (Ralph et al., 2007). Predicted increases in sea level rise are thus expected to negatively affect seagrass ecosystems by reducing the irradiance available to deep-adapted plants, exposing them to more extreme environmental conditions and physiological stress (Ondiviela et al., 2020).

Ocean acidification, the reduction in pH as a result of increases in concentrations of dissolved CO_2 absorbed from the atmosphere, poses a threat to marine ecosystems and in particular to calcifying organism such as maërl and cold-water corals. Responses in these organisms are species-specific but include reduced calcification, reduced rates of repair and weakened calcified structures. It is predicted that regions in the North Atlantic will be exposed to multiple stressors by 2100, experiencing at least one critical change in water temperature (+2°C), aragonite saturation horizon (shoaling above 1000 m) and/or reduction in dissolved oxygen (>5%) (Puerta et al., 2020).

9 STATE OF POLICY

Text Passages from the Paris Agreement Relevant for Blue Carbon

Preamble
"Recognising the importance of the conservation and enhancement, as appropriate, of sinks and reservoirs of the greenhouse gases referred to in the Convention"

Article 5.
"Parties should take action to conserve and enhance, as appropriate, sinks and reservoirs of greenhouse gases as referred to in Art. 4.1(d) of the Convention"

Article 13. Each party shall regularly provide the following information:

a) *"A national inventory report of anthropogenic emissions by sources and removals by sinks of greenhouse gases, prepared using good practice methodologies accepted by the Intergovernmental Panel on Climate Change and agreed upon by the Conference of the Parties serving at the meeting of the Parties to the Paris Agreement"*

All Parties to the United Nations Framework Convention on Climate Change (UNFCCC) are required to submit National Reports on the implementation of the convention, which include information on emissions by sources and removals by sinks of GHGs. In 2013, the IPCC, with expert input from IUCN and the Blue Carbon Initiative, published a landmark 2013 Supplement to the *2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands (2013 Wetland Supplement).* This Wetland Supplement provides national-level inventory methodological guidance on wetlands, including a chapter on coastal wetlands with default emissions factor values. The guidance covers the estimation of carbon stock (and changes in stock) in drained land, rewetted organic soils, coastal wetlands and constructed wetlands for wastewater treatment. This Supplement provides the necessary standard methodology to incorporate coastal wetlands into climate mitigation (Herr et al., 2019). To date, as few as 16 countries have incorporated coastal wetlands into their National GHG Inventory Report, and USA and Australia are two of these countries.

As of yet, there is no methodology to enable the inclusion of marine carbon/potential blue carbon ecosystems into National GHG Inventory reporting. Inclusion into GHG inventory requires stringent methodologies and the existing science is currently not adequate to allow for this. However, there are opportunities within the Paris Agreement and Nationally Determined Contributions.

Paris Agreement and Nationally Determined Contributions (NDCs)

In December 2015, the Paris Agreement was adopted by all 196 Parties to the UNFCCC at COP21. One of the important outcomes of the Paris Agreement is that countries can independently determine how to lower their emissions, which they outline in pledges called NDCs. Every five years, Parties are asked to submit a revised NDC (Art 4.9 of the Paris Agreement) that is more ambitious than the previous one, signifying a progression in actions. Parties can develop their NDC actions and priorities based on a portfolio of measures including the conservation and restoration of nature as a climate change solution. The recognition of the roles that natural ecosystems can play in climate change mitigation and adaptation are often referred to as nature-based solutions. Thus far, 28 countries include a reference to coastal wetlands in terms of mitigation in their NDCs and 59 countries include coastal ecosystems and the coastal zone into their adaptation strategies. Ireland is not one of these countries.

Most NDCs still fall short of effective mitigation and adaptation policy programmes. The diversity of the commitments made by the various countries means that the question of coherent organisation of policies capable of implementing these commitments is still unresolved (Herr & Landis, 2016).

Table 5.

Pathways for Ireland to integrate blue carbon and marine carbon habitats into UNFCCC policy. Mitigation frameworks require stringent methodologies, and the existing science is not adequate to allow for the inclusion of marine carbon/potential blue carbon ecosystems.

Habitat	Mitigation	Adaptation
Saltmarsh	Inclusion in National GHG Inventory (Use of IPCC Wetlands Supplement) Inclusion in NDCs	Inclusion in NDCs
Seagrass	Inclusion in National GHG Inventory (Use of IPCC Wetlands Supplement) Inclusion in NDC	Inclusion in NDCs
Coastal reed swamps	Inclusion in National GHG Inventory (Use of IPCC Wetlands Supplement) Inclusion in NDCs	Inclusion in NDCs
Macroalgae		Inclusion in NDCs
Maërl		Inclusion in NDCs
Phytoplankton		Inclusion in NDCs
Cold-water corals		Inclusion in NDCs
Bivalve Reefs		Inclusion in NDCs

In terms of inclusion in NDCs, countries have used a number of categories/action types for blue carbon efforts (Herr et al., 2019):

i) LULUCF – including coastal wetlands as part of LULUCF

ii) Conservation, Protection and Restoration of Habitats– including coastal wetland adaptation solutions along with other relevant marine habitats with reference to conservation management and protection

iii) Coastal Zone Management – including information and making specific reference to planning tools such as Integrated Coastal Zone Management

iv) Fisheries – including information and see the need to priorities adaptation in job generating sectors using coastal and marine resources

By integrating blue carbon habitats into UNFCCC policy and ensuring better conservation and protection, Ireland can progress towards one of its goals outlined in its Climate Action plan, which is to achieve 26.8 $MtCO_2$eq. abatement through LULUCF actions over the period 2021 to 2030, comprised of: better management of grasslands, tillage land and non-agricultural wetlands (1.4 $MtCO_2$eq. cumulative abatement) (Climate Action Plan, 2019).

Finance mechanisms for blue carbon projects in Ireland
Currently, to avail of climate finance mechanisms for carbon projects from voluntary markets (Voluntary Carbon Standard/Verra) there must be a standard methodology in place, much like integration into mitigation policy frameworks. Verra recently released a methodology for tidal wetland and seagrass restoration projects (VM0033). Implementation of this would require a large-scale project (extensive seagrass or saltmarsh restoration site) with relatively straightforward carbon flux pathways. In addition, initial transaction costs tend to be high. Therefore, at present, it is unlikely that Ireland can benefit from these finance mechanisms.

10 FUTURE RESEARCH PRIORITIES

Several future research priorities under the headings where there are critical knowledge gaps may be recommended. These should align with Ireland's Climate Action Plan (2019) which aims to "upgrade land-use and habitat mapping systems to establish the baseline condition of wetlands and inform the development of best-practice guidelines for wetland management".

• **Extent** - Increase accuracy of the extent of Irish blue carbon and potential blue carbon ecosystems. Accuracy of the extent of seagrass beds and saltmarshes should be improved and the extent of coastal reed swamps which are likely to have high soil organic content but simultaneously the potential for higher methane emissions. In addition, there are limited data on the extent of natural seaweed beds (including *A. nodosum* beds, kelps, maërl), and cold-water corals.

• **Productivity** - Produce accurate data on productivity of Irish blue carbon and potential blue carbon ecosystems.

• **Carbon fluxes** - Examine carbon storage capacity and fluxes for all habitats; develop methods to enable the capacity to trace carbon from donor to sink habitats; identify degraded sites (seagrass/saltmarsh) and quantify the carbon emissions and associated greenhouse gases from degradation; examine the fate of carbon from macroalgal beds and other potential blue carbon systems, and identify donor sites and sink locations.

• **Future threats** - Examine future threats to all blue carbon and potential blue carbon ecosystems; i.e. examine the stability of Irish saltmarshes in the face of sea-level rise; examine single and interactive responses of seagrasses, macroalgae, phytoplankton, maërl and cold-water corals to global warming, eutrophication and ocean acidification.

• **Management** - Based on more accurate data filling the knowledge gaps outlined above, effective management strategies need to be drawn up to ensure the conservation and preservation of the carbon sequestration capacity of Ireland's marine and coastal habitats. Management in potential blue carbon ecosystems is more challenging and scientists and stake-holders will need to work together to ensure positive outcomes and alignment with Ireland's climate adaptation plan and the EU Green Deal. In addition, restoration of previously degraded saltmarshes and seagrass beds should take place.

These research priorities map directly onto the Marine Institute's strategic initiative of "Deepening Our Knowledge - developing an integrated, multidisciplinary understanding of the structure, functioning and dynamics of ocean ecosystems, coupled with the goods and services they provide to society" (Marine Institute Strategy, Building Ocean Knowledge, Delivering Ocean Services, 2018-2022).

11 POTENTIAL RESEARCH INITIATIVES

The EU Green Deal has made reference to the importance of nature-based solutions including healthy and resilient seas and oceans (EU Green Deal, 2020). Furthermore, the Commission will consider drafting a nature restoration plan which may provide funding mechanisms to support Member States reaching this ambition – this may offer research potential for Ireland.

Irish scientists are currently collaborating on key blue carbon seagrass projects in Europe (e.g. Denmark, Germany, Spain), and the Marine Institute's INFOMAR Team could be key partners in macroalgal or other vegetation mapping initiatives in Ireland. Although research into blue carbon ecosystem responses to climate change (elevated CO_2 and temperature) has been ongoing in the USA for many years (e.g., Smithsonian Environmental Research Center, Global Change Research Wetland), it is only now becoming established in Europe; a long-term field site testing the effect of climate change on saltmarsh dynamics is run by the University of Hamburg (MERIT Marsh Ecosystem Response to Increased Temperature). Given that our current understanding of how climate change will affect carbon sequestration in European blue carbon habitats is mainly derived from a single site in the Wadden Sea, there is potential to establish a long-term saltmarsh/seagrass monitoring site in Ireland to test the effect of climate change on these habitats in the North Atlantic.

12 CONCLUSION

Ireland has two main blue carbon ecosystems; saltmarsh and seagrass beds - which despite their limited areal extent, contain climate-relevant pools of carbon. Conservation and appropriate management of these habitats will ensure substantial CO_2 emissions are avoided, and there is the possibility for inclusion in mitigation policy frameworks (National Inventory Report on GHGs to the UNFCCC). In addition, Ireland has a vast marine territory which contains other ecosystems that are an essential part of the biosphere's process of capturing, fixing and storing carbon. Macroalgae, maërl, phytoplankton, cold-water corals and bivalve reefs are classed as potential blue carbon ecosystems. Of these, phytoplankton has significant carbon sequestration potential but gradients in productivity and limitations in our understanding of carbon fluxes hampers the ability to accurately quantify this contribution. In addition, estimates of carbon stocks in macroalgae beds, cold water corals and bivalve reefs are hampered by a lack of data on extent and productivity. However, there is scope for Ireland to include potential blue carbon ecosystems in Nationally Determined Contributions (NDCs) as part of the Paris Agreement. Overall, the challenges outlined in this report represent an opportunity for scientists and policy makers to work together to optimise the climate potential of Ireland's coastal and marine carbon resources.

Acknowledgements

13 REFERENCES

Al-Haj, A.N. and Fulweiler, R.W., 2020. A synthesis of methane emissions from shallow vegetated coastal ecosystems. *Global Change Biology*, 26(5), 2988-3005.

Beca-Carretero, P., Varela, S. and Stengel, D.B., 2020. A novel method combining species distribution models, remote sensing, and field surveys for detecting and mapping subtidal seagrass meadows. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 30(6), 1098-1110.

Beca-Carretero, P., Stanschewski, C.S., Julia-Miralles, M., Sanchez-Gallego, A. and Stengel, D.B., 2019. Temporal and depth-associated changes in the structure, morphometry and production of near-pristine *Zostera marina meadows in western Ireland. Aquatic Botany*, 155, 5-17.

Baer, J. and Stengel, D.B., 2010. Variability in growth, development and reproduction of the non-native seaweed *Sargassum muticum* (Phaeophyceae) on the Irish west coast. *Estuarine, Coastal and Shelf Science*, 90(4), 185-194.

Bosence, D. and Wilson, J., 2003. Maerl growth, carbonate production rates and accumulation rates in the NE Atlantic. Aquatic Conservation: *Marine and Freshwater Ecosystems,* 13(S1), S21-S31.

Boolukos, C.M., Lim, A., O'Riordan, R.M. and Wheeler, A.J., 2019. Cold-water corals in decline–A temporal (4 year) species abundance and biodiversity appraisal of complete photomosaiced cold-water coral reef on the Irish Margin. Deep Sea Research Part I: *Oceanographic Research Papers,* 146, 44-54.

Brophy, J.T., Perrin, P.M., Penk, M.R., Devaney, F.M. and Leyden, K.J., 2019. Saltmarsh Monitoring Project 2017-2018. Irish Wildlife Manuals, No. 108. National Parks and Wildlife Service, Department of Culture, Heritage and the Gaeltacht, Ireland.

Buitenhuis, E.T., Vogt, M., Moriarty, R., Bednaršek, N., Doney, S.C., Leblanc, K., Quéré, C.L., Luo, Y.W., O'Brien, C., O'Brien, T. and Peloquin, J., 2013. MAREDAT: towards a world atlas of MARine Ecosystem DATa. *Earth System Science Data,* 5(2), 227-239.

Burrows, M.T., Kamenos, N.A., Hughes, D.J., Stahl, H., Howe, J.A. and Tett, P., 2014. Assessment of carbon budgets and potential blue carbon stores in Scotland's coastal and marine environment. Scottish Natural Heritage Commissioned Report No. 761.

Chmura, G.L., Anisfeld, S.C., Cahoon, D.R. and Lynch, J.C., 2003. Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles*, 17(4), 1111.

Climate Action Plan, 2019. https://www.gov.ie/en/publication/ccb2e0-the-climate-action-plan-2019/. Date Accessed 20/11/2020.

Cloern, J.E., Foster, S.Q. and Kleckner, A.E., 2014. Phytoplankton primary production in the world's estuarine-coastal ecosystems. *Biogeosciences*, 11(9), 2477.

Cott, G.M., 2021. Case Study: Saltmarshes. In: Devoy, R., Cummins, V., Brunt, B., Bartlett D., and Kandrot, S (Eds), Chapter 12, Coastal Wetlands. Shorelines: The Coastal Atlas of Ireland. Cork: Cork University Press.

Cott, G.M., Chapman, D.V. and Jansen, M.A., 2013. Salt marshes on substrate enriched in organic matter: the case of ombrogenic Atlantic salt marshes. *Estuaries and Coasts,* 36(3), 595-609.

Cott, G.M., Jansen, M.A. and Chapman, D.V., 2012. Salt-Marshes on peat substrate: where blanket bogs encounter the marine environment. *Journal of Coastal Research*, 28(3), 700-706.

Crooks, S., Windham-Myers, L. and Troxler, T., 2019. Defining blue carbon: the emergence of a climate context for coastal carbon dynamics. In: Windham-Myers, L., Crooks, S. and Troxler, T.G. eds., A blue carbon primer: the state of coastal wetland carbon science, practice and policy. CRC Press, Boca Raton.

Cross, J.R. and Collins, K.D. 2017. Management Guidelines for Ireland's Native Woodlands. Jointly published by the National Parks & Wildlife Service (Department of Arts, Heritage, Regional, Rural & Gaeltacht Affairs) and the Forest Service. Forest Service, Department of Agriculture, Food & the Marine, Kildare Street, Dublin 2, Ireland.

Curtis, T.G.F. and Skeffington, M.S., 1998. The salt marshes of Ireland: an inventory and account of their geographical variation. *Biology and Environment: Proceedings of the Royal Irish Academy* (pp. 87-104). Royal Irish Academy.

Davidson, N.C. and Finlayson, C.M., 2019. Updating global coastal wetland areas presented in Davidson and Finlayson (2018). *Marine and Freshwater Research*, 70(8), 1195-1200.

De Grave, S., Fazakerley, H., Kelly, L., Guiry, M.D., Ryan, M. and Walshe, J., 2000. A study of selected maërl beds in Irish waters and their potential for sustainable extraction. *Marine Resource Series*, No. 10. Marine Institute, Dublin.

De Grave, S. and Whitaker, A., 1999. A census of maërl beds in Irish waters. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 9(3), 303-311.

Duarte, C.M., Wu, J., Xiao, X., Bruhn, A. and Krause-Jensen, D., 2017. Can seaweed farming play a role in climate change mitigation and adaptation? *Frontiers in Marine Science,* 4, p.100.

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

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

Edwards, A., 1980. Ecological studies of the kelp, *Laminaria hyperborea*, and its associated fauna in South-West Ireland, *Ophelia,* 19:1, 47-60,

EU Green Deal https://ec.europa.eu/info/strategy/priorities-2019-2024/european-green-deal_en. Date Accessed. 20/11/2020.

Falkowski, P., 2012. Ocean science: the power of plankton. *Nature,* 483(7387), S17-S20.

Filgueira, R., Strohmeier, T. and Strand, Ø., 2019. Regulating services of bivalve molluscs in the context of the carbon cycle and implications for ecosystem valuation. In Smaal, A., Ferreira, J., Grant, J., Petersen, J., Strand, Ø. (Eds.), Goods and Services of Marine Bivalves. Springer, Cham (231-251).

Foden, J. and Brazier, D.P., 2007. Angiosperms (seagrass) within the EU Water Framework Directive: a UK perspective. *Marine Pollution Bulletin*, 55(1-6), 181-195.

Fodrie, F.J., Rodriguez, A.B., Gittman, R.K., Grabowski, J.H., Lindquist, N.L., Peterson, C.H., Piehler, M.F. and Ridge, J.T., 2017. Oyster reefs as carbon sources and sinks. *Proceedings of the Royal Society B: Biological Sciences*, 284(1859), 20170891.

Fossitt, J.A., 2000. A guide to habitats in Ireland. Heritage Council, Ireland

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

François, R., Thibaud, M., Michel, L.N. and Gilles, L., 2018. Seagrass organic matter transfer in *Posidonia oceanica* macrophytodetritus accumulations. *Estuarine, Coastal and Shelf Science,* 212, 73-79.

Freiwald, A., Fossa, J.H., Grehan, A., Koslow, T. and Roberts, J.M., 2004. Cold-Water Corals: Out of Sight—No Longer Out of Mind. Centre, UNEPWC, ed, 1-84.

Gacia, E., Duarte, C.M., Marbà, N., Terrados, J., Kennedy, H., Fortes, M.D. and Tri, N.H., 2003. Sediment deposition and production in SE-Asia seagrass meadows. *Estuarine, Coastal and Shelf Science,* 56(5-6), 909-919.

Grall, J. and Hall-Spencer, J.M., 2003. Problems facing maerl conservation in Brittany. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 13, S55-S64.

Hall-Spencer, J.M., Kelly, J. and Maggs, C.A., 2008. Assessment of maerl beds in the OSPAR area and the development of a monitoring program. Department of Environment, Heritage and Local Government: Ireland.

Hall-Spencer, J.M., 1998. Conservation issues concerning the molluscan fauna of maerl beds. *Journal of Conchology Special Publication 2*, 271–286.

Hastings, R., Cummins, V. and Holloway, P., 2020. Assessing the impact of physical and anthropogenic environmental factors in determining the habitat suitability of seagrass ecosystems. *Sustainability,* 12(20), 8302.

Hendriks, I.E., Olsen, Y.S., Ramajo, L., Basso, L., Steckbauer, A., Moore, T.S., Howard, J. and Duarte, C.M., 2014. Photosynthetic activity buffers ocean acidification in seagrass meadows. *Biogeosciences,* 11(2), 333-346.

Herr, D, Vegh, T. and Von Unger, M., 2019. State of international policy for blue carbon actions. In Windham-Myers, L., Crooks, S. and Troxler, T.G. eds., A blue carbon primer: the state of coastal wetland carbon science, practice and policy. CRC Press, Boca Raton.

Herr, D. and Landis, E., 2016. Coastal blue carbon ecosystems. Opportunities for Nationally Determined Contributions. Policy Brief. Gland, Switzerland: IUCN and Washington, DC, USA.

Hession, C., Guiry, M.D., McGarvey, S. and Joyce, D., 1998. Mapping and assessment of the seaweed resources (*Ascophyllum nodosum, Laminaria* spp.) off the west coast of Ireland. *Marine Resource Series Volume 5.* Marine Institute, Dublin.
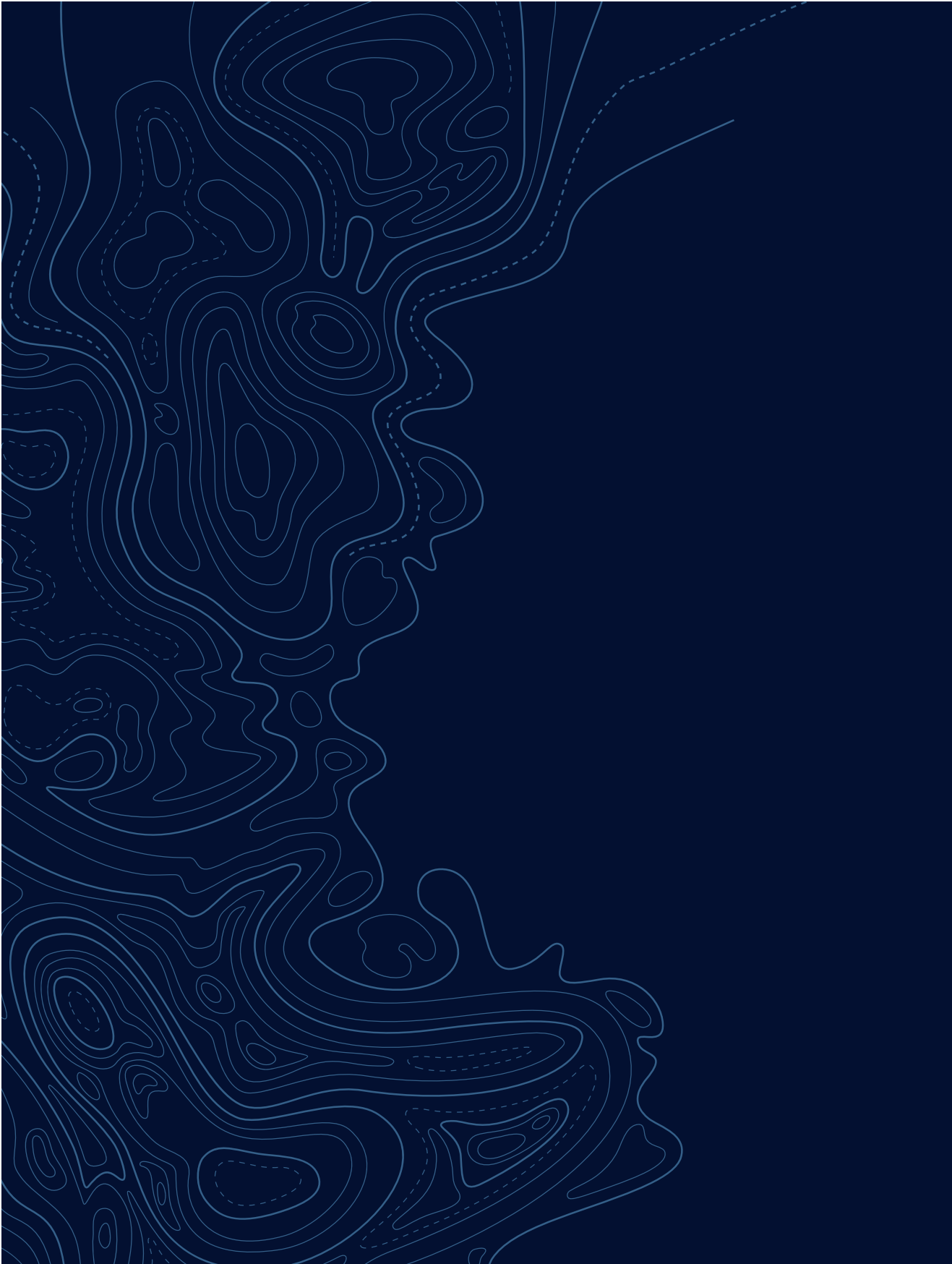
Hill, R., Bellgrove, A., Macreadie, P.I., Petrou, K., Beardall, J., Steven, A. and Ralph, P.J., 2015. Can macroalgae contribute to blue carbon? An Australian perspective. *Limnology and Oceanography,* 60(5), 1689-1706.

Hily, C., Grall, J., Chauvaud, L., Lejart, M. and Clavier, J., 2013. CO_2 generation by calcified invertebrates along rocky shores of Brittany, France. *Marine and Freshwater Research,* 64(2), 91-101.

Howard, J., Sutton-Grier, A., Herr, D., Kleypas, J., Landis, E., Mcleod, E., Pidgeon, E. and Simpson, S., 2017. Clarifying the role of coastal and marine systems in climate mitigation. *Frontiers in Ecology and the Environment,* 15(1), 42-50.

Huang, R.J., Thorenz, U.R., Kundel, M., Venables, D.S., Ceburnis, D., Ho, K.F., Chen, J., Vogel, A.L., Küpper, F.C., Smyth, P.P.A. and Nitschke, U., 2013. The seaweeds *Fucus vesiculosus* and *Ascophyllum nodosum* are significant contributors to coastal iodine emissions. *Atmospheric Chemistry and Physics,* 13, 5255-5264.

Huang, R.J., Seitz, K., Buxmann, J., Pöhler, D., Hornsby, K.E., Carpenter, L.J., Platt, U. and Hoffmann, T., 2010. In situ measurements of molecular iodine in the marine boundary layer: the link to macroalgae and the implications for O3, IO, OIO and NOx. *Atmospheric Chemistry & Physics Discussions,* 10(1).

Infomar, 2019. The Real Map of Ireland. Marine Institute, Ireland. http://hdl.handle.net/10793/1673

IPCC, 2013. Coastal Wetlands. In: Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M. and Troxler, T.G., 2014. 2013 supplement to the 2006 IPCC guidelines for national greenhouse gas inventories: Wetlands. IPCC, Switzerland.

IPCC, 2014. Climate change 2014: synthesis report. In: Pachauri, R.K., Meyer, L.A. (Eds.), In Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change IPCC. Geneva, Switzerland.

Jayathilake, D.R. and Costello, M.J., 2018. A modelled global distribution of the seagrass biome. *Biological Conservation,* 226, 120-126.

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

Keller, J.K., 2019. Greenhouse gases. In: Windham-Myers, L., Crooks, S. and Troxler, T.G. eds., A blue carbon primer: the state of coastal wetland carbon science, practice and policy. CRC Press, Boca Raton.

Kelly, E. (ed.) 2005. The role of kelp in the marine environment. Irish Wildlife Manuals, No. 17. National Parks and Wildlife Service, Department of Environment, Heritage and Local Government, Dublin, Ireland.

Keng, F.S.L., Phang, S.M., Abd Rahman, N., Elvidge, E.C.L., Malin, G. and Sturges, W.T., 2020. The emission of volatile halocarbons by seaweeds and their response towards environmental changes. *Journal of Applied Phycology,* 1-18.

Kirwan, M.L. and Megonigal, J.P., 2013. Tidal wetland stability in the face of human impacts and sea-level rise. *Nature,* 504(7478), 53-60.

Krause-Jensen, D., Lavery, P., Serrano, O., Marbà, N., Masque, P. and Duarte, C.M., 2018. Sequestration of macroalgal carbon: the elephant in the Blue Carbon room. *Biology Letters,* 14(6), p.20180236.

Krause-Jensen, D. and Duarte, C.M., 2016. Substantial role of macroalgae in marine carbon sequestration. *Nature Geoscience,* 9(10), 737-742.

Krumhansl, K.A. and Scheibling, R.E., 2012. Production and fate of kelp detritus. *Marine Ecology Progress Series*, 467, 281-302.

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

Lee, H.Z., Davies, I.M., Baxter, J.M., Diele, K. and Sanderson, W.G., 2020. Missing the full story: First estimates of carbon deposition rates for the European flat oyster, *Ostrea edulis. Aquatic Conservation: Marine and Freshwater Ecosystems,* 30(11), 2076-2086.

Lee, J.Y., Tett, P., Jones, K., Jones, S., Luyten, P., Smith, C. and Wild-Allen, K., 2002. The PROWQM physical–biological model with benthic–pelagic coupling applied to the northern North Sea. *Journal of Sea Research,* 48(4), 287-331.

Lejart, M., Clavier, J., Chauvaud, L. and Hily, C., 2012. Respiration and calcification of *Crassostrea gigas*: contribution of an intertidal invasive species to coastal ecosystem CO_2 fluxes. *Estuaries and Coasts,* 35(2), 622-632.

Lindenbaum, C., Bennell, J.D., Rees, E.I.S., McClean, D., Cook, W., Wheeler, A.J. and Sanderson, W.G., 2008. Small-scale variation within a *Modiolus modiolus* (Mollusca: Bivalvia) reef in the Irish Sea: I. Seabed mapping and reef morphology. *Journal of the Marine Biological Association of the United Kingdom*, 88(1), 133-141.

Lovelock, C.E. and Duarte, C.M., 2019. Dimensions of blue carbon and emerging perspectives. *Biology Letters,* 15(3), p.20180781.

Loughnane, C. and Stengel, D.B., 2002. Attached *Sargassum muticum* (Yendo) Fensholt found on the west coast of Ireland. *The Irish Naturalists' Journal,* 27(2), 70-72.

Lüning, K., 1990. *Seaweeds: their environment, biogeography, and ecophysiology*. John Wiley & Sons, New York, US.

Lynch, S.A., Morgan, E., Carlsson, J., Mackenzie, C., Wooton, E.C., Rowley, A.F., Malham, S. and Culloty, S.C., 2014. The health status of mussels, *Mytilus spp.,* in Ireland and Wales with the molecular identification of a previously undescribed haplosporidian. *Journal of Invertebrate Pathology*, 118, 59-65.

Macreadie, P.I., Serrano, O., Maher, D.T., Duarte, C.M. and Beardall, J., 2017. Addressing calcium carbonate cycling in blue carbon accounting. *Limnology and Oceanography Letters,* 2(6), 195-201.

Mann, K.H., 2000. Ecology of coastal waters: with implications for management: Blackwell Science, Oxford.

Marine Institute, 2018. Marine Institute Strategic Plan 2018-2022; Building Ocean Knowledge, Delivering Ocean Services. Marine Institute, Ireland. http://hdl.handle.net/10793/1382

Mazarrasa, I., Marbà, N., Serrano, O., Lovelock, C.E., Lavery, P.S., Fourqurean, J.W., Kennedy, H., Mateo, M.A., Krause-Jensen, D., Steven, A.D. and Duarte, C.M., 2015. Seagrass meadows as a globally significant carbonate reservoir. *Biogeosciences,* 12 (16).

Mazarrasa, I., Samper-Villarreal, J., Serrano, O., Lavery, P.S., Lovelock, C.E., Marbà, N., Duarte, C.M. and Cortés, J., 2018. Habitat characteristics provide insights of carbon storage in seagrass meadows. *Marine Pollution Bulletin,* 134, 106-117.

McCorry, M. and Ryle, T., 2009. Saltmarsh monitoring report 2007– 2008. NPWS, Dublin, Ireland: Department of Environment, Heritage and Local Government.

Mcleod, E., Chmura, G.L., Bouillon, S., Salm, R., Björk, M., Duarte, C.M., Lovelock, C.E., Schlesinger, W.H. and Silliman, B.R., 2011. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. *Frontiers in Ecology and the Environment,* 9(10), 552-560.

Mcowen, C.J., Weatherdon, L.V., Van Bochove, J.W., Sullivan, E., Blyth, S., Zockler, C., Stanwell-Smith, D., Kingston, N., Martin, C.S., Spalding, M. and Fletcher, S., 2017. A global map of saltmarshes. *Biodiversity Data Journal,* (5).

Megonigal, J.P., Chapman, S., Langley, A., Crooks, S., Dijkstra, P. and Kirwan, M., 2019. Coastal wetland responses to warming. In: Windham-Myers, L., Crooks, S. and Troxler, T.G. eds., A blue carbon primer: the state of coastal wetland carbon science, practice and policy. CRC Press, Boca Raton.

MERC, 2005. Survey of Sensitive Subtidal Benthic Communities in Kilkieran Bay and Islands SAC and Kingstown Bay SAC. Produced by MERC on Behalf of Marine Institute in Partnership with National Parks & Wildlife Service. Department of Environment, Heritage and Local Government.

Millard, D., 2019. http://www.bim.ie/media/bim/content/our-work/seaweed-workshop-ennis-2019/BIM-Seafood-Workshop-Ennis-2019-David-Millard-Seaweed-Sites-Equipment-and-Layout.pdf. Date Accessed 14/11/2020.

Morris, J.T. and Callaway, J.C., 2019. Physical and biological regulation of carbon sequestration in tidal marshes. In: Windham-Myers, L., Crooks, S. and Troxler, T.G. eds., A blue carbon primer: the state of coastal wetland carbon science, practice and policy. CRC Press, Boca Raton.

Morris, J.T., Edwards, J., Crooks, S. and Reyes, E., 2012. Assessment of carbon sequestration potential in coastal wetlands. In: Recarbonization of the biosphere (pp. 517-531). Springer, Dordrecht.

Murray, B.C., Jenkins, W.A., Sifleet, S., Pendleton, L. and Baldera, A., 2010. Payments for blue carbon: potential for protecting threatened coastal habitats. Policy brief. Durham: Nicholas Institute for Environmental Policy Solutions. Duke University.

Murray, R.H., Erler, D.V. and Eyre, B.D., 2015. Nitrous oxide fluxes in estuarine environments: response to global change. *Global Change Biology,* 21(9), 3219-3245.

Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestvedt,J., Huang, J., Koch, D., Lamarque, J.-F., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T., and Zhang, H.: Anthropogenic and Natural Radiative Forcing, In: Climate Change 2013: The Physical Science Basis, Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, edited by: Stocker, T. F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S. K., Boschung, J., Nauels, A., Xia, Y., Bex, V., and Midgley, P. M., Cambridge University Press, Cambridge, UK and New York, NY, USA, 659– 740, 2013.

Narayan, S., Beck, M.W., Reguero, B.G., Losada, I.J., Van Wesenbeeck, B., Pontee, N., Sanchirico, J.N., Ingram, J.C., Lange, G.M. and Burks-Copes, K.A., 2016. The effectiveness, costs and coastal protection benefits of natural and nature-based defences. *PloS One*, 11(5), e0154735.

Nellemann, C., Corcoran, E., Duarte, C.M., Valdés, L., DeYoung, C., Fonseca, L. and Grimsditch, G., 2009. Blue Carbon. A Rapid Response Assessment. United Nations Environment Programme, GRID-Arendal (Available at http://www.grida.no).

Neubauer, S.C. and Megonigal, J.P., 2015. Moving beyond global warming potentials to quantify the climatic role of ecosystems. *Ecosystems,* 18(6), 1000-1013.

Ní Longphuirt, S.N., Eschmann, C., Russell, C. and Stengel, D.B., 2013. Seasonal and species-specific response of five brown macroalgae to high atmospheric CO_2. *Marine Ecology Progress Series,* 493, 91-102.

Nitschke, U., Dixneuf, S., Schmid, M., Ruth, A.A. and Stengel, D.B., 2015. Contribution of living and degrading kelp to coastal iodine fluxes. *Marine Biology,* 162(9), 1727-1738.

Norton, T.A. and Burrows, E.M., 1969. Studies on marine algae of the British Isles. 7. *Saccorhiza polyschides* (Lightf.) Batt. *British Phycological Journal,* 4(1), 19-53.

NPWS, 2014. Kilkieran Bay and Islands SAC (site Code: 2111) Conservation Objectives Supporting Document-marine Habitats and Species (Unpublished). https://www.npws.ie.

O'Boyle, S. and Silke, J., 2010. A review of phytoplankton ecology in estuarine and coastal waters around Ireland. *Journal of Plankton Research,* 32(1), 99-118.

Ondiviela, B., Galván, C., Recio, M., Jiménez, M., Juanes, J.A., Puente, A. and Losada, I.J., 2020. Vulnerability of *Zostera noltei* to sea level rise: the use of clustering techniques in climate change studies. *Estuaries and Coasts,* 43(8), 2063-2075.

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

United Nations / Framework Convention on Climate Change, 2015. Adoption of the Paris Agreement, 21st Conference of the Parties, Paris: United Nations. Date Accessed: 18/11/2020.

Pendleton, L., Donato, D.C., Murray, B.C., Crooks, S., Jenkins, W.A., Sifleet, S., Craft, C., Fourqurean, J.W., Kauffman, J.B., Marbà, N. and Megonigal, P., 2012. Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems. *PloS One,* 7(9), e43542.

Poffenbarger, H.J., Needelman, B.A. and Megonigal, J.P., 2011. Salinity influence on methane emissions from tidal marshes. *Wetlands,* 31(5), 831-842.

Puerta, P., Johnson, C., Carreiro-Silva, M., Henry, L.A., Kenchington, E., Morato, T., Kazanidis, G., Rueda, J.L., Urra, J., Ross, S. and Wei, C.L., 2020. Influence of water masses on the biodiversity and biogeography of deep-sea benthic ecosystems in the North Atlantic. *Frontiers in Marine Science,* 7, 239.

Ralph, P.J., Durako, M.J., Enriquez, S., Collier, C.J. and Doblin, M.A., 2007. Impact of light limitation on seagrasses. *Journal of Experimental Marine Biology and Ecology,* 350(1-2), 176-193.

Roberts, J.M., Wheeler, A.J. and Freiwald, A., 2006. Reefs of the deep: the biology and geology of cold-water coral ecosystems. *Science,* 312(5773), 543-547.

Rossiter, T., Furey, T., McCarthy, T. and Stengel, D.B., 2020. Application of multiplatform, multispectral remote sensors for mapping intertidal macroalgae: A comparative approach. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 30(8), 1595-1612.

Scally, L., Pfeiffer, N. and Hewitt, E., 2020. The monitoring and assessment of six EU Habitats Directive Annex I Marine Habitats. Irish Wildlife Manuals, No. 118. National Parks and Wildlife Service, Department of Culture, Heritage and the Gaeltacht, Ireland.

Schiermeier, Q., 2007. Convention discourages ocean fertilization. *Nature* doi:10.1038/news.2007.230

Schmidt, A.L., Coll, M., Romanuk, T.N. and Lotze, H.K., 2011. Ecosystem structure and services in eelgrass *Zostera marina* and rockweed *Ascophyllum nodosum* habitats. *Marine Ecology Progress Series,* 437, 51-68.

Smale, D.A., Moore, P.J., Queirós, A.M., Higgs, S. and Burrows, M.T., 2018. Appreciating interconnectivity between habitats is key to blue carbon management. *Frontiers in Ecology and the Environment,* 16(2), 71-73.

Smale, D.A., Burrows, M.T., Moore, P., O'Connor, N. and Hawkins, S.J., 2013. Threats and knowledge gaps for ecosystem services provided by kelp forests: a northeast Atlantic perspective. *Ecology and Evolution,* 3(11), 4016-4038.

Smith, S.V., 2013. Parsing the Oceanic Calcium Carbonate Cycle: A Net Atmospheric Carbon Dioxide Source or a Sink? Association for the Sciences of Limnology and Oceanography. Waco, Texas.

Smithsonian Environmental Research Center, Global Change Research Wetland. https://serc.si.edu/gcrew. Date Accessed 22/11/2020.

Snelgrove, P.V., Soetaert, K., Solan, M., Thrush, S., Wei, C.L., Danovaro, R., Fulweiler, R.W., Kitazato, H., Ingole, B., Norkko, A. and Parkes, R.J., 2018. Global carbon cycling on a heterogeneous seafloor. Trends in *Ecology & Evolution,* 33(2), 96-105.

Stengel, D.B. and Dring, M.J., 1997. Morphology and *in situ* growth rates of plants of *Ascophyllum nodosum* (Phaeophyta) from different shore levels and responses of plants to vertical transplantation. *European Journal of Phycology*, 32(2), 193-202.

Trevathan-Tackett, S.M., Kelleway, J., Macreadie, P.I., Beardall, J., Ralph, P. and Bellgrove, A., 2015. Comparison of marine macrophytes for their contributions to blue carbon sequestration. *Ecology,* 96(11), 3043-3057.

Tully, O. and Clarke, S., 2012. The status and management of oyster (*Ostrea edulis*) in Ireland. *Irish Fisheries Investigations* (24). Marine Institute, Ireland. http://hdl.handle.net/10793/828

US EPA https://www.epa.gov/energy/greenhouse-gas-equivalencies-calculator Date Accessed:22/1/2021

van der Heijden, L.H. and Kamenos, N.A., 2015. Calculating the global contribution of coralline algae to carbon burial. *Biogeosciences,* 12(10), 7845-7877.

VM0033. https://verra.org/methodology/vm0033-methodology-for-tidal-wetland-and-seagrass-restoration-v1-0/. Date Accessed: 18/11/2020.

Waycott, M., Duarte, C.M., Carruthers, T.J., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, J.W., Heck, K.L., Hughes, A.R. and Kendrick, G.A., 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences,* 106(30), 12377-12381.

Went, A.J. 1962. Historical notes on the oyster fisheries of Ireland. *Proceedings of the Royal Irish Academy,* C, 62, 95-223.

Wilkes, R., Bennion, M., McQuaid, N., Beer, C., McCullough-Annett, G., Colhoun, K., Inger, R. and Morrison, L., 2017. Intertidal seagrass in Ireland: pressures, WFD status and an assessment of trace element contamination in intertidal habitats using *Zostera noltei. Ecological Indicators,* 82, 117-130.

Zamanian, K., Pustovoytov, K. and Kuzyakov, Y., 2016. Pedogenic carbonates: Forms and formation processes. *Earth-Science Reviews,* 157, 1-17.