Initial natural capital accounts for the UK marine and coastal environment

FINAL REPORT

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Summary

The aim of this project was to advance the development of natural capital accounts for the UK marine and coastal environment. Limited to the use of existing, available data, this analysis built on previous work, some very recently completed. The scale and complexity of the task was considerable not only because of the extent of the UK marine environment--approximately three times that of terrestrial environment, and much less accessible--but because of overlapping attribution of assets in coastal zones, the high mobility of marine natural assets and scarcity of information.

Following ONS guidance we assessed the extent of UK marine and coastal assets defined using EUNIS habitat classification. Examining UK marine and coastal habitats for the variety of ecosystem goods and services delivered, we focused on those services that provide significant and important benefits to the UK economy and societal wellbeing. Using logic chain analysis, we assessed the critical conditions for the sustainable delivery of services from these habitats, undertook an economic valuation of each of these key ecosystem services and discussed potential beneficiaries of UK marine ecosystem goods and services. Outputs include a systematic dataset, with gaps identified, in support of these accounts and prioritised recommendations for the improvement of the data and asset coverage.

Even with the consideration of only seven ecosystems services, some narrowly described, the value of the flow of goods and services from UK marine and coastal ecosystems is impressive, both in its scale and growth over the past 5 years. The harvest of finfish and shellfish is the ecosystem service that is perhaps most familiar to people for several reasons ---employment, cultural history, culinary tradition--- but in the larger context its economic value appears relatively minor. The relative importance of regulating services, namely climate regulation (specifically carbon sequestration), waste remediation and natural hazard protection, compared to the other services is most notable, in 2016 more than double the other services considered combined. Cultural value is clearly underestimated as limited to recreation, here only coastal visits, it leaves behind on-the-water experience as well as the broader cultural appreciation of UK maritime heritage.

Although not generally included in natural capital accounts, abiotic services are significant from two important perspectives: firstly, the significance of their contribution, materially and economically, and secondly, the potential impact of their development on the areas from which they are sourced. Exploitation of abiotic resources can have a significant detrimental influence on surrounding ecosystems. A better understanding of the interactions and trade-offs between these two types of resources is essential if both biotic and abiotic natural

capital assets and their services are to be developed with minimal detrimental effect to each other and the wider environment. Valuing marine and coastal ecosystems is difficult for many reasons, not least of which is the paucity of data and information on the broad spectrum of ecosystem services delivered, the interaction between several services, the associated critical supporting ecosystem services, and the conditions required to ensure the sustainable delivery of renewable services.

Although developing at a rapid pace the basic science remains inadequate to address the global challenges faced in this field. Data that we were able to access much was incomplete or inconsistent, often lacking the benefit of regular monitoring. Over a third of UK marine and coastal habitats fall in the category of just ‘Seabed’ or ‘Known unknown’. Extent of our broad-scale habitats, the necessary first level of understanding, must be the priority.

In terms of natural capital accounting a number of methodological issues remain widely debated: overlap between marine and coastal accounts with other UK natural capital; suitability of information developed from survey data as opposed to modelling; appropriate level of spatial disaggregation. Other issues that may need to be addressed with implications for the direction of funds for filling data gaps is the development of new technology and software and how analysis should reflect changing environmental conditions, whether due to extreme weather events, ecological recovery, restoration or an altered management regime.

We need to better understand the various ecological processes associated with the delivery of key ecosystem services, such as related to heavy metal movement in the marine environment, nutrient remediation in the deep sea, effectiveness of natural hazards and the boundaries relevant for carbon processing (at present not matching well EUNIS habitat categories). A better scientific understanding will enable more appropriate condition indicators to be developed. With much of UK history and culture having definite maritime links, more effort is needed to develop a better understanding of society’s appreciation and valuation of marine and coastal habitats. Better knowledge of the potential users and beneficiaries could open opportunities for joint working and civil society involvement.

Natural capital accounting by its very nature, i.e., the stepwise progression of the analysis, the link between biophysical and social science, the need for regular monitoring, will help place science and economic evidence at the forefront of decision-making. Identifying gaps in monitoring and evaluation will need to be addressed to enable policy to be more securely based on a sound understanding of the costs and benefits, including biophysical trade-offs, of different policy and development options. Understanding the ecosystem services associated with coastal habitats will be increasingly important for developing policy and management plans to address climate change impacts and enhance ecological as well as economic resilience in the UK coastal and marine environment.

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List of acronyms

AA	Affected Area
AIH	Available Intertidal Habitat
BAP	Biodiversity Action Plan
BEIS	Department for Business, Energy and Industrial Strategy
BGS	British Geological Survey
BOD	Biochemical Oxygen Demand
C	Carbon
CEFAS	Centre for Environment, Fisheries and Aquaculture Science
CEH	Centre for Ecology and Hydrology
CS	Countryside Survey
CSM	Common Standards Monitoring
DAFOR	Dominant, Abundant, Frequent, Occasional, Rare -DAFOR scale is used to assess relative abundance of plant species in a given area
Defra	Department for Environment, Food and Rural Affairs
DUKES	Digest of United Kingdom Energy Statistics
EA	Environment Agency
EEZ	Exclusive Economic Zone
EPA	Environmental Protection Agency
EQS	Environmental Quality Standards
EU UWWTD	European Union Urban Waste Water Treatment Directive
EUNIS	European Nature Information Systems
GBDVS	Great Britain Day Visits Survey
GDP	Gross Domestic Product
HABMOS	Habitat Map of Scotland
ICES	International Council for the Exploration of the Sea
Irish UWWTP	Irish Urban Waste Water Treatment Plants
JNCC	Joint Nature Conservation Committee
MENE	Monitor of Engagement with Natural Environment Survey
MMO	Marine Management Organisation
MSFD	Marine Strategy Framework Directive
N	Nitrogen
NACE EU	Nomenclature of Economic Activities - Nomenclature statistique des activités économiques dans la Communauté européenne
NCA	Natural Capital Accounts
NE	Natural England
NERC	Natural Environment Research Council
NIEA	Northern Ireland Environment Agency

NRW	Natural Resources Wales
NSW	National Survey for Wales
OMBT	Opportunistic Macroalgae Blooming Tool
ONS	Office for National Statistics
OSPAR	The Oslo-Paris Convention for the Protection of the Marine Environment of the North-East Atlantic
OWF	Offshore Wind Farm
P	Phosphorus
POC	Particulate Organic Carbon
QA	Quality Assurance
RR	Resource Rent
SEEA	United Nations System of Environmental Economic Accounting
SEPA	Scottish Environment Protection Agency
SIC	Standard Industrial Classification
SNA	System of National Accounts
SNH	Scottish Natural Heritage
SPNS	Scotland’s People and Nature Survey
SSB	Shelf Sea Biodiversity
UK HO	United Kingdom Hydrographic Office
UK NEA	United Kingdom National Ecosystem Assessment
UK SSB	UK Shelf Sea Biogeochemistry programme
UWWTP	Urban Waste Water Treatment Plants
WFD	Water Framework Directive
WPS	Water sports Participation Survey
WWTP	Wastewater Treatment Plants

1 Introduction

Aiming to better recognise the benefits of nature, in 2011 the UK Government committed to incorporating natural capital in the UK Environmental Accounts by 2020. The following year the Office for National Statistics (ONS) together with the Department of Environment and Rural Affairs (Defra) published a 'roadmap' detailing Government priorities for scoping and developing various types of accounts. The Government's commitment to this goal was reiterated in 2018 with the publication of the 25 Year Environment Plan (25YEP) and the UK Natural Capital: Interim Review and Revised 2020 Roadmap with the objective of developing a complete suite of accounts for all UK broad habitats by 2020. Accordingly, this project has developed an initial set of natural capital accounts for the marine and coastal ecosystems within UK waters in line with ambitions set out in the 25 Year Environment Plan, the Revised 2020 Roadmap and OSPAR aims on natural capital accounting. Thus, this work will enable Defra and ONS to significantly advance their work in producing a full set of natural capital accounts for all UK broad habitats by 2020.

The definition of natural capital adopted in this report is the one elaborated by the Natural Capital Committee (2014), specifically the 'elements of nature that directly or indirectly produce value to people, including ecosystems, species, freshwater, land, minerals, air and oceans'. Following Defra / ONS (2016) guidance, an assessment of the extent and condition of marine and coastal assets was carried out to supply physical accounts as the basis for monetary accounts based on the valuation of goods and benefits provided by marine and coastal natural capital.

The approach adopted in this project is taken from the Defra / ONS report '*Scoping UK Coastal Margin Ecosystem Accounts*' (2016) and existing national level accounts. This work builds on past and soon-to-be-completed work in the UK and internationally, benefitting from progress on the United Nations System of Environmental Economic Accounting (SEEA) framework, the SEEA meeting in Bangkok in 2018 (UNECE 2018) and the advice from the UK Natural Capital Committee, alongside the vision of the 2020 Roadmap.

Our initial study used existing available data to provide estimates of the value of specific ecosystem services flowing from broad-scale marine and coastal habitats within UK waters. Outputs include a systematic dataset, with gaps identified, in support of these accounts and prioritised recommendations for the improvement of data and asset coverage.

The first section of the report considers the ecological element, defining the marine environment as all habitats significantly influencing and influenced by marine processes and organisms. Thus, coastal and intertidal habitats are defined from a marine perspective to include those habitats that provide marine ecosystem services. We then reviewed the broad spectrum of ecosystem services provided by UK marine and coastal ecosystems before focusing on a few key ecosystem services considered to provide significant and important benefits to the UK economy and societal well-being. The next step examined the ability of the UK marine and coastal habitats to deliver these services. We identified important indicators of habitat condition through the lens of the specific ecosystem service under consideration. Following this we considered the monetary valuation for each ecosystem service, data needs and potential data sources available. The development of logic chains for the separate services facilitated the identification of economic valuation options. The valuation methods selected followed accepted standards within the limits of actual data available. The value of the marine and coastal environment as related to the specific ecosystem services highlighted is provided wherever possible but should be understood to be a conservative or low-bound estimate of the true value of these resources. Additionally, we identify and discuss the potential beneficiaries of marine ecosystem services in the UK. Throughout the report we highlight unresolved methodological issues and data gaps, including any potential solution. The final section examines the usefulness of such accounts for policy and management and considers the potential and feasibility of refining the accounts presented herein and discusses further developing these initial marine and coastal accounts.

2 Conceptual framework for assessment of marine and coastal ecosystems

The marine ecosystem services and benefits included in this report is based on those identified in the UK National Ecosystem Assessment Follow-on Report (UKNEA-FO 2014). Although recent work on terrestrial natural capital accounts may also include some coastal and littoral marine habitats (e.g. saltmarsh), it was felt that services flowing from these habitats were assessed from a terrestrial perspective with little, if any, consideration of their importance in the marine context (Defra / ONS 2016). This report examines ecosystem services from a marine perspective and includes those coastal and intertidal habitats that provide marine ecosystem services such as the regulating services of waste (nutrient) remediation or natural hazard protection.

2.1 Key habitats defined

In this report, following the recommendation in the Defra/ONS Principles of Natural Capital Accounting (ONS, 2017), we use the UK National Ecosystem Assessment (UK NEA) (2011) broad habitat definitions based on the European Nature Information System (EUNIS) habitat codes (European Environment Agency 2019). As in the UK NEA (2011), we include EUNIS sub-component categories of the habitat type. EUNIS classification incorporates a hierarchical structure to define habitats from a very broad scale (level 1) down to fine, species specific, scale (level 4, 5 or 6). Here we use levels 2 and 3, which are based on substrate type incorporating some distinctive, defining biological features important to the delivery of the specific ecosystem service of interest (European Environment Agency 2019). Habitats included in this study are presented in Table 1.

2.1.1 Marine Ecosystems

The definition of Marine Ecosystems provided by the European Environment Agency under the European Nature Information System (EUNIS) (European Environment Agency 2019) is:

> *Marine ecosystems comprise habitats directly connected to the oceans, i.e. part of the continuous body of water which covers the greater part of the earth's surface and which surrounds its land masses. Marine waters may be fully saline, brackish or almost fresh. Marine habitats include those below spring high tide limit (or below mean water level in non-tidal waters) and enclosed coastal saline or brackish waters, without a permanent surface connection to the sea but either with intermittent surface or sub-surface connections (as in lagoons). Rockpools in the supralittoral zone are considered as enclaves of the marine zone. Marine ecosystems also include littoral habitats, which are subject to wet and dry periods on a tidal cycle including tidal*

saltmarshes; marine littoral habitats, which are normally water-covered but intermittently exposed due to the action of wind or atmospheric pressure changes; and freshly deposited marine strandlines characterised by marine invertebrates. Waterlogged littoral saltmarshes and associated saline or brackish pools above the mean water level in non-tidal waters or above the spring high tide limit in tidal waters are included with marine habitats, as are constructed marine saline habitats below water level as defined above (such as in marinas, harbours, etc), which support a semi-natural community of both plants and animals. The marine water column includes bodies of ice in some cases.

2.1.2 Coastal Ecosystems

The definition of Coastal Ecosystems provided by the European Environment Agency under the European Nature Information System (EUNIS) (European Environment Agency 2019) is:

Coastal ecosystems encompass habitats that are above spring high-tide limit (or above mean water level in non-tidal waters) occupying coastal features and characterised by their proximity to the sea, including coastal dunes and wooded coastal dunes, beaches and cliffs. This category includes free-draining supralittoral habitats adjacent to marine habitats, which are normally only affected by spray or splash, such as strandlines characterised by terrestrial invertebrates and moist and wet coastal dune slacks, and dune-slack pools. It excludes supralittoral rock pools and habitats adjacent to the sea which are not characterised by salt spray, wave or sea-ice erosion.

Table 1. Description of EUNIS level 2 and level 3 habitats

Ecosystem	EUNIS Level 2	EUNIS Level 3	Habitat name	Description
Marine	A1		Littoral rock and other hard substrata	Littoral rock includes habitats of bedrock, boulders and cobbles which occur in the intertidal zone (the area of the shore between high and low tides) and the splash zone.
	A2		Littoral sediment	Littoral sediment includes habitats of shingle (mobile cobbles and pebbles), gravel, sand and mud or any combination of these which occur in the intertidal zone.
		A2.1	Littoral coarse sediment	Littoral coarse sediments include shores of mobile pebbles, cobbles and gravel, sometimes with varying amounts of coarse sand.
		A2.2	Littoral sand and muddy sand	Shores comprising clean sands (coarse, medium or fine-grained) and muddy sands with up to 25% silt and clay fraction.
		A2.3	Littoral mud	Shores of fine particulate sediment, mostly in the silt and clay fraction (particle size less than 0.063 mm in diameter), though sandy mud may contain up to 40% sand (mostly very fine and fine sand).
		A2.4	Littoral mixed sediments	Shores of mixed sediments ranging from muds with gravel and sand components to mixed sediments with pebbles, gravels, sands and mud in more even proportions.
		A2.5	Coastal saltmarshes and saline reedbeds	Angiosperm-dominated stands of vegetation, occurring on the extreme upper shore of sheltered coasts and periodically covered by high tides.
		A2.6	Littoral sediments dominated by aquatic angiosperms	Dominants are [*Eleocharis acicularis*], [*Eleocharis parvula*], [*Zostera*] spp.
		A2.7	Littoral biogenic reefs	The Littoral Biogenic Reefs habitat contains two biological subtypes, littoral [Sabellaria] reefs (A2.71) and mixed sediment shores with mussels (A2.72).
	A3		Infralittoral rock and other hard substrata	Infralittoral rock includes habitats of bedrock, boulders and cobbles which occur in the shallow subtidal zone and typically support seaweed communities.
	A4		Circalittoral rock and other hard substrata	Circalittoral rock is characterised by animal dominated communities (a departure from the algae dominated communities in the infralittoral zone).
	A5		Sublittoral sediment	Sediment habitats in the sublittoral near shore zone (i.e. covering the infralittoral and circalittoral zones), typically extending from the extreme lower shore down to the edge of the bathyal zone (200 m).

	EUNIS Level 2	EUNIS Level 3	Habitat name	Description
		A5.1	Sublittoral coarse sediment	Coarse sediments including coarse sand, gravel, pebbles, shingle and cobbles which are often unstable due to tidal currents and/or wave action.
		A5.2	Sublittoral sand	Clean medium to fine sands or non-cohesive slightly muddy sands on open coasts, offshore or in estuaries and marine inlets.
		A5.3	Sublittoral mud	Sublittoral mud and cohesive sandy mud extending from the extreme lower shore to offshore, circalittoral habitats.
		A5.4	Sublittoral mixed sediments	Sublittoral mixed (heterogeneous) sediments found from the extreme low water mark to deep offshore circalittoral habitats.
	A6		Deep Sea	The sea bed beyond the continental shelf break, which occurs at variable depth, but generally over 200 m.
		A6.1	Deep-sea rock and artificial hard substrata	Deep-sea benthic habitats with substrates predominantly of bedrock, immobile boulders or artificial hard substrates.
		A6.2-A6.5	Deep-sea sand, muddy sand, mud, mixed	Deep-sea benthic habitats with substrates predominantly of sand, muddy sand, mud or mixed particle size or gravel.
Coastal	B1		Coastal dunes and sandy shores	Sand-covered shorelines of the oceans, their connected seas and associated coastal lagoons, fashioned by the action of wind or waves.
	B2		Coastal shingle	Beaches of the oceans, of their connected seas and of their associated coastal lagoons, covered by pebbles, or sometimes boulders, usually formed by wave action.
	B3		Rock cliffs, ledges and shores, including the supralittoral	Rock exposures adjacent to the oceans, their connected seas and associated coastal lagoons, or separated from them by a narrow shoreline.

Source: European Environment Agency (EEA) 2019

2.2 Extent of key habitats

2.2.1 Defining habitats

Current legislation requires the UK to report on EUNIS level 3 habitat extent within its waters. To comply with this, and the requirement to make habitat data available for use by statutory nature conservations bodies, businesses and general public, JNCC has developed an interactive mapping tool that integrates survey data obtained at a range of spatiotemporal scales using a variety of methodologies. The product, known as the 'Combined Survey/Model EUNIS level 3 (L3) Map (hereafter referred to as the Combined Map), is created by combining data from field surveys, carried out by statutory nature conservation agencies under legislative reporting requirements, with the UKSeaMap (JNCC 2019). The latter is a suite of broad-scale predictive habitat maps developed by overlaying oceanographic models with substrate maps. Ellwood (2014) summarises the approach and confidence assessment as follows:

> *Before the available data could be combined into a single layer a rule-based approach was applied to select higher quality data in areas of overlap. The process includes a 5-stage decision tree to follow for each pair of maps with an area of overlap and a 3-step confidence assessment for each map.*
>
> ***3-step confidence assessment****: A qualitative score indicating the likelihood of a particular habitat being correctly mapped within a study area was calculated by scoring 3 factors likely to have a large effect on the overall accuracy of the habitat assignments: 1. Remote sensing coverage (0, 1 or 2 points) 2. Amount of sampling (0 or 1 point) 3. Distinctness of class boundaries, if remote sensing used (0 or 1 point).*
>
> *The final score for each map is between 0 and 4 with 4 representing the 'best' type of map. Note, however, that this is a qualitative assessment, therefore a score of 4 does not equate to a perfect or 100 % accurate map.*

Ellwood (2014) details the 5-stage decision tree, maps the extent of EUNIS level 3 habitats and provides an indication of where data were obtained from surveys or predictive modelling. Currently, only 9% of data used in the UKSeaMap are derived from surveys (JNCC 2019) and predictive modelling remains the best available method for mapping marine habitats in UK waters.

UKSeaMap 2018 covers EUNIS Category A (Marine Habitats). The models used to create this map are not suitable, however, for use with littoral habitats as the associated substrate (i.e. seabed) data cannot be used to accurately predict the extent of such dynamic habitats (JNCC 2019). Accordingly, the extent of Littoral habitat for EUNIS categories A1 (Littoral rock and other hard substrata) and A2 (Littoral sediment) were obtained using the 2016 version of the Combined Map. The Combined Map is released periodically. The latest version (2018), is currently undergoing QA and validation and is due for public release in May 2019 but was not available for use with this project.

Extent accounts were also required for coastal habitats (EUNIS Category B). Information on EUNIS Category B was obtained from partner organisations Natural Resources Wales (NRW), Scottish Natural Heritage (SNH), Natural England (NE) and Environment Agency (EA). At the time of writing we were unable to source suitable data from Northern Ireland Environment Agency (NIEA)[1] therefore the extent of littoral and coastal habitats in Northern Ireland is not included. Where data were provided using different classification systems or reporting mechanisms (e.g. Biodiversity Action Plan (BAP), Phase 1 Habitat Survey, OSPAR) the original classifications were aligned to EUNIS categories based on agreed conversion methods (JNCC 2018a).

2.2.2 Calculating habitat extent

Data were imported as Shapefiles into ArcGIS (Version 10.1). Individual layers corresponding to EUNIS level 3 habitats were extracted. Area values for each individual polygon within the habitat layer were calculated in hectares then added together to produce an extent value (ha) for each habitat.

Table 2 provides details of the extent (ha) and data source for each EUNIS level 2 or 3 habitat used for the extent accounts.

[1] The Northern Ireland Marine Mapviewer, a recently launched online tool, provides information on habitats within Northern Ireland marine and coastal waters (DAERA 2019).

Table 2. Extent of UK marine and coastal habitats and data source

EUNIS L2 / L3[a]	Habitat name	Area (ha)	Reference[b]
A1	Littoral rock and other hard substrata	21,656	JNCC 2017
A2.1	Littoral coarse sediment	7,248	JNCC 2017
A2.2	Littoral sand and muddy sand	187,831	JNCC 2017
A2.3	Littoral mud	100,303	JNCC 2017
A2.4, A2.6, A2.7	Littoral mixed sediments, sediments dominated by aquatic angiosperms, biogenic reefs	15,807	JNCC 2017
A2.5	Coastal saltmarshes and saline reedbeds	52,832	JNCC 2017
A3	Infralittoral rock and other hard substrata	292,127	JNCC 2019
A4	Circalittoral rock and other hard substrata	491,616	JNCC 2019
A5.1	Sublittoral coarse sediment	16,497,908	JNCC 2019
A5.2	Sublittoral sand	26,484,814	JNCC 2019
A5.3	Sublittoral mud	6,149,456	JNCC 2019
A5.4	Sublittoral mixed sediments	1,241,882	JNCC 2019
A6.1	Deep-sea rock and artificial hard substrata	633,871	JNCC 2019
A6.2-A6.5	Deep-sea mixed, sand, muddy sand, mud	2,887,260	JNCC 2019
B1	Coastal dunes and sandy shores	96,518	Scotland's Environment 2017, Lle 2019, Defra 2019
B2	Coastal shingle	10,494	Scotland's Environment 2017, Lle 2019, Defra 2019
B3	Rock cliffs, ledges and shores, including the supralittoral	25,542	Scotland's Environment 2017, Lle 2019, Defra 2019
Seabed	Mainly infralittoral. (No substrate data available therefore predictive modelling not possible.)	347,937	JNCC 2019
Known unknown	'Known unknown' habitat. (No survey data for some coastal and littoral habitats. Shallow sublittoral habitats that cannot be assessed using bathymetric surveys or physical surveys.)	33,068,352	JNCC 2019
Total		**88,613,454**	

[a] EUNIS level 3 habitats were retained where evidence was found showing variation in ecosystem service delivery from habitats at this level and aggregated to level 2 where further separation was not supported by the literature.
[b] Reference refers to publication date not date of data collection.

2.2.3 Limitations to extent calculations

This report uses the extent of habitats within UK marine and coastal waters including areas beyond the Exclusive Economic Zone (EEZ) out to the continental shelf because ecosystem services, such as mineral extraction, are provided by the UK continental shelf seas. Figure 1 shows the legislative boundaries for UK territorial waters, EEZ and continental shelf.

Our original aim was to use broad habitat categories at EUNIS Level 3 (European Environment Agency 2019). EUNIS level 3 habitats were retained where evidence was found showing variation in ecosystem service delivery from habitats at this level and were aggregated to level 2 where further separation was not supported by the literature. Habitats were not separated at levels lower than EUNIS level 3 nor higher than EUNIS level 2.

This presented difficulties for categories A2.6 (Littoral sediments dominated by aquatic angiosperms) and A2.7 (Littoral biogenic reefs) as these habitats are defined by level 4 species information (EEA 2019a). We took the decision to incorporate these into Category A2.4 (Littoral mixed sediments) but acknowledge that any future development of marine and coastal natural capital accounts would need to disaggregate these habitats.

There are several limitations to the development of accurate extent accounts for marine and coastal habitats, most significantly the paucity of data from temporally or spatially replicated surveys (in accessible coastal and littoral habitats). Habitat extent data in the JNCC Combined Map are often taken from a single extensive survey carried out over several years and not repeated. In general, such field surveys are required to report condition of priority or protected habitats for relevant environmental legislation. For example, a saltmarsh survey will assess condition of known areas of saltmarsh and only record other habitats present within that site.

The shelf-sea (EUNIS categories A3 – A5) and deep-sea habitats (EUNIS category A6) present huge challenges to the acquisition of reliable survey data. With an average depth of 80 m, mapping this habitat through field surveys is financially and technically difficult (Hooper et al. 2019). Therefore, using predictive modelling for shelf-sea and deep-sea areas, with additional validation surveys where resources are available, remains the most efficient method for establishing extent (JNCC 2019).

The extent of the UK seas is 88,613,000 ha (JNCC 2018b), the EEZ is currently mapped at 72,912,300ha, yet the extent of marine habitats shown in Table 2 is 55,545,102ha (including the Seabed category). The difference of 33,068,352 ha between the habitat extent calculated using the UKSeaMap and Combined Map, represents habitats of unknown classification due to lack of information. This extent is likely to consist of areas of coastal and littoral habitats for which no survey data exists. In addition, there are missing data comprising shallow sublittoral habitats that cannot be assessed by bathymetric techniques (to inform the predictive model) nor assessed using surveys. The extent of EUNIS habitats provided in Table 2 also incorporates the area of continental shelf beyond the EEZ.

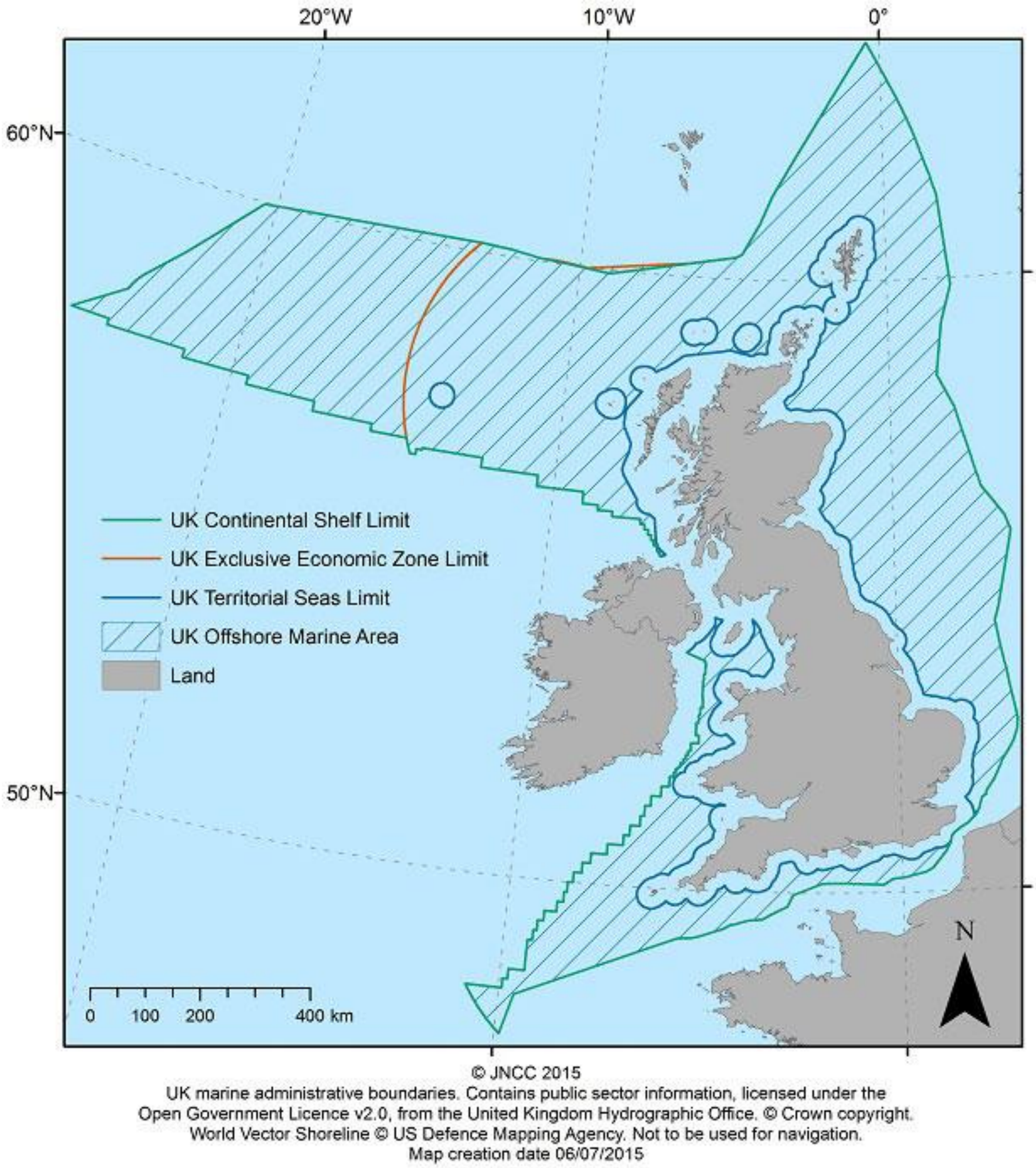

Figure 1. UK offshore marine area: the extent of UK Territorial Seas, Exclusive Economic Zone (EEZ) and Continental Shelf Limits (JNCC 2015)

2.3 Key ecosystem services delivered by marine and coastal habitats

2.3.1 Ecosystem services considered

Marine and coastal habitats provide myriad ecosystem services that benefit humans and our decision on which services to include in this analysis was informed by the relevance of the benefit and availability of data. Tables 3a and 3b set out the ecosystem services considered subdivided into provisioning, regulating, and cultural categories. Finfish / shellfish is the only *provisioning* service included because fisheries are accounted for in the SNA and data are, to a large degree, available from national and international agencies. In contrast, data concerning other provisioning services (see Table 3a) are scarce or not easily disentangled in SNA, presenting conceptual challenges for their accounting.

The *regulating* services included in this analysis are:

- Waste (nutrient) remediation, which in addition to providing a natural solution to human waste disposal provides the benefit of a healthier environment for humans as well as other life;
- Natural hazard protection, specifically the contribution of coastal and littoral habitats to the moderation of coastal erosion and flooding; and
- Climate regulation, specifically the contribution of carbon sequestration and storage to combatting climate change, thus providing the benefit of a healthy climate for human life;

With regard to *cultural* services, we only include 'places and seascapes' focusing on the recreational benefits provided by coastal and marine environments (e.g. nature watching, sea angling).

The Defra / ONS (2016) principles make a clear distinction between UK natural capital accounts, which include abiotic services, and the ecosystems accounts considered by System of Environmental-Economic Accounting (SEEA) Experimental Ecosystem Accounts) (UN et al. 2014), which exclude abiotic services. These by definition are not considered the result of ecosystem condition involving constant processes and therefore lack the main feature of regeneration or renewability. However, in this report, we have decided to *include* the abiotic services of wind for energy electricity generation and aggregate extraction and the related benefits these provide to society. The rationale is that the realisation of the benefits of abiotic services can impact marine habitats (for example, the location of wind turbines has an impact on the seabed, and aggregate extraction causes physical changes to the benthic habitat and associated biological community). The wider benefit of the marine environment, for example, as a median of transport, could be considered in future iterations of the assessment of marine ecosystem services and natural capital accounts. These impacts could be managed for greater sustainability of marine ecosystem services overall with the aid of information made available by ecosystem accounts for the marine environment.

Table 3a. Final ecosystem services included in the UK marine and coastal natural capital accounts

Services	Included	Not included
Provisioning services	Finfish and shellfish	Algae including seaweed
		Ornamental materials
		Genetic resources
		Water supply
		Aquaculture
Regulating services	Waste (nutrient) remediation	Remediation of other waste (e.g. heavy metals, pesticides) Nutrient processing within the water column
	Natural hazard protection	
	Climate regulation (carbon)	Temperature moderation
Cultural services	Recreational places and seascapes	Amenity Education Other cultural services
Abiotic services	Renewable energy (OWFs)	Infrastructure support
	Aggregates extraction	Transportation medium

Table 3b. Ecosystem service benefits assessed in the UK marine and coastal natural capital accounts

Ecosystem Service	Benefit assessed	Benefit not assessed
Finfish and shellfish	Food provision (wild)	Animal feed
		Fertiliser
		Aquaria specimens
		Medicinal compounds
		Biofuel
Waste (nutrient) remediation	Clean waters	
Natural hazard protection	Coastal erosion and flood prevention	
Climate regulation (carbon)	Healthy climate	Temperature mediation effects on health
Recreational places and seascapes	Tourism and nature watching	Spiritual well-being and Cultural heritage Aesthetic benefit Educational benefit
Wind energy	Electricity from a renewable source (OWF)	Other renewable energy (e.g., wave energy, floating solar) Non-renewable energy sources – coal, oil, gas
Aggregates	Construction material	Other abiotic products (e.g., deep sea minerals)

Finfish and shellfish, climate regulation, recreational spaces, aggregate extraction, and wind energy, as ecosystem services flowing from the marine environment have benefitted from extensive research. Therefore, we have drawn on existing studies for the assessment and development of accounts for these services. However, with regard to waste (nutrient) remediation and natural hazard protection, these have not previously been examined in such detail and significant knowledge gaps exist. Therefore, we decided to focus on these important ecosystem services in the preparation of initial natural capital accounts for marine and coastal ecosystems. Scientific data are fundamental to understanding the ecological processes underpinning the provision of ecosystem services and their significance. Scientific data could also be used to produce models and obtain estimates of likely impact and significance.

2.3.2 Waste (nutrient) remediation

Waste, with regard to the marine environment, can refer to any substance (organic, chemical or man-made) which enters the marine environment through anthropogenic input (Watson et al. 2016). The range of waste products entering the marine ecosystem can be grouped into three broad categories:

- Nutrients and organic matter; (e.g. nitrogen compounds; sewage (both human and agricultural));
- Biological wastes/contaminants (e.g. pathogens); and
- Persistent contaminants (e.g. plastics; petroleum products; heavy metals such as cadmium, arsenic, mercury).

Watson et al. (2016) define waste remediation as:

> *"The removal of waste products from a given environment by ecosystem processes[2] that act to reduce concentrations of wastes by the mechanisms of cycling/detoxification, sequestration/storage and export".*

Humans utilise abiotic and biotic processes within the marine and coastal environment to provide essential waste remediation services (Thurber et al. 2014). Without these ecosystems, potentially hazardous and deleterious waste products would need to be processed by society to avoid possible harmful consequences for human health (Armstrong et al. 2012; Watson et al. 2016). Initially this project aimed to examine waste remediation services provided for both nutrients and heavy metals. However, due to limited information and data quantifying the ecosystem service of heavy metal removal in marine environments, the study focused on remediation of nutrient waste across marine and coastal habitats. Here we focus on the ecosystem service of waste (nutrient) remediation provided by benthic habitats defined under EUNIS level 3. As such, nutrient cycling within the water column is beyond the scope of this initial study. However, it is acknowledged that nutrient cycling represents a valuable ecosystem service and benefit provided by marine and coastal waters and should be considered for future accounts.

Nutrient processing can be both a regulating service, i.e., removal of excess nitrogen flowing from land run-off (see e.g. Kitidis et al. 2017) and a provisioning service through supply of nutrients from riverine input through coastal systems to support biological processes in the marine environment (Sharples et al. 2016). Riverine sources supply around 10% of organic

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[2] ecosystem processes, in this context, are physical and chemical transformations naturally occurring through biotic and abiotic processes including, *inter alia*, sequestration through burial and bioturbation; dilution through wave action; export through surface or hydrological cycling.

and inorganic nutrients to the shelf sea environment (EUNIS habitats A3 – A5). These support biogeochemical processes including primary production and nutrient cycling (Watson et al. 2016; Kitidis et al. 2017; Sharples et al. 2016).

Excess nutrients flowing into the marine environment from land-based activities, including agriculture and urban run-off, can result in increased nutrient loading within estuarine ecosystems (Kadiri et al. 2014) leading to the development of dense macroalgal mats (Raffaelli 1999; Thornton 2016). Compounds of nitrogen and phosphorus are limited in marine environments, yet these nutrients are vital to stimulate primary production[3] (Armstrong et al. 2012). As in estuarine ecosystems, excess nutrients in sublittoral zones and beyond cause similar increases in primary production leading to algal ‘blooms’ and the rapid growth of eutrophic deoxygenated zones in the water column and sediment with negative impacts on fish and benthic invertebrate species (Watson et al. 2016; Kitidis et al. 2017).

By sequestering nutrients within plant tissue and sediment, coastal and littoral sediment habitats can provide a ‘buffer’ reducing the flow of excess nutrients into the marine environment (Deegan et al. 2012; Karpuzcu and Stringfellow 2012; Etheridge et al. 2015; Sousa et al. 2017; Redelstein et al. 2018). Soft sediment habitats may also prevent nutrient loading moving upstream from marine sources (Drake et al. 2008).

2.3.2.1 Littoral coarse sediment, Littoral sand and muddy sand, Littoral mud, Littoral mixed sediments

In contrast to the open sea environment where nutrients can be limited, estuaries are amongst the most naturally nutrient rich systems on Earth (Teichberg et al. 2010). Inputs of allochthonous (‘new’) nitrogen and phosphorus from land, via rivers and groundwater seepage, combine with tidal input from marine sources and autochthonous (‘recycled’) nutrients in the sediment to maintain a constant supply of nutrients which stimulates primary production in estuarine food-webs (Neilson and Cronin 1981; Raffaelli et al. 1999; Day et al. 2013).

As discussed above, the development of dense macroalgal mats, an indicator of eutrophication, can have significant impacts on ecosystem services provided by marine environments (Raffaelli 1999; Thornton 2016). For example, each summer, a large dense

[3] In marine ecosystems, phytoplankton and certain bacteria are able to convert inorganic matter into biomass using energy from solar radiation or chemical energy. They are the first link in the food chain and are therefore called the primary producers (autotrophs). All other life depends on the energy provided by these primary producers (Coastal Practice Network 2019).

macroalgal mat appears off the coast of Qingdao, China. In 2008 the mat, largely comprised of *Ulva prolifera* (Kong et al. 2010), covered an area of 12000 km^2 making this bloom the largest recorded (Liu, D. et al. 2009). Further research discovered the source to be rafts for expanded *Porphyra yezoensis* aquaculture 180 km north along the Yellow Sea coast (Liu, D. et al. 2009; Liu, F. et al. 2010).

Biomass and extent of macroalgal mats are important indicators of the ability of littoral sediment habitats (broadly aligned to EUNIS category A2) to provide the nutrient processing / sequestration service. For UK accounts, we selected the Opportunistic Macroalgae Blooming Tool (OMBT) developed to assess macroalgal mats, on littoral sediments, as an indicator of water quality under the Water Framework Directive (Best et al. 2011). Under the WFD OMBT, the presence (extent and biomass) of macroalgal mats indicates poor water quality, which could suggest retention of nutrients in low energy littoral sediment habitats and thus indicating a high level of nutrient input beyond the system's ability to process. However, there is agreement that certain types of estuaries are more susceptible to the effects of eutrophication (Kadiri et al. 2014), particularly those with a limited tidal range and restricted flushing (Scanlan et al. 2007). These micro-tidal estuaries have a range of <2 m and reduced water exchange resulting in lower dilution of effluents, with excess nutrients remaining available within the shallow water and sediment for longer before being flushed out to sea (McLusky and Elliott 2004). By contrast, in higher energy macro-tidal estuaries, water column mixing ensures excess nutrients do not remain within the system for long periods (Kadiri et al. 2014).

The OMBT is a multi-metric index composed of five metrics: (i) percentage cover of the available intertidal habitat (AIH); (ii) total extent of area covered by algal mats (affected area (AA)) or affected area as a percentage of the AIH (AA/AIH, %); (iii) biomass of AIH ($g\ m^2$); (iv) biomass of AA ($g\ m^2$); (v) presence of entrained algae (percentage of quadrats). This tool can be applied to all littoral sediment habitats (Best et al. 2011).

2.3.2.2 Coastal saltmarsh

In general, saltmarsh habitats are deemed less vulnerable to eutrophication and widely regarded as attenuators of nutrient enrichment (Simas and Ferreira 2007). These, and other wetland systems, are acknowledged as contributing to improving water quality through filtration of nutrients and other waste products (Simas and Ferreira 2007). The ability of saltmarsh habitats to process or sequester nutrients is dependent upon several factors the two most important being extent of habitat (enabling zonation to develop), and plant community (with the associated above / below ground biomass) (Sousa et al. 2017; Redelstein et al. 2018).

Extent is an important indicator of saltmarsh capacity for nutrient processing and / or storage: Sousa et al. (2017) found that 90% of nutrient stocks were retained in plants and sediment in mid-high saltmarsh (low marsh is closer to the sea and therefore inundated for longer periods within a tidal cycle). These areas develop through landward development with zonation of plant species occurring in larger areas of saltmarsh (Sousa et al. 2017). However, it has not been possible, within the time and financial constraints of this project, to determine the extent of high, medium and low saltmarsh in the UK, nor is it possible to determine proximity of the habitat to any nutrient source (either direct or diffuse).

Recent research on the temperate Rio de Aveiro, Portugal (one of Europe's largest continuous saltmarshes) found variation in nutrient stocks between different plant species (Sousa et al. 2017). Samples were taken every two months from February to December 2012 with no repeated surveys in subsequent years. Results found total N stock of 38100 tonnes across the 4400ha mixed species saltmarsh. Although this research was carried out in Portugal, results can be used to provide indication of the capacity of UK coastal saltmarsh habitats to provide nutrient sequestration/storage services. Saltmarsh habitats across western Europe are functionally similar and share some common saltmarsh genera (such as *Spartina* spp.).

2.3.2.3 Shelf-sea sediments

Three important indicators for the capacity of shelf sea sediment habitats to process nutrients were identified from the results of an extensive NERC funded project on UK Shelf Sea Biogeochemistry (UK SSB) (Kröger et al. 2018) and research carried out by Watson (et al. 2016) on the role of shelf and deep-seas in waste processing (including nutrients). Whilst it is acknowledged that bioturbation plays a vital role in the sequestration of excess nutrients within benthic sediments, there are insufficient data currently available to support the provision of a metric that would enable bioturbation to be used as an indicator of the capacity of shelf-sea sediment to provide waste (nutrient) remediation (Kröger et al. 2018). In addition, export or dilution of nutrients through hydrological processes within the water column, is not quantifiable as an indicator for the purposes of this study (Watson et al. 2016). However, variation in sediment type may be a useful indicator for N processing service in shelf-seas.

Research by Kitidis et al. (2017) found higher rates of N removal in cohesive sediments (e.g. mud, sandy mud and muddy sand). These sediments comprise particles <0.18 mm which are able to pack tightly together and, as a consequence, are more difficult to re-suspend through hydrological action. Sediment comprising larger particles (i.e. advective sediment), particularly if these are coarse, are more easily re-suspended through wave action or current flow (Gray and Elliott 2010). Advective sediments re-mineralise a higher quantity of nutrients thus making them more available to flow into the water column and deep sea. The boundary value between the different types of sediment is silt content of approximately 8% (Kröger et al. 2018). Watson et al. (2016) also found that variation in sediment types was likely to result in differing rates of N removal.

It is estimated that only 30-65% N is removed inside estuaries prior to flowing to the shelf sea habitat (Sharples et al. 2016). Indeed, Jickells and Weston (2011) point out that estuaries with short water residence times will have less effect on the quantity of material reaching the shelf. Nutrients flowing from estuarine and coastal habitats can settle and be retained in sediment in the shelf sea system (Watson et al. 2016).

Key knowledge gaps relating to nutrient sequestration were identified following research under the NERC UK SSB Project (Kröger et al. 2018). In particular, there is a pressing need to determine the rate of N (and other nutrients) sequestered within shelf-sea habitats and the flow of these essential nutrients to deep-sea environments for primary production and nutrient cycling.

2.3.2.4 Deep-sea

There is limited knowledge and understanding around the process of nutrient cycling within the soft-sediment deep-sea ecosystem. However, for the purposes of these accounts, it would be reasonable to suppose that hydrological processes and substrate differences facilitating shelf-sea nutrient processing would, to a certain extent, be similarly applicable in the deep-sea. In addition, biological processes selectively remove nitrogen from organic matter and release nitrate, nitrite and ammonium back into the water column (Thurber et al. 2012). Some nitrogen is retained within the sediment and, through bioturbation, is further buried or released to continue the cycle (Thurber et al. 2012; Watson et al. 2016). Yet, despite these vital processes, there is limited research or data on the mechanisms supporting nutrient remediation service within the deep-sea, largely due to its inaccessibility.

2.3.2.5 Coastal dunes and sandy shore

Sand dunes are dynamic systems with dunes closer to the sea undergoing constant shifting through sediment transport and other coastal processes (Sigren et al. 2014; Hanley et al. 2014). These systems rely on zonation through plant species succession which results in dune areas further from the sea becoming more stable (Hanley et al. 2014). Nutrients flow through the system and are not retained. Therefore, sand dunes do not provide effective nutrient removal services (JNCC 2004). Indeed, a low soil organic content results in high N leaching (>65%) (Defra / CEH 2010). However, the presence of artificial structures, such as sea walls, can constrain the succession process leading to coastal squeeze and consequential loss of ecosystem function (Hanley et al. 2014). In addition, eutrophication of dunes can lead to further reduction in function as dunes become colonised by N tolerant species such as nettles (*Urtica dioica),* thistles *(Cirsium* sp.) and ruderal grasses (e.g. *Arrhenatherum elatius*) (Defra / CEH 2010). These plants effectively advance the succession process resulting in a reduction in vegetation zonation and ecosystem function. The presence of these ruderal species is indicative of excess nutrient retention within the dune system and, therefore, reduction in N removal out to sea.

The Common Standards Monitoring Guidance for Sand Dunes (JNCC 2004) uses the DAFOR scale (Dominant, Abundant, Frequent, Occasional, Rare) for plant community to assess habitat condition (e.g. zonation / succession). Under the CSM, nutrient enrichment indicator species should be not >F or, collectively, have <5% coverage (JNCC 2004). Using this as a baseline we propose a sliding scale of abundance and coverage potentially indicative of a sand dune habitat's decreasing capacity to process nutrients.

2.3.3 Natural hazard protection

Coastal ecosystems afford protection for inland habitats against natural hazards such as erosion and coastal flooding that may occur as a result of extreme weather events or high tides. The effectiveness of this protection may vary by the type and condition of the habitat as well as the extent and orientation of the coastal ecosystem vis-à-vis the hazard. Here we examine littoral and coastal habitats in terms of their effectiveness to provide natural hazard protection.

2.3.3.1 Littoral coarse sediment, Littoral mixed sediment, Littoral sand and muddy sand

Möller et al. (1999) compared wave height attenuation between saltmarsh and sandflats at adjacent sites on the North Norfolk coast. Results showed wave height decreased by an average of 15.29% over a 200 m sandflat. Measurements were taken during field-based experiments using short burst wave energy.

2.3.3.2 Littoral mud

A field-based study by Möller and Spencer (2002) examined variation in wave height and wave energy attenuation between saltmarsh and littoral mud at two sites in Essex (Tillingham and Bridgewick). These were characterised by differences in shore profile with Tillingham's shoreline gradually transitioning between mudflat and saltmarsh whereas the boundary between the two habitats at Bridgewick was formed by a 1.5 – 2.0 m cliff.

Results from the study showed wave height reduction of 20.57% over 147 m (0.14% m^{-1}) and wave energy reduction of 35.25% over 147 m (0.24% m^{-1}) across Tillingham mudflats. However, wave height and wave energy *increased* across 102 m Bridgewick mudflats (height: 23.91% (0.23% m^{-1}), energy: 55.06% (0.54% m^{-1})) (Möller and Spencer 2002).

2.3.3.3 Coastal saltmarsh

Saltmarsh is acknowledged as an effective 'buffer' against natural hazards including coastal flooding and storms (Shepard et al. 2011; Möller et al. 2014; Leonardi et al. 2018). Due to the complexities and dynamic variability in parameters, it is difficult to quantify the most effective extent of saltmarsh for natural hazard protection. Confounding factors include wave height, height of saltmarsh and whether the leading edge is gradual or a 'cliff', plant species and vegetation structure, and tidal height (Möller et al. 1999; Möller and Spencer 2002; Möller et al. 2014).

Using this summary supported by results from tank-based work, *in situ* experiments and habitat spatial models (Möller et al. 1999; Möller and Spencer 2002; Temmerman et al. 2012; Möller et al. 2014), we identified three important indicators for saltmarsh capacity to provide natural hazard protection:

(i) Distance from land boundary (i.e. width of saltmarsh): Field experiments in Tillingham (Essex) recorded a 63% wave height reduction for saltmarsh width >200 m with most wave height energy dissipated in the first 10 – 50 m (Möller et al. 1999). However, other studies found significant wave height reduction over much shorter distances (Möller and Spencer 2002; Möller et al. 2014). In field-based experiments, Möller and Spencer (2002) recorded a >80% wave attenuation over >160 m saltmarsh under low energy conditions. More recent work using recreated saltmarsh in a 300 m flume recorded 20% wave attenuation over a 40 m width of saltmarsh (Möller et al. 2014).

(ii) Distance travelled by wave: Möller and Spencer (2002) found wave height and wave energy attenuation were greater across saltmarsh compared to littoral mud. It is worth noting that the shore profile at Tillingham showed a gradual upward incline and a transition zone between littoral mud and saltmarsh. At Bridgewick, the saltmarsh / mudflat boundary was formed by a 1.5 – 2.0 m cliff. Results from these experiments found wave height attenuation across Tillingham saltmarsh of 87.37% over 163 m (0.54% m^{-1}) and across Bridgewick saltmarsh of 43.81% over 10 m (4.38% m^{-1}). Wave energy (Joules m^{-1}) was reduced by 98.92% (0.61% m^{-1}) across Tillingham saltmarsh and 79.13% over 10 m (7.91% m^{-1}) across Bridgewick saltmarsh (Möller and Spencer 2002).

(iii) Habitat contiguousness and degree of homogeneity: Using modelled data, Temmerman et al. (2012) found a difference in wave inundation and dissipation depending upon the degree of spatial homogeneity and proximity to channels. Even a 50% random marsh die-off (provided this was not adjacent to channels) only resulted in a small reduction in storm protection.

These indicators could be developed further through experimental work and the use of remote sensing to create ecosystem service maps for saltmarsh. Machine learning technology, for example, can be used to quantify the natural hazard protection afforded by saltmarsh and produce maps showing areas of good service provision whilst highlighting areas where management intervention would be necessary to ensure enhanced natural hazard protection.

Man-made sea defences have been constructed to provide coastal communities with protection from storm surges and other natural hazards from the marine environment (Spencer et al. 2016). These have undoubtedly been beneficial in supporting the ecosystem service provided by the natural coastal habitat. However, the combined effects of coastal erosion and sea level rise may result in a reduction in the overall extent of saltmarsh and particularly high- and mid-marsh through ‘coastal squeeze’ – a process whereby landward accretion of marsh is impeded by hard structures such as sea walls and dikes (Spencer et al. 2016.). The value of saltmarsh ecosystems could be derived by looking at the economic value of the land that they protect, both when they are located on the seaward side of man-made defence structures or where land is protected naturally.

2.3.3.4 Coastal dunes and sandy shore

Dunes have a significant role in buffering storms and other extreme natural events, providing an important element of coastal defence (Sigren et al. 2014; Hanley et al. 2014; Sigren et al. 2018). Recent research has highlighted the importance of vegetation, particularly mature forbs, in reducing erosion and scarp retreat. Indeed, Sigren et al. (2014) found the presence of vegetation reduced the dune scarp retreat by over 30%. Sand dunes are dynamic habitats with well-established dunes showing successional zones. It is acknowledged that the presence of a diversity of native plant species enhances the capacity of dunes to provide natural hazard protection (Hanley et al. 2014). Current monitoring of sand dunes, required under existing legislation (JNCC 2004), could also record zonation as an indication of the maturity of the plant community and thereby its ability to withstand storm activity.

2.3.3.5 Shingle bank

Shingle banks are created through wave action and storm surges throwing the pebbles above the high-water mark. Without human intervention, these banks can be significantly higher than the land behind. For example, the Portland end of Chesil Beach, near Weymouth in Dorset, is 14 m high and towers over the low-lying land it protects (May 2003). As the slope and height of a shingle bank are created by wave action, the only indicator of capacity to provide effective natural hazard protection is the extent of the bank.

3 Indicators of habitat condition relevant to ecosystem services' delivery for the ecosystem services considered in this report

3.1 Development of suitable indicators

Generally speaking the extent of a particular habitat gives a reliable indication of its condition or capability to provide ecosystem goods and services. The capacity of a particular ecosystem to provide goods and services, however, is likely to depend on a combination of current and historical impacts by human or natural agents on the environment. Human activities can have positive or negative impacts on the environment, which correspondingly enhance or impair the subject habitat's ability to provide ecosystem goods and services, either currently or over the long term. Negative impacts can derive from a range of human activities including but not limited to trawling, sewage outfall, and installation of hard infrastructure. Positive human influence can manifest itself through management activities which protect an area from extractive activity, remove or impede invasive species, or intercept toxic flows, for example. Activities with positive or negative impacts can affect different habitats with different intensity or for different durations. The resilience of the various habitats should be considered in valuation as well as in the planning and management of these areas.

The capacity of the environment to deliver ecosystems goods and services thus depends on its 'condition' which cumulatively reflects human interventions past and present as well as exogenous environmental change. The condition of the habitat is important in assessing the value of the natural capital as condition determines the flow of particular ecosystem goods and services over time. To determine condition of any habitat vis-à-vis its ability to deliver targeted goods and services one must focus on the critical factors, as defined by scientific evidence influencing ecosystem processes and resulting in the delivery of ecosystem services. This information will help improve our understanding not only of sustainability of ecosystem services but trade-offs in the delivery of multiple ecosystem services from any particular habitat.

Our examination of the condition of UK marine and coastal habitats with regard to their capacity to deliver key ecosystem services began with the development of a logic chain for each ecosystem service included in this report. To populate the logic chain with the required information we began by focusing on the ecosystem service in question, asking which habitats might be involved in contributing to the delivery of this service. Having defined the relevant habitats, we then needed to determine the characteristics of these habitats that enable them to deliver a specific ecosystem service on a sustainable basis. Focusing on these identified characteristics, we then identified the measurable aspects of these characteristics, assuming we have the appropriate technology, that could be practically used as indicators of the capacity of the habitat to produce goods and services.

With information provided by these indicators we should be able to assess whether the habitat was delivering at full capacity (the uppermost level of its capability) or to some degree less, say, a certain percentage of full capacity. Presumably this percentage figure could then be applied to the assessment of value flow to help determine the natural capital value of the habitat. Of course, if the impairment to capacity were judged to be only a temporary impediment then the pattern of recovery would need to be considered in the asset value projection calculations.

3.2 Logic chains

Logic mapping can be an excellent framework for enhancing the focus and robustness of an evaluation of a process or chain of activities. A logic chain or map provides a systematic way of “visualising the key steps required in order to turn a set of resources or inputs into activities that are designed to lead to a specific set of changes or outcomes.” (Hillis 2010). Logic chains, generally ‘read’ from left to right, lead one step-wise through a sequence from initial input through specified processes to projected outcomes. As an evaluation approach it useful for assessing complex systems and exploring the underlying ‘mechanisms’ by which given inputs under certain conditions generate specific outputs.

Using peer reviewed and grey literature we determined enabling characteristics and thus suitable indicators by understanding how the physical features and ecological processes of a habitat contribute to the delivery of the defined ecosystem service within each habitat. Relationships between the various habitats and how these are understood to affect the provision of ecosystem good and services is less clearly understood and not addressed here. Focusing on the enabling characteristics in each logic chain (cf. the central block in Figures 2 - 8), we highlight (as noted with *) the most important indicators needed to assess the condition of individual habitats with specifically with regard to their ability to provide the identified ecosystem services. Tables 4 and 5 provide more detailed analysis of indicators for habitat condition required to deliver the ecosystem services of waste (nutrient) remediation and natural hazard protection. These, together with top indicators for the other ecosystem services identified, are aggregated by habitat and ecosystem service in the matrix of Table 6. It will be noted that some indicators are useful for signalling capacity to deliver more than one ecosystem service.

Logic chains for the several key ecosystem services considered, namely wild-caught fish and shellfish, waste remediation, natural hazard protection, climate regulation, recreation, wind energy and aggregates extraction, are presented in Figures 2 - 8 below.

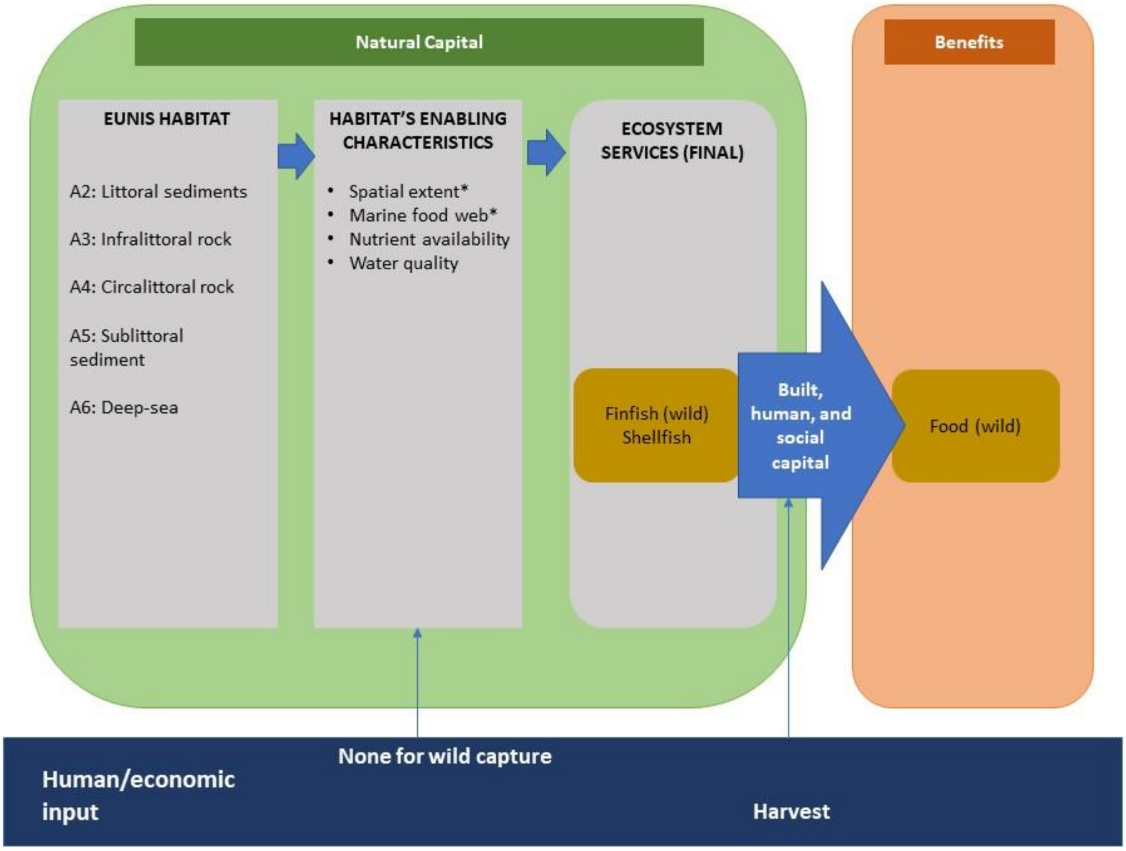

Figure 2. Logic chain for finfish and shellfish
* denotes those enabling characteristics deemed most important for the delivery of the provisioning service of finfish and shellfish.

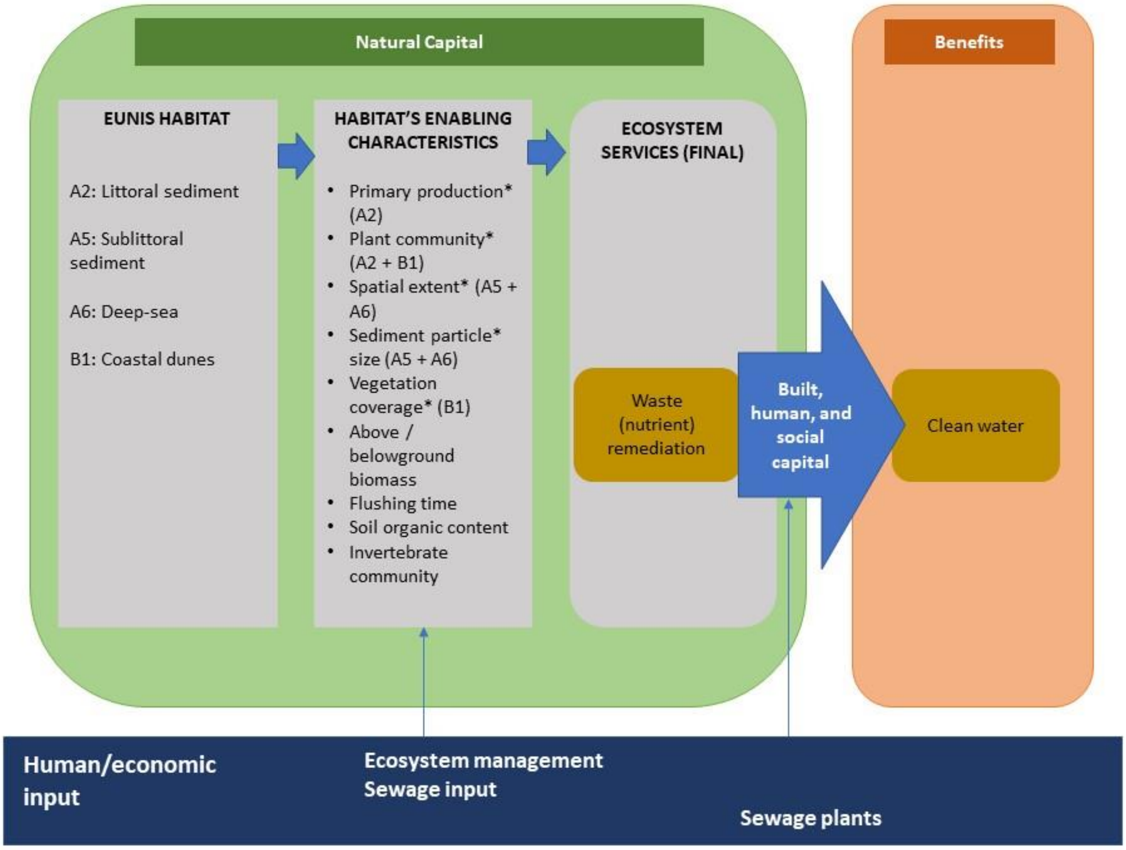

Figure 3. Logic chain for waste (nutrient) remediation
* denotes those enabling characteristics deemed most important for the delivery of the regulating service of waste (nutrient) remediation.
Note: Nutrient remediation and primary production within the water column is not considered here.

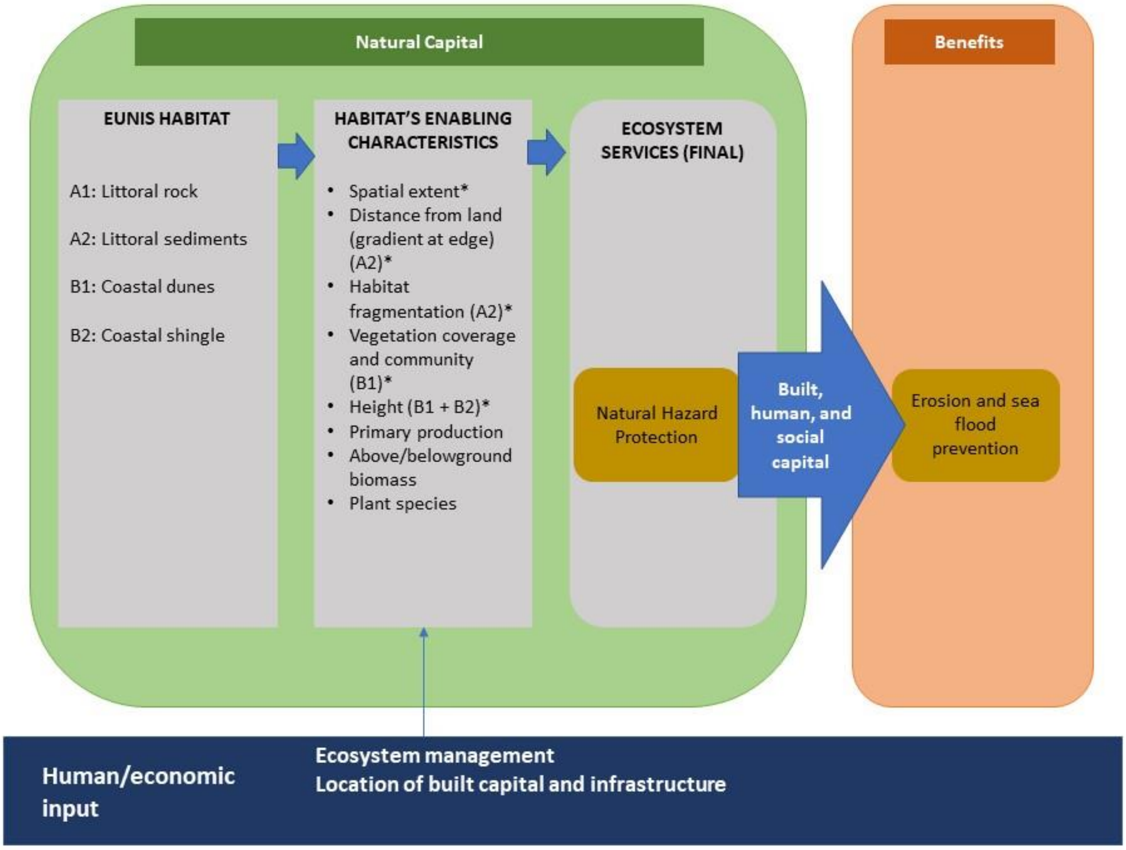

Figure 4. Logic chain for natural hazard protection
* denotes those enabling characteristics deemed most important for the delivery of the regulating service of natural hazard protection.

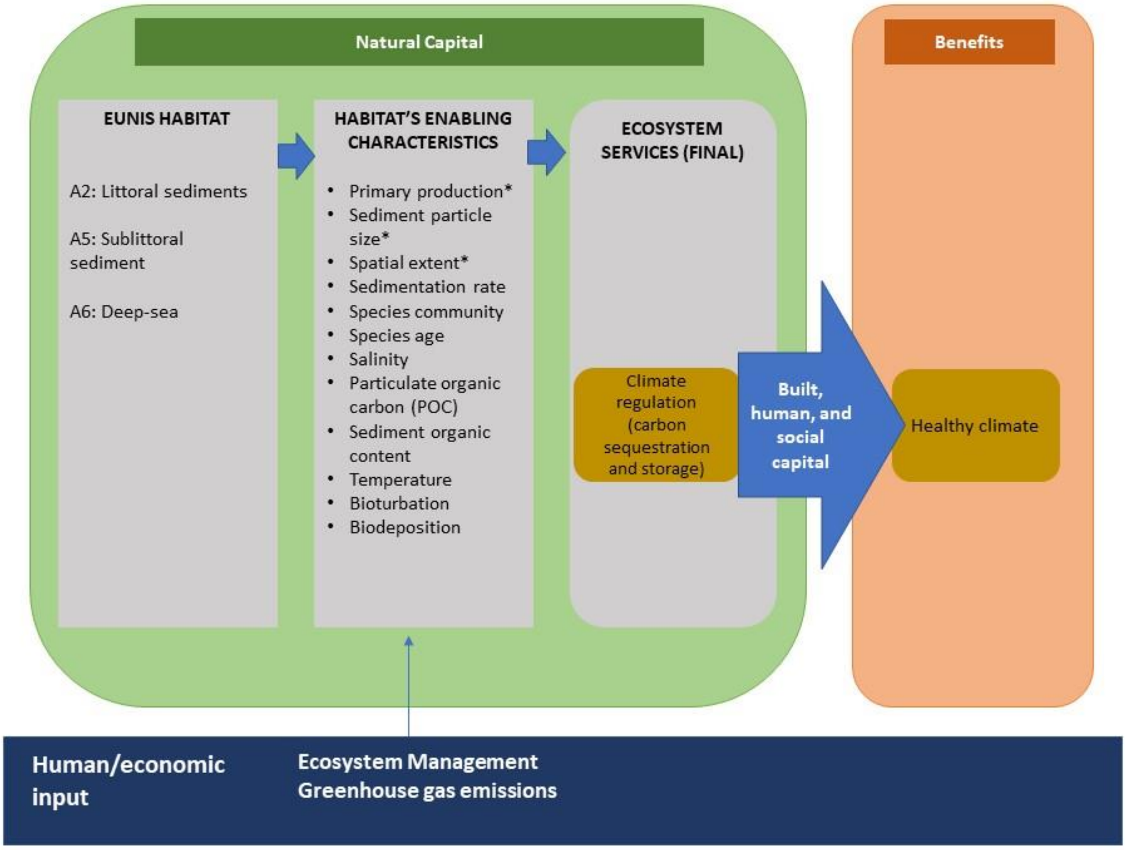

Figure 5. Logic chain for climate regulation (carbon)
* denotes those enabling characteristics deemed most important for the delivery of the regulating service of climate regulation (carbon sequestration and storage).

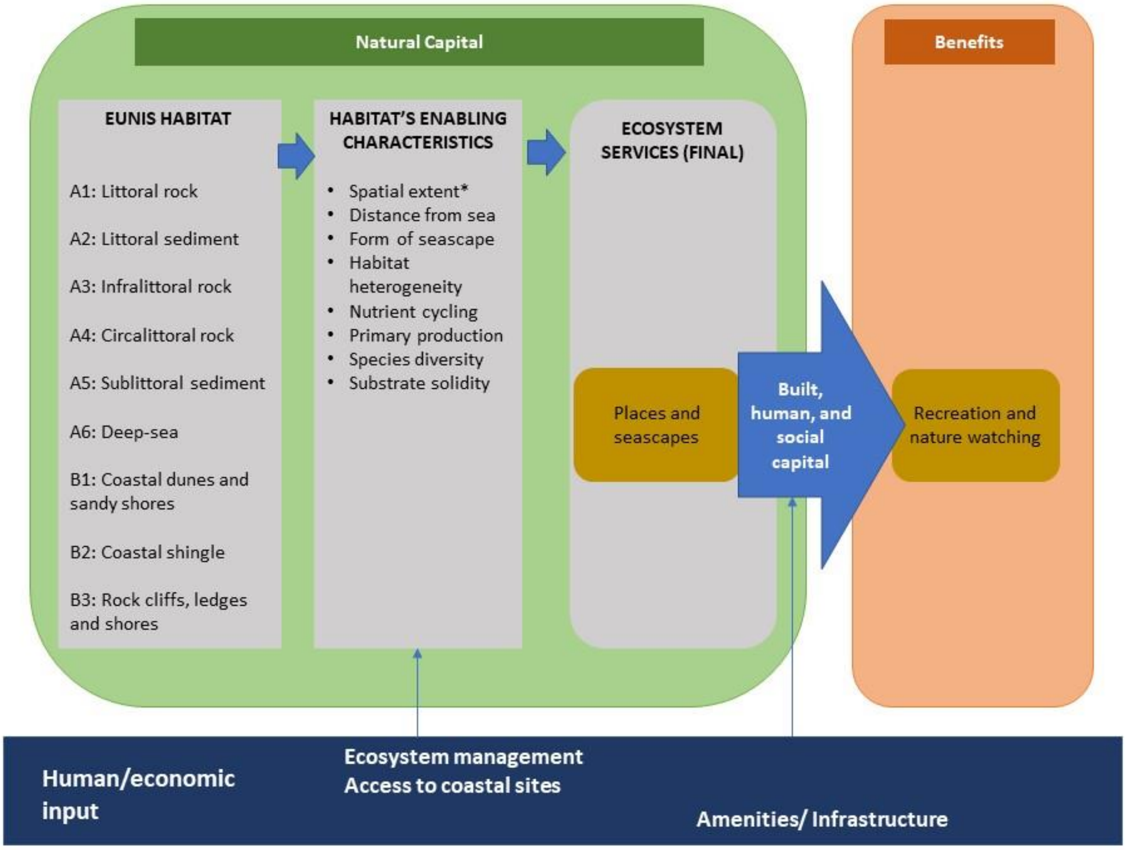

Figure 6. Logic chain for recreational places and seascapes
* denotes those enabling characteristics deemed most important for the delivery of the cultural service of recreational places and seascapes.

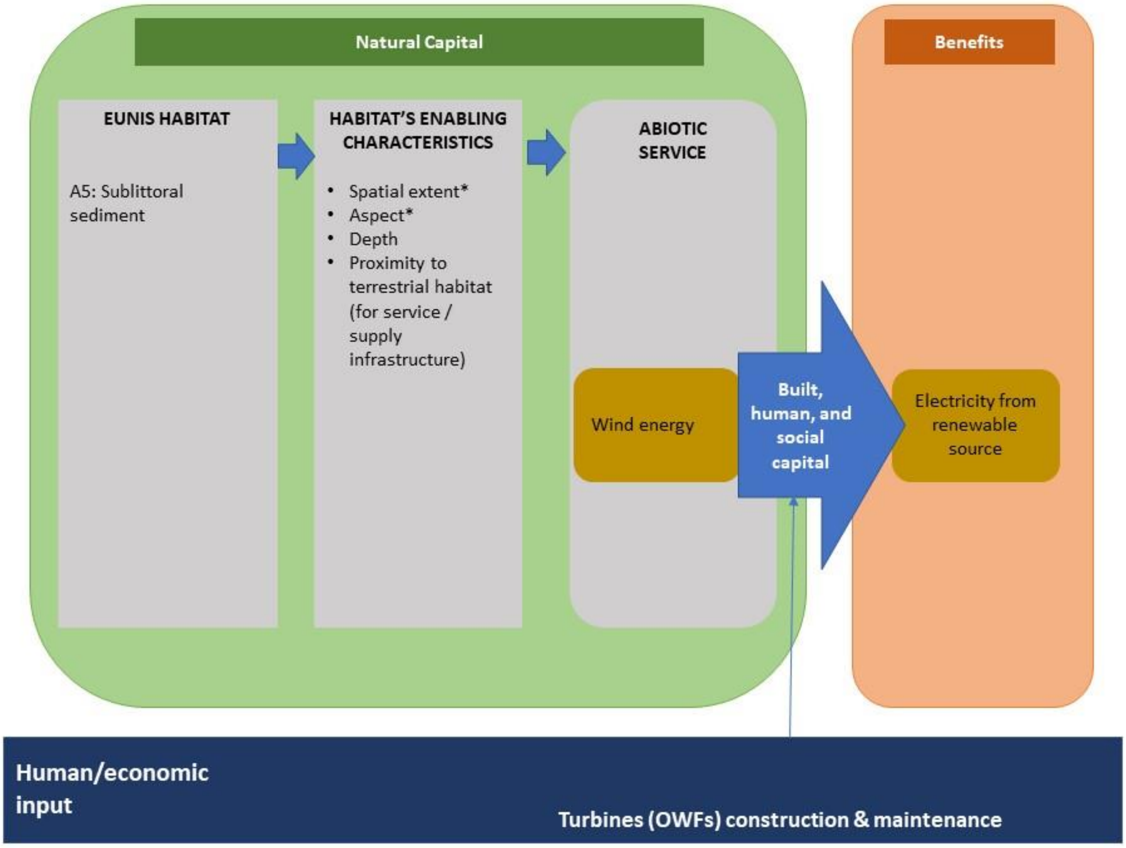

Figure 7. Logic chain for wind energy
* denotes those enabling characteristics deemed most important for the delivery of the abiotic service of wind energy.

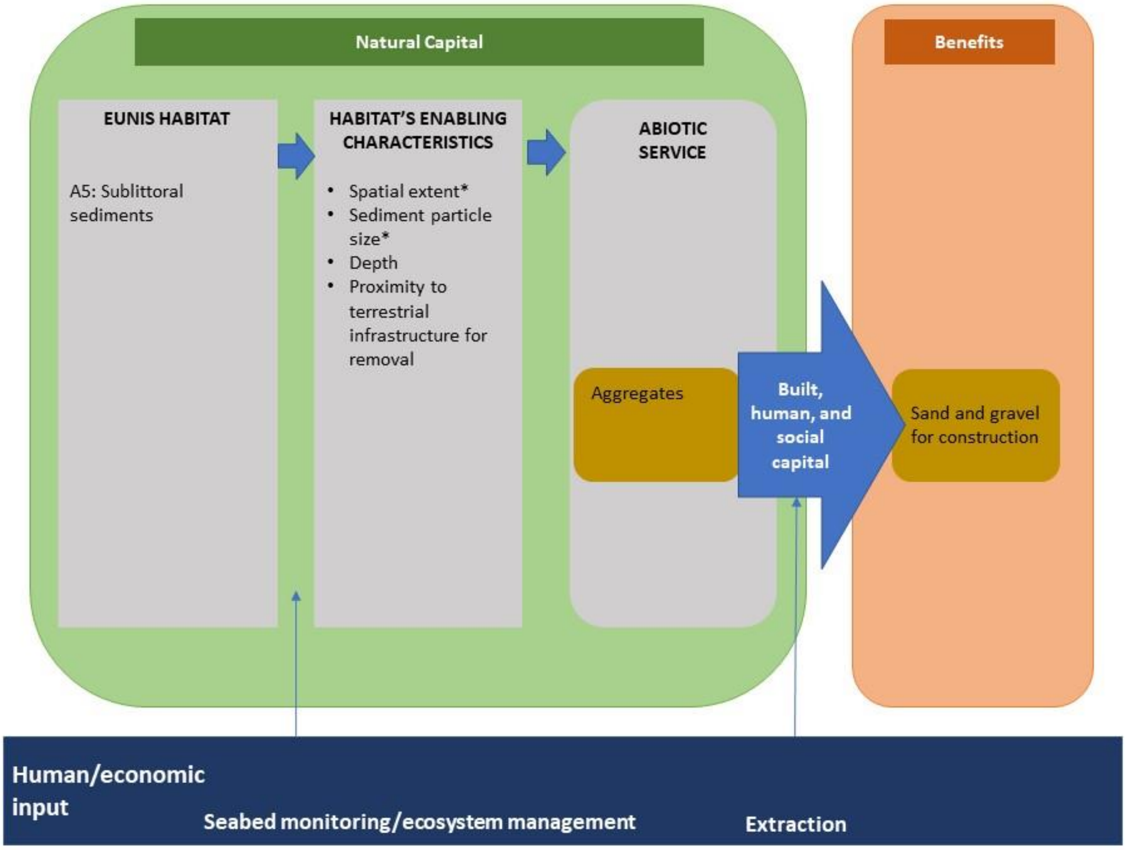

Figure 8. Logic chain for aggregates
* denotes those enabling characteristics deemed most important for the delivery of the abiotic service of aggregates.

Table 4. Indicators of delivery of the ecosystem service waste (nutrient) remediation

Habitat	**EUNIS L2/L3**	**Primary and secondary indicators (metric)**	**Metric relevant to proportion of potential ecosystem service delivered**				**Total habitat extent (ha)**
			>75%	50–74%	25–49%	0–24%	
Littoral sediments[a]	A2.1-A2.4, A2.6, A2.7	Biomass and extent of green macroalgal mats. Assessed using the Opportunistic Macroalgae Blooming Tool (see Section 2.3.1.1 for further details and references).	0.8 Algal cover <5% low biomass	0.6 Algal cover <15% biomass (<500 g m^2)	0.4 Algal cover >15% and biomass >500 g m^2	0.2 High coverage	311,189
Saltmarsh	A2.5	Extent	NA	NA	NA	NA	52,832
Saltmarsh	A2.5	Dominant plant species and aboveground biomass.	NA	NA	NA	NA	52,832
Sublittoral sediment	A5	Substrate type and hydrological processing.	NA		NA	NA	50,374,060
Deep-sea mixed, sand, muddy sand, mud	A6.2–A6.5	Substrate type and hydrological processing	NA		NA	NA	2,887,260
Coastal dunes and sandy shores	B1	Extent and dynamic capacity.	NA	NA	NA	NA	96,518
Coastal dunes and sandy shores	B1	DAFOR (Dominant, Abundant, Frequent, Occasional, Rare) scale for abundance of nutrient enrichment indicator species (*Urtica dioica, Cirsium* sp.)	Never >Frequent or all <5% coverage.	Frequent to Abundant or all <10% coverage.	Abundant - Dominant or all <50% coverage.	Dominant or all >50% coverage.	96,518

[a] Littoral coarse sediment; littoral sand and muddy sand; and littoral mixed sediments; littoral sediments dominated by aquatic angiosperms; littoral biogenic reefs

Note: NA – Data not available (currently)

Table 5. Indicators of delivery of the ecosystem service natural hazard protection

Habitat	EUNIS L2/L3	Key indicators (metrics) for ecosystem service delivery.	Metric relevant to proportion of potential ecosystem service delivered				Total extent of habitat (ha)
			>75% Excellent	50–74% Good	25–49% Adequate	0–24% Poor	
Littoral rock and other hard substrata	A1	Extent and aspect	NA	NA	NA	NA	21,656
Littoral sediments[a]	A2.1, A2.2, A2.4, A2.6, A2.7	Distance wave travels	>200 m	101-200 m	50-100 m	<50 m	210,886
Littoral mud	A2.3	Distance wave travels e.g. Tillingham 35.25% over 147 m (0.14%/m).	>150 m	101–49 m	50–100 m	<50 m	100,303
Coastal saltmarsh and saline reedbed	A2.5	1. Distance (m) from land (width) 2. Distance wave travels Tillingham 87.37% over 163 m (0.54% m), Bridgewick 43.81% over 10 m (4.38% m)	>200 m 63% wave height reduction	51–200 m most wave energy dissipated in first 10–50 m	10–50 m 10–20% wave reduction over 40 m saltmarsh	<10 m	52,832
Coastal saltmarsh and saline reedbed	A2.5	Spatial heterogeneity and proximity to channels. NB modelled data	<50% patchy	<60% patchy	<75% patchy	>75% patchy	52,832
Coastal dunes	B1	Extent of habitat.	NA	NA	NA	NA	96,518
Coastal dunes	B1	Dune colonised by mature vegetation	NA	NA	NA	NA	96,518
Coastal shingle	B2	Extent	NA	NA	NA	NA	10,494
Rock cliffs, ledges and shores	B3	Extent and substrate type (potential erosion)	NA	NA	NA	NA	25,542

[a] Littoral coarse sediment; littoral sand and muddy sand; and littoral mixed sediments; littoral sediments dominated by aquatic angiosperms; littoral biogenic reefs

Note: NA - data not available (currently).

Table 6. Indicators by EUNIS habitat and ecosystem service provision (cf. Table 7 for indicator definitions)

EUNIS Habitat			Ecosystem Service						
			Provisioning	Regulating			Cultural	Abiotic	
Habitat category	**Habitat name**	**Habitat area (ha) and proportion of overall area (%)**	**Finfish and shellfish**	**Waste (nutrient) remediation**	**Natural hazard protection**	**Climate regulation (carbon)**	**Recreational places and seascapes**	**Wind energy**	**Aggregates**
A1	Littoral rock and other hard substrata	21,656ha 0.03%	-	-	2	-	2	-	-
A2.1	Littoral coarse sediment	7,248ha 0.01%	2 8	1 2	2 4	1 2 5	2	-	-
A2.2	Littoral sand and muddy sand	18,7831ha 0.22%	2 8	1 2	2 4	1 2 5	2	-	-
A2.3	Littoral mud	100,303ha 0.12%	2 8	1 2	2 4	1 2 5	2	-	-

Habitat category	Habitat name	Habitat area (ha) and proportion of overall area (%)	Finfish and shellfish	Waste (nutrient) remediation	Natural Hazard protection	Climate regulation (carbon)	Recreational places and seascapes	Wind energy	Aggregates
A2.4, A2.6, A2.7	Littoral mixed sediments, Littoral sediments dominated by aquatic angiosperms, Littoral biogenic reefs	15,807ha 0.02%	2 8	1 2	2 4	1 2 5	2	-	-
A2.5	Coastal saltmarshes and saline reedbeds	52,832ha 0.06%	-	2 3 4	2 3 4	1 2 3	2	-	-
A3	Infralittoral rock and other hard substrata	292,127ha 0.34%	2 8	-	-	-	2	-	-
A4	Circalittoral rock and other hard substrata	49,1616ha 0.57%	2 8	-	-	-	2	-	-

Habitat category	Habitat name	Habitat area (ha) and proportion of overall area (%)	Finfish and shellfish	Waste (nutrient) remediation	Natural Hazard protection	Climate regulation (carbon)	Recreational places and seascapes	Wind energy	Aggregates
A5.1	Sublittoral coarse sediment	16,497,908ha 19.23%	2 8	2 5	-	2 5 7	2	2 6	2 5
A5.2	Sublittoral sand	26,484,814ha 30.87%	2 8	2 5	-	2 5 7	2	2 6	2 5
A5.3	Sublittoral mud	6,149,456ha 7.17%	2 8	2 5	-	2 5 7	2	2 6	2 5
A5.4	Sublittoral mixed sediments	1,241,882ha 1.45%	2 8	2 5	-	2 5 7	2	2 6	2 5
A6.1	Deep-sea rock and artificial hard substrata	633,871ha 0.74%	2 8	-	-	-	2	-	-

Habitat category	Habitat name	Habitat area (ha) and proportion of overall area (%)	Finfish and shellfish	Waste (nutrient) remediation	Natural Hazard protection	Climate regulation (carbon)	Recreational places and seascapes	Wind energy	Aggregates
A6.2-A6.5	Deep-sea mixed, sand, muddy sand, mud	2,887,260ha 3.37%	2 8	2 5	-	2 5 7	2	-	-
B1	Coastal dunes and sandy shores	96,518ha 0.11%	-	2 3	2 3	2 3	2	-	-
B2	Coastal shingle	10,494ha 0.01%	-	-	2	-	2	-	-
B3	Rock cliffs, ledges and shores, including the supralittoral	25,542ha 0.03%	-	-	2	-	2	-	-
	Seabed and ‘known unknown’ habitat	30,735,389ha 35.82%							
	Total habitat	85,800,000ha							

Having defined the information needed to assess the current capacity of the broad scale marine and coastal habitats to deliver key ecosystem services, it remained to be seen if the necessary information was available. Following an examination of published literature, various previous assessments and accessible databases, we were able to develop Table 7 identifying suitable data sources to support the proposed indicators in Table 6.

Table 7. Indicator and corresponding data source for indicators shown in Table 6

Indicator	Indicator detail	Indicator data source	Responsible agency
1	Primary production (littoral habitats) measured as biomass and extent of macroalgae	WFD	EA, SEPA, NIEA, NRW
2	Extent of habitat measured at appropriate habitat scale	UKSeaMap and Combined Map (JNCC 2017; 2019)	JNCC
3	Habitat surveys (coastal and littoral habitats)	Defined by reporting tool (e.g. CSM, CS)	JNCC, NE, NRW, SNH, NIEA, EA, CEH
4	Distance from land, length of habitat, habitat fragmentation, measured using existing maps with additional earth observation supported by ground-truthing.	UKSeaMap and Combined Map (JNCC 2017; 2019)	JNCC
5	Sediment type	WFD (littoral habitats) MSFD GES descriptor 6 (JNCC 2010) BGS	EA, SEPA, NIEA, NRW JNCC BGS
6	Aspect / wind direction	Data collected by UKHO, MMO	UKHO, MMO
7	Primary production (marine habitats excl. littoral sediment)	MSFD GES descriptors 4, 5. (JNCC 2010)	JNCC
8	Functioning marine food web	MSFD GES descriptor 4 (JNCC 2010)	JNCC

Note: Indicators correspond to key habitat enabling characteristics identified (*) in logic chains (Figures 2 - 8).

Much of the information needed is largely unavailable at the scale of the broad scale habitats as defined herein. Given the paucity of information regarding condition of specific habitats in relation to their ability to deliver the service identified, we have noted this as an area of future development and recommend the use of extent of suitable habitat as the relevant variable to determine the expected future delivery of the identified ecosystem services over time.

Of the seven ecosystem services considered, five have benefitted previously from considerable research and thus only two, namely waste (nutrient) remediation and natural hazard protection, were selected for in-depth assessment of enabling characteristics and critical indicators. An explanation of the development of suitable indicators for these two ecosystem services is described in Section 2.3

As noted above the habitats identified do not exist in isolation and have complex relationships with neighbouring habitats. Contiguous land-use or physical infrastructure can affect the ability of identified habitats to deliver various ecosystem services. For example, if an area of saltmarsh is located next to a seawall, then its ability to mediate a high water or storm event may vary (see Section 2.3.2.3). The issue of context and its effect on the sustainable delivery of ecosystem services from a given habitat is an area for further study, especially pressing for coastal margins facing increasing population pressure and climate change impacts.

In the assessment of finfish and shellfish as a provisioning service, given that mobile finfish species benefit from a number of marine habitats, the applicable condition reflecting the ability of habitats to provide this ecosystem service was extent and general ecological condition, according to MSFD reporting, good overall. Climate regulation capacity, specifically carbon sequestration, was assessed by extent of habitat with suitable sediment type and primary production.

The UK mainland alone has approximately 17,820 km of coastline which can be enjoyed by the populace, including a vast area available to the boating community. Numerous small islands offer additional recreational opportunities with more coastline and unique communities. Conditions of water quality have improved considerably since the application of the Water Framework Directive restrictions and efforts to tackle waste and plastic pollution. As overall conditions for recreation were deemed good no restriction was applied and extent of habitat overall was relevant.

Conditions for aggregate extraction and wind turbine installation were based on the extent of habitats suitable for those activities, additional information regarding site conditions was likely to be available to commercial operators but not to the writers of this report. Licensing required for this activity provided a non-biophysical limit on the applicable area for ecosystem service delivery. The extent of area suitable for providing the two abiotic services considered – wind energy and aggregates – was defined by the relevant licensing authorities which consider similar criteria in determining exploitable sites.

4 Initial natural capital services accounts for UK marine and coastal ecosystems

In this section we examine valuation methodology options for the services reported in Tables 3a and 3b drawing upon existing valuation work including for the Natural Capital Accounting Roadmap, considering the service account structures, relevant classifications and physical assets assessed previously. We also assess the data needs and sources, including availability (present and future) and accessibility, together with non-monetary information, at both national and sub-national levels. The importance and potential of using spatially disaggregated data are also considered.

In order to fill in a set of initial marine economic accounts, we need to define the valuation methodology adopted for each ecosystem service selected (see Table 3a). This implies identifying an appropriate valuation method for each selected service, together with an investigation of the related biophysical and economic data needs. Biophysical data in fact are not restricted to the extent and condition of the assets that provide that specific service. There may be a need to integrate extent and condition data with other relevant biophysical data to identify the relevant physical flows which apply to an appropriate unit value. For example, for valuing the benefit of a healthy climate, biophysical data related to the ecosystem service of carbon (C) sequestration and storage, both the extent and condition of UK saltmarshes, and the C storage rate by saltmarshes have to be available. In terms of economic data, for monetary accounts, exchange values (those values related to the market) are preferred. So, to stay with the healthy climate example, the abatement cost (clean-up cost based on the cost of replacing the service by other means) is preferred to the social cost of carbon as a value for carbon.

The marine environment physical accounts will form the basis for calculations for the monetary accounts. We provide information on the extent of the different habitats and sub-habitats considered indicating the extent of the stocks, especially relative to their ability to provide the ecosystem services of interests (see Tables 4, 5 and 6). We report in detail the biophysical and economic data used for the calculation of each ecosystem service defined in Tables 3a and 3b, as well as the results reported in the accounting tables, in the following sub-sections.

Once we defined the asset under investigation and the sub-habitats involved (Table 1), we decided which ecosystem services and benefits on which to focus based on biophysical and economic data availability and research priority.

In the following sub-sections, we consider, for each ecosystem service:

- the preferred and alternative valuation methodologies in line with natural capital accounts;
- the logic chain applied, which includes the supply of services by different ecosystem types;
- an assessment of data needs and potential data sources available;
- the actual data available and the applied valuation method; and
- the biophysical and economic data that will feed into the physical and monetary accounting tables.

Based on literature search and expert consultation, we assessed the ideal set of biophysical and economic data needs and potential data sources available by ecosystem service depending on the valuation method selected. We have developed logic chains, which are, in this report, simple models linking the ecosystem or habitat, the features and characteristics of the habitat enabling it to provide the relevant ecosystem service, and the welfare benefit. In our logic chain diagrams (Figs. 2 – 8), we show the human/economic input that, in conjunction with the natural capital, provides the benefits to society. More complex logic chains, outside the scope of this study, would include: the economic activities related to the benefit provided and the impacts of these on habitats, the positive effects of human interventions and management, and the users of the service. These could potentially highlight the effects of climate change, which may play a role in the projection of future services flows but also help us to understand past changes revealed by the time series of the accounts. We considered the valuation options available for a specific benefit and then chose the preferred option based on good practice, whenever possible, or data availability. For this choice, issues of spatially disaggregated data have played an important role. At the biophysical level, the use of EUNIS tables, for example, has not always been possible for the assessment of all the ecosystem services (see waste (nutrient) remediation Section 2.3), or some sub-habitats definitions have been considered too vague to provide precise information. Economic data are not always transferable from the sub-national or regional level making their aggregation not possible in many cases (see also Luisetti et al. 2014).

4.1 Methodology for developing physical and monetary accounts

In this section we illustrate the methodology followed to obtain the accounts for marine and coastal habitats by ecosystem service, explain the methodology/data and calculations for the physical accounts (Table 19), and the calculations used for the monetary accounts (Table 20).

4.1.1 Fish and shellfish, renewable energy and aggregate extraction – the resource rent approach.

For certain environmental assets for which extraction of the resource is undertaken, direct payments from the extractors to government are often required in the form of levies or royalties. These payments can be treated as a rent for the environmental resources, that is, a value for its extraction similar to the exchange value concept required in the accounts (e.g., stumpage price for timber). In practice, it can be difficult to isolate this rent from other taxes and levies paid by the extractors, therefore the resource rent has to be calculated from the data available, which in most cases are derived from the System of National Accounts (SNA) or from the financial documents of companies operating in a specific industrial sector. Following the guidance in SEAA Central Framework (UN, 2014) and Defra / ONS (2016), provisioning ecosystem services are here valued using a residual value resource rent approach. In this report, following best practice, abiotic services are also valued using the resource rent approach. Resource rent provides a gross measure of the return on the environmental asset. In other words, it aims at isolating the surplus value added to the marketed output from the environmental asset, after considering other operational costs and normal returns. This approach to the calculation of the resource rent is acknowledged to produce low estimates of monetary contribution that environmental assets and ecosystem services provide to the national economy. Nevertheless, it is considered suitable for this specific application and, generally, it has been advised as an appropriate methodology in several previous natural capital and ecosystem services accounting guidelines (Defra / ONS 2016).

The resource rent can be obtained by using a residual value approach. The value of the resource rent then is derived through assessment of the relationships between the relevant variables shown in Table 8 (ONS 2017; UN et al. 2014).

Table 8. Data needs for deriving the resource rent using a residual value approach

	Output or Turnover (Sales extracted environmental assets at basic prices)
Less	Operating costs
Less	Intermediate consumption
Less	Compensation of employees
Less	Other taxes on production
Plus	Other subsidies on production
Equals	**Gross operating surplus – SNA basis**
Less	Specific subsidies on extraction
Plus	Specific taxes on extraction
Equals	**Gross operating surplus – resource rent derivation**
Less	User costs of produced assets (consumption of fixed capital + return to produced assets)
Equals	**Resource rent**

Source: ONS 2017; UN et al. 2014

The data needed for the calculation of the resource rent should ideally be extracted from annual financial statements published by private companies operating in the relevant industrial sector. This would allow the isolation with increased detail of the contribution of natural assets to the production process. However, often these documents are not available or not disclosed by the relevant industries or companies. Therefore, as an alternative, the resource rent is usually derived from the SNA using data within the Input-Output tables, with the advantage of reliable yearly data and the disadvantage of lower detail due to industrial sector aggregation.

In particular, data on output, operating costs, intermediate consumption, compensation of employees, other taxes and subsidies on production, and gross operating surplus can be directly derived from the financial documents published by private companies or from the SNA Input-Output tables related to relevant industrial sector classification. Where possible, data regarding taxes and subsidies on extraction can be derived from the relevant industrial sector financial documentation. The ecosystem services user costs should be also derived for the specific industrial sector from relevant financial documentation. If not possible, the consumption of fixed capital can be obtained from the capital accounts available in the SNA, and the return to produced assets by using the British Government Securities 10-year Nominal Par Yield[4] (Bank of England 2019).

[4] The annual rate of interest which, if used to discount future dividends and the sum due at redemption, makes the price of a 10-year government security equal to the nominal (par) value. For more information see https://www.bankofengland.co.uk/statistics/details and https://www.bankofengland.co.uk/working-paper/2001/new-estimates-of-the-uk-real-and-nominal-yield-curves.

Resource rent, RR, is then obtained as:

$$RR_t = GOS_t - (CFC_t + RA_t)$$

$$GOS_t = OUT_t - (OC_t + IC_t + EC_t + NetTAX_t)$$

where, GOS_t is the gross operating surplus at time t (directly obtainable from the SNA adjusted for specific taxes and subsidies on extraction) calculated as the total output sales (OUT_t) less the costs to produce the output, that is, operating costs (OC_t), intermediate consumption costs (IC_t), employment costs (EC_t), and taxation ($NetTAX_t$); CFC_t is the consumption of fixed capital and RA_t is the return on produced assets.

The procedure outlined has been adapted for the provisioning services, biotic and abiotic, valuation to account for specificities related to each service. Details of the adjustments used for the calculations are described in the following sub-sections for finfish and shellfish, offshore wind energy (OWFs), and aggregates extraction.

4.1.2 Fish and shellfish

Figure 2 shows the logic chain applied to wild finfish and shellfish capture. The provision of fish and shellfish is a relevant ecosystem service provided by different coastal and marine environment habitats, and their interactions. It is estimated that in the 2017 a total of 581 thousand tonnes of sea fish and shellfish caught in the UK Exclusive Economic Zone (EEZ) was landed in the UK and abroad by UK vessels (Williamson et al. 2018).

With reference to EUNIS Table 1, the habitats mainly providing the service are A1 to A6, comprising the majority of coastal and marine habitats. Indeed, the provision of finfish and shellfish is the result of complex interactions spanning the whole life-cycle of many species, regarding different biophysical processes (primary production, larval and gamete supply, spawning, nutrient cycling, etc.). As represented in the logic chain, there are many factors influencing the provision of wild finfish and shellfish, an ecosystem service for which the interaction between environmental and human management aspects is of utmost importance. Indeed, if on the one hand habitat conditions and biophysical characteristics and processes assure the basic prerequisite for sustaining finfish and shellfish production, sustainable human management practices (e.g. maximum sustainable yield and marine protected areas) are needed to mitigate the many impacts related to human pressures affecting finfish and shellfish provision (fishing itself, coastal development, shipping, waste disposal, etc.).

The physical supply of finfish and shellfish is quantified using marine finfish and shellfish catch data recorded by the International Council for the Exploration of the Sea (ICES 2019b). Catch statistics are presented as nominal catches in tonnes live weight. Catch figures cover data from all commercial, artisanal, and subsistence fisheries, and include recreational catches, where available. The statistics do not include non-recorded catch and discards data. In this report, catch data are reported for the UK fleet and considering only the catch within the UK EEZ (see Figure 9). Indeed, only the ICES statistical rectangles (ICES 2019a) overlapping with the EEZ have been considered as shown in Figure 9. To derive the resource rent, data regarding the landings of marine finfish and shellfish from the UK fleet in the UK and abroad as collected by the Marine Management Organisation (MMO) (2013-2017) are used. Landings data are related to the UK commercial fleet only. Landings data considered in this report include quantity landed in the UK and in foreign ports by the UK fleet, as landing in both sites directly contribute to national economy. Further research and supra-national effort are required to harmonise the method used to assess the monetary value of fisheries in natural capital accounting in order to avoid double counting at the macro-level. It is indeed necessary to interpret methodological issues in the light of international fishing policies and fishing quotas mechanism, in addition to a better use of stock assessment data.

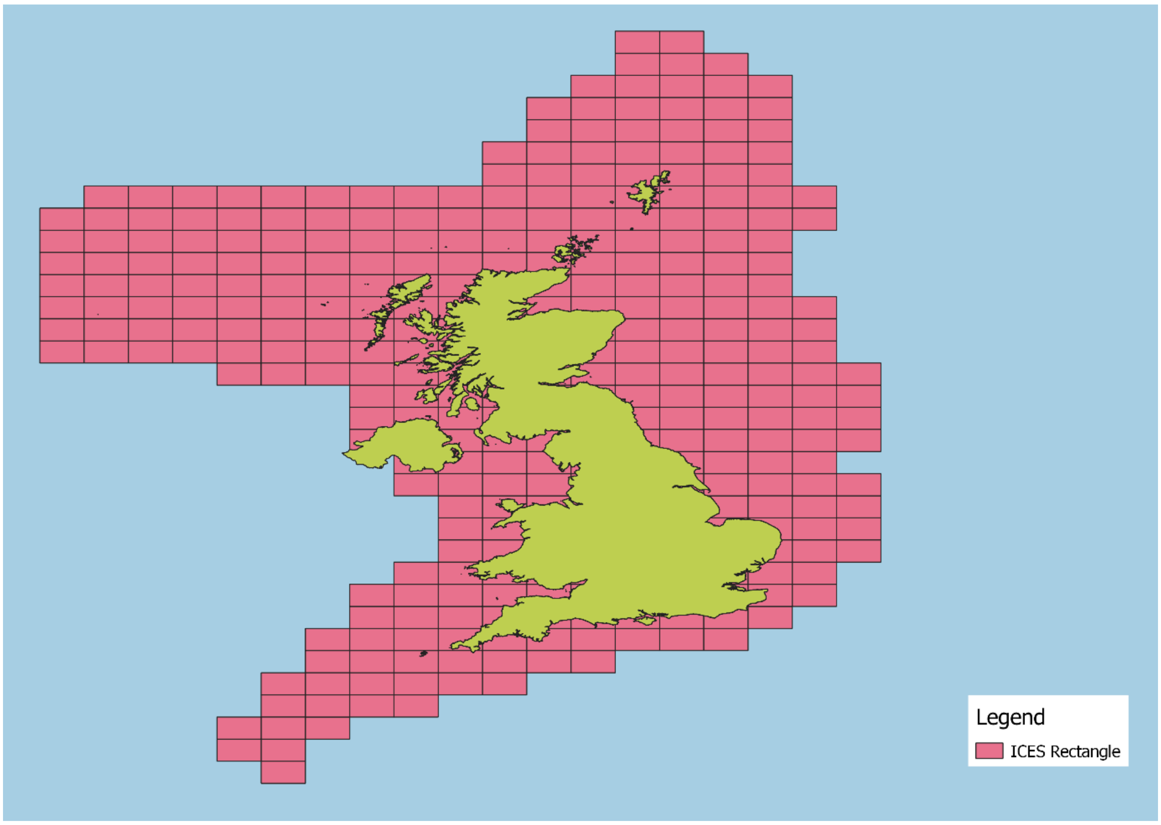

Figure 9. Area of UK marine habitat: ICES rectangles considered for catch and landings data (ICES 2019a)

Figure 10 shows the total quantity of finfish and shellfish catches and landings for the UK, between 2012 and 2017. The total catch has increased of 16% between 2012 and 2017, resulting in an increase in the quantity landed of 19% between 2012 and 2016. Figure 10 also shows a sharp increase in catches and landings in 2014, equal to 19% and 17% higher than previous year respectively, followed by a marked decline in catches and landings for 2015, both 7% lower than the previous year.

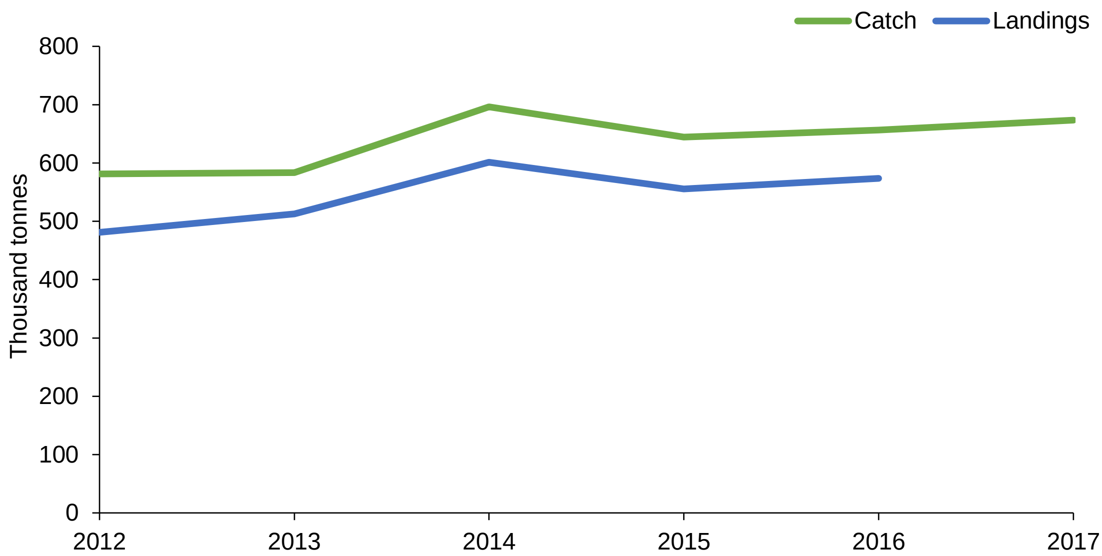

Figure 10. Finfish and shellfish catch and landings from UK marine waters, 2012-2017 (ICES 2019b; MMO 2013-2017)

A resource rent is obtained for the whole “fisheries and aquaculture” sector in the SNA (SIC 03) and a resource rent to output ratio is calculated, which is then applied to the total landings value considered as the total output sales of the sector. The calculation of the resource rent is obtained directly using the Gross Operating Surplus recorded in the SNA, subtracting the ecosystem services user costs. Data on the fixed capital consumption and net produced fixed capital are derived from the Capital Accounts tables for the whole sector “fisheries and aquaculture”. To derive the return to produced asset, the British Government Securities 10-year Nominal Par Yield is applied to the net produced fixed capital in each relevant year. The 10-year Nominal Par Yield is converted in real terms using the GDP deflator (Bank of England 2019).

The total value of marine finfish and shellfish provision in the UK waters was of £146.6 million in the 2016, with an increase of 19% on the 2012 value and an annual average increase of 11% in the same period (See Table 9). The trend of the values presented in Table 9 depends on two main factors. The first factor relates to the trend of total value of landings during the accounting period. In Table 9, the trend of value of landings partially explains the difference between 2015 and 2016. Between those two years, indeed, landings value increased of £156.5 million (+24.5%, in 2017 prices). The second factor concerns the data employed in resource rent calculations, in particular the trend of the gross operating surplus from SNA tables. In Table 9, the trend of resource rent values particularly impacts the difference between 2012 and 2013 and between 2015 and 2016, when, respectively, a decrease and an increase of the gross operating surplus are reported.

Table 9. Value of finfish and shellfish provision from UK marine waters, 2012-2016 (£million, 2017 prices)

	2012	2013	2014	2015	2016
Finfish and shellfish	122.8	89.7	97.3	81.4	146.6

Data used in this section are collected by national and international fisheries policy organisations and therefore can be considered to have the highest confidence level and precision available (ICES 2019b; MMO 2013-2017). Data limitations concern the information employed in the resource rent calculation relating to the whole “fisheries and aquaculture” sector. More detailed information enabling the isolation of the contribution of the wild fisheries sector would allow us to obtain more precise estimates. On this point, it is worth noting that resource rent is considered a suitable approach to be used in this initial account, but further research is needed on improving accounting approaches for finfish and shellfish provision. Indeed, a better approach would simultaneously model links between biophysical (e.g. stock assessment, nursery grounds, food web analysis, environmental pressures), institutional (e.g. fishing quotas, supra-national governance) and economic (e.g. fishing effort and pressure, industrial costs) elements. Finally, recreational fishing is included in catch data (where data are available) but not landings data. Data on discharges by ICES rectangle are not readily available.

4.1.3 Waste (nutrient) remediation

Waste remediation (breakdown, detoxification and burial/removal/neutralisation) is an important service for the health of the marine environment and the provision of many ecosystem services. Different coastal and marine ecosystems work together to provide waste (nutrient) remediation ecosystem service. Remediation capacity thresholds vary for each habitat depending upon the particular pollutant or nutrient being processed. The logic chain for this service is presented in Figure 3.

In this report, wastewater discharged from urban wastewater treatment plants (UWWTP) is used as a proxy for the provision of pollutant and nutrient bioremediation service by coastal and marine habitats in the UK. As a working assumption, we consider the UK coastal and marine environment capable to meet all the demand for the service. In other words, we are assuming that the demand for the effluent from sewage treatment being remediated is totally met by the coastal and marine environment supply of waste remediation services. In practice, not all demand will be met resulting in a deterioration in condition of the asset which would be recorded in the condition accounts. The information used to assess the service, in both physical and monetary terms, relates to the quantity of pollutants and nutrients discharged by UWWTPs in coastal and marine waters in the UK, and the value for treating a unit of the same pollutant or nutrient. Concerning the former, data collected for the EU Urban Waste Water Treatment Directive (UWWTD) are compared to data collected from the Irish Environmental Protection Agency (EPA) in order to estimate the quantity that UK UWWTPs discharge in coastal and marine waters. Concerning the latter, the shadow price of treating a unit of pollutant reported in Hernandez-Sancho et al. (2010) is used, which is based on the cost avoided for providing the same treatment (See Table 10). Only N, P and biochemical oxygen demand (BOD which measures the quantity of organic compounds in water) are considered in this report.

Table 10. Cost avoided for the treatment of selected pollutants (2017 prices)

Pollutant	Avoided cost (£/kg)
Nitrogen	23.54
Phosphorus	71.29
Biochemical Oxygen Demand (BOD)	0.05

Source: adapted from Hernandez-Sanchez et al. 2010
Note: all values expressed in 2017 real terms using GDP deflator published by ONS (2019).

The EU UWWTD reports the list of 325 UK UWWTPs that are licensed to directly discharge treated wastewater in coastal and marine waters, which are shown in Figure 11. Of those, 209 discharge into estuarine and transitional waters and 116 into coastal and marine waters. For each of the UWWTPs considered, the UWWTD reports the wastewater loading that enters the plant (expressed in population equivalent units), but not the corresponding quantity of nitrogen, phosphorus and BOD discharged. In order to estimate those quantities for the UK, we consider the data for Ireland as a proxy, due to the similarities in wastewater treatment management and the geographical proximity between the two countries. In particular:

- Wastewater loading that enters the Irish UWWTPs close to the coast and licensed to directly discharge in estuarine and coastal waters recorded in the UWWTD for 2015; and
- Quantity of N, P and BOD discharged from the same plants recorded by EPA for 2015

Regression analysis is then employed to estimate for the Irish UWWTPs the average relation between loading entering in and nutrients discharged by the UWWTPs, controlling for other characteristics of the UWWTPs such as the water body type and the treatment type. The values obtained are then transferred to the UK UWWTPs. The linear regression analysis can be formalised as:

$$Dis_i = \beta_1 * Load_i + \beta_2 * Control_i + \varepsilon_i$$

Coefficients (β_1) expressing the relationship between loading that enters the UWWTPs ($Load_i$) and the quantity discharged (Dis_i) are summarised in Table 11.

Table 11. Summary of the linear regression analysis

Discharge (Kg/year)	Coefficient (S.D)	R-squared	N
Nitrogen	1.51** (0.024)	0.98	56
Phosphorus	0.29** (0.004)	0.98	56
BOD	1.82** (0.035)	0.93	56

** statistically significant (p = <0.01).

Therefore, for each population equivalent unit of wastewater[5] entering the UWTTPs, discharges are on average 1.51 kg/year of N, 0.29 kg/year of P, and 1.82 kg/year of BOD.

[5] The unit of measure used to describe the size of a wastewater discharge. A population equivalent unit (i.e. 1 PE) is the biodegradable load matter in wastewater having a five-day biochemical oxygen demand (BOD) of 60g of oxygen per day. Population equivalent doesn't necessarily reflect the actual population of a community.

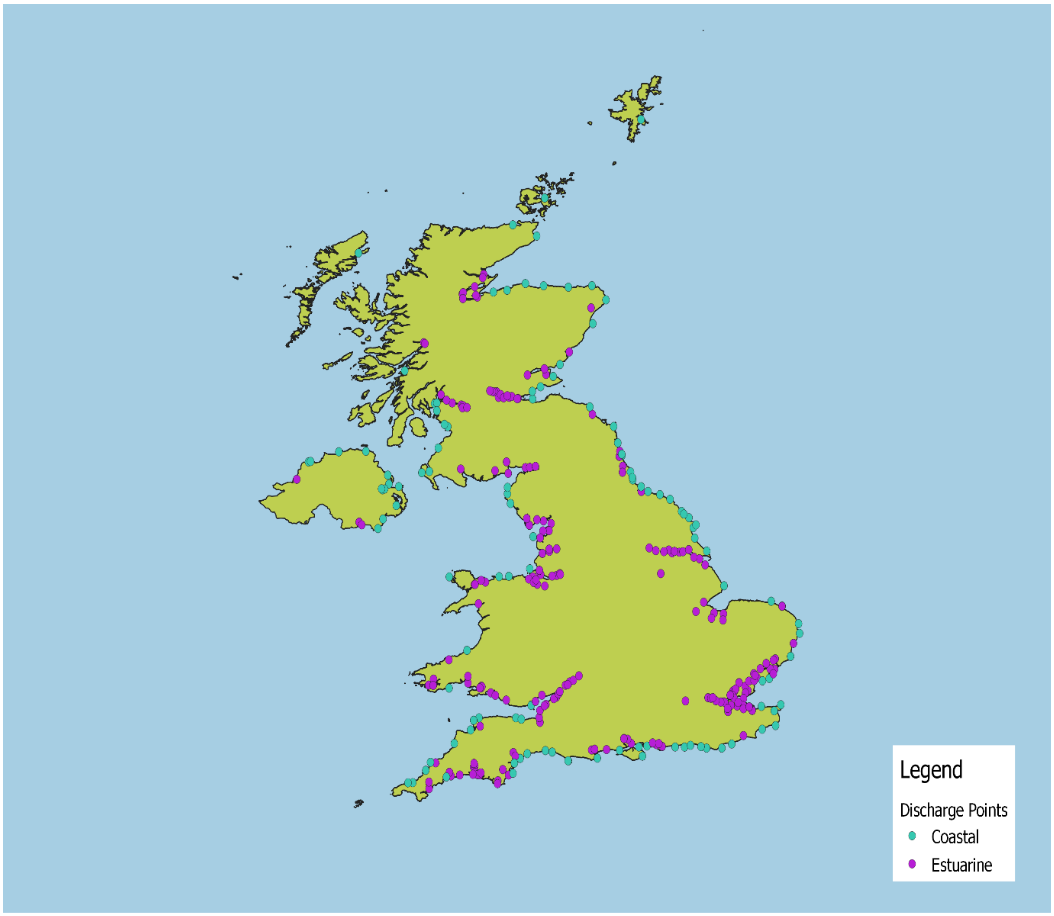

Figure 11. Urban wastewater treatment plants discharge points in UK marine and coastal waters
Source Cefas adapted from EEA (2019b)

The estimated values of the coefficients shown in Table 12 are transferred to 2015 UK data on wastewater loading that enters the UWWTPs discharging in estuarine and coastal waters as recorded for the EU UWWTD. Results regarding the total quantity of pollutants discharged and the monetary valuation are summarised in Table 12.

Table 12. Valuation of wastewater discharge in UK marine and coastal habitats (2017 prices)

	Nitrogen	Phosphorus	BOD
Estimated discharge UK (t/year)	45,666.9	8,614.0	54,816.9
Shadow price (£/kg)	23.54	71.29	0.05
Total value (£mil/year)	1,074.99	614.09	2.74

Note: all values expressed in 2017 real terms using GDP deflator published by ONS (2019).
Therefore, the total monetary valuation of wastewater discharged in the UK coastal and marine habitats is estimated to be £1,69billion/year (reference year is 2015 and value expressed in 2017 prices).

This method is likely to underestimate the actual value of the waste and nutrient remediation service provided by the UK coastal and marine environment, as only some of the elements discharged and only some of the pollution sources are accounted for (e.g. non-point sources are excluded). Finally, the availability of data on pollutants discharges specifically for UK UWWTPs would have permitted more accurate estimations to be calculated.

4.1.4 Natural hazard protection

Figure 4 shows the logic chain for the ecosystem service of natural hazard protection provided by UK coastal and marine habitats. The ecosystem service of natural hazard protection relates to the moderating effect that coastal habitats have on natural hazards, such as storm surges and coastal flooding, thus diminishing the risk to human life and economic resources. Different marine and coastal habitats can contribute differently to natural hazard protection depending largely on location.

The initial difficultly when assessing and valuing natural hazard protection is whether to disaggregate the different services included, e.g. flood protection, wave and tidal dissipation, storm protection, etc, or consider them as a whole. A further difficulty stems from assessing the different degree of protection provided by different coastal habitats, therefore identifying the most suitable indicators to use. Other than conceptual difficulties, a precise assessment should be grounded on complex bio-economic modelling, including several characteristics of the coastal and marine environment coupled with social and economic attributes. To provide a monetary value for this regulating ecosystem service, three methods are usually employed. A replacement cost approach is generally considered more in line with natural capital accounting requirements, as it values the cost of replacing the service provided by natural habitats with man-made equivalent protection. Other possible valuation options, considered less suitable for natural capital accounting as more akin to welfare values approach, are the damage-cost-avoided approach, consisting of valuing the cost of damage avoided due to the presence of the natural habitat protecting the coast, and the value-transfer-approach, consisting of using economic values from other studies of similar context. If the former of the two could require modelling the vulnerability of certain habitats to natural events and data on economic and social activities existing on the land protected, the latter is limited by the difficulty of finding strictly comparable study sties. All three alternative valuation approaches are, in principle, viable options for natural capital accounting.

Due to the lack of more suitable data, in this report a simplified version of the replacement cost method is employed (Norton et al. 2014). Only saltmarshes are considered in assessing the provision of protection from recurrent (e.g. waves, tides) and infrequent (e.g. storms, floods) natural disturbances. The main information used to calculate a monetary value for the service relates to: (i) the minimum width of saltmarsh needed to generate excellent provision of coastal protection ecosystem service, estimated to be 200 m (see Table 5); (ii)) the annual capital cost of building a seawall, estimated to be £2,116.30 per metre (in 2017 prices) by the Environmental Agency (EA, 2007); and (iii) the extent of UK saltmarshes from Table 2. It is assumed that the considered capital cost of building a seawall would be incurred if the natural defence provided by saltmarshes did not exist. This working assumption is required to obtain an exchange value for the service provision. In order to use the EA estimation concerning the capital cost of man-made defence, it is necessary to translate the extent of saltmarsh into a linear measurement. We therefore assume that all the saltmarshes in the UK have at minimum the width necessary to effectively deliver the service as defined in Table 2 (i.e. 200 m). Dividing 1ha (10,000 m^2) by 200 m, the length of the coast protected would be 50 m for every 1 ha of saltmarsh. In Table 13, data and calculations are summarised, and the total value of the ecosystem service natural hazard protection is estimated to be equal to £5.59 billion[6].

[6] The monetary estimates for sea defence service reported in the study *Scoping UK coastal margin ecosystem accounts* (Defra and ONS 2016) are related to saltmarshes in England and equal to £2.53 billion (2014 prices). The estimates are based on the UK NEA (2011) and are derived using the same approach of this report.

Table 13. Simplified replacement cost approach to value natural hazard protection

Cost seawall per m/year EA Agency (2017 prices)	£2,116.30
Extent of UK saltmarshes	52,832 ha
Estimated length of coast protected [a]	2,641,600 m
Total estimated monetary value ecosystem service	£5.59 billion

[a] Assumes all saltmarshes are 200 m wide
Note: all values expressed in 2017 real terms using GDP deflator published by ONS (2019).

This method is likely to overestimate the value of the ecosystem service. Indeed, it is implicitly assumed that all of the saltmarshes provide the same level of natural hazard protection. At the same time, other habitats providing the service are not considered. Also, stating that all the saltmarshes have a minimum width of 200 m is a strong working assumption. Finally, the different value of the economic activities protected is relevant to the value of the service and is not accounted for in this approach.

4.1.5 Climate regulation (carbon sequestration and storage)

Figure 5 shows the logic chain applied to the ecosystem service of climate regulation specifically, carbon sequestration and storage. The important welfare benefit provided by this service is maintaining an equitable climate which facilitates the existence of life on this planet as we know it. The SEEA EEA (2012) describes accounting for both the storage and sequestering of carbon as one of the main challenges of ecosystem services accounting in physical terms, which is still an on-going discussion. The SEEA EEA suggests that the confusion arises because, despite being sequestration and storage of carbon two very different ecosystem services, the expression 'carbon sequestration' often includes both services. In order to account for both stocks and flows of carbon, the SEEA EEA suggests considering the service of carbon sequestration as *'the net accumulation of carbon in an ecosystem due to both growth of the vegetation and accumulation in below-ground carbon reservoirs'*, and the carbon storage service as *'the avoided flow of carbon resulting from maintaining the stock of above- and below-ground sequestered in the ecosystem'*. The main habitats providing this service in the UK, following the EUNIS classification shown in Table 1, are saltmarshes, littoral, sublittoral and deep-sea sediments as shown in the 'Habitats' column and of the logic chain. Seagrasses, and other rooted vegetated marine systems, also provide the service of carbon sequestration and storage in UK waters, but precise data on this are currently lacking. In the case of vegetated systems (e.g., saltmarshes and seagrasses), the plants capture CO_2 from the atmosphere and then provide long term storage of that carbon through the root systems into the sediments. This is sometimes known as 'blue carbon'. It is important to specify that the process valued in monetary terms is sequestration as defined by the SEEA EEA (2012) (see also ONS 2016, p.22). Carbon capture/fixation without carbon burial/accumulation in the sediments does not raise the welfare benefit, the benefit comes with carbon burial when the CO_2 is locked away long-term. Looking closer to the water column and seabed, although carbon can occur in many forms, due to data limitation, for economic valuation and natural accounting purposes, we focused on particulate organic carbon (POC). Within the SEEA EEA definition, saltmarshes and seagrasses are economically valued for their carbon captured in the standing stock and that stored below ground. Therefore, for saltmarshes, we extrapolated and used the combined above- and below-ground carbon sequestration rate from Luisetti et al. (2019), which is equal to 0.86 t/ha/yr. Within the water column, to avoid double counting, we only valued the POC deposited and then stored in the shelf sea sediments using the lowest estimates reported in Table 14. This POC, which includes seaweed/algae fragments and phytoplankton, being a transient store capturing/fixing carbon for a short-term only acts an 'intermediate' service (Fisher and Turner 2008; Fisher et al. 2009). However, the economic value is calculated on the benefit provided (climate regulation), which in this case is carbon sequestration, as defined in the SEEA EEA Technical Recommendations (2012): the accumulation of carbon in carbon sinks which determines a long-term (i.e. at least several decades) sequestration through the removal of CO_2 from the atmosphere. Therefore, since the POC that allows the benefit of the long-term carbon sequestration in shelf sea sediments to happen already includes the parts of seaweed/algae and phytoplankton that provide the carbon capture/fixation (i.e. the short-term flow of exchanges of CO_2), if we were to account separately also for the carbon capture/fixation in the phytoplankton and seaweed than we would end up in double counting the service and overestimating the benefit. In Figure 5, the second column lists the habitat characteristics and the enabling features for the (final) ecosystem service of climate regulation (carbon sequestration and storage) to be provided. On the left of Figure 5, is an example of human pressures that can have an impact on the coastal and marine habitats providing climate regulation. On the other hand, the management of coastal and marine habitats through habitat restoration may significantly improve the provision of this natural climate change mitigation solution.

To calculate the flow of services provided by marine and coastal habitats for climate regulation (carbon sequestration and storage), the following biophysical and economic data are needed:

- Extent of the marine and coastal habitats providing the service;
- Carbon sequestration rate (possibly per tonne of CO_2 equivalent [tCO_2e] /ha/year as these are also the unit of measures used for the monetary value of carbon);
- The monetary value of carbon.

The extent of marine habitats that provide this service are shown in Table 2. However, the boundaries relevant for carbon processing do not match well to EUNIS sediment type boundaries. For example, between mud and sand, key for carbon processing is the transition between permeability regimes (diffusion/advective) and oxygenation of the sediment as a result.

Since EUNIS A5.1 and A5.4 seem very spatially restricted and undefined, we have focused on mud and sand for which some estimates are available (see Table 14). Therefore, based on data availability, the sub-habitat selected for the calculations were: A2.5 (Coastal saltmarshes and saline reedbeds), A5.2 (Sublittoral sand), and A5.3 (Sublittoral mud). Other publications, for example Luisetti et al. (2019) report instead an area for UK saltmarsh of 42,712 ha, which includes the modern UK estuarine areas and is based on Davidson and Buck (1997).

We rely on the recent estimates of carbon burial rate found in the literature to estimate the ecosystem service flow of carbon sequestration and storage in saltmarshes (Luisetti et al., 2019)[7], and seabed sediments (de Haas et al. 1997; Oliver Legge, pers. comm.) (see Table 14).

Table 14. Estimates of the carbon burial rates for selected habitats

Habitat type	Carbon burial rates Tonnes/ha/yr
A5.1 Sublittoral coarse sediment	Not known
A5.2 Sublittoral sand	0.08
A5.3 Sublittoral mud	0.12

Source: de Haas et al. 1997; O. Legge pers. comm.

We chose to take a conservative approach for the aggregation of carbon buried in UK marine habitats by selecting the lowest burial rates available with the reviewed ranges of sedimentation rates (see Table 14).

[7] For saltmarshes, we extrapolated and used the combined above- and below-ground carbon sequestration rate from Luisetti et al. (2019), which is equal to 0.86 t/ha/yr or 3.14 tCO2e/ha/yr, which is consistent with the lower bound for carbon sequestration reported in the ONS/Defra scoping study for UK coastal margins (Defra/ONS, 2016). The extrapolated combined carbon sequestration and storage rate was calculated on the 1% of currently sequestered and stored carbon in the UK saltmarshes (over an aerial extent of 42,712 ha) standing stock of plants using data in Beaumont et al. 2014 and in the sediments using data from Rees et al. (2000), Adams (2008) and Parkes (2003).

For a given region, unless there is a radical impact or change, for example in trawling distribution, which would affect carbon cycling at the upper sediment layers (via mixing, faunal changes), the carbon burial rate should stay approximately the same year to year. Accordingly, we have assumed that the flow of carbon storage is the same for all the years considered. It could be envisaged that over decadal timescales perhaps upper carbon levels would shift depending on environmental forcing[8], which will eventually affect deeper carbon concentrations and burial, but it is difficult to say how significant such a change might be.

The economic value of carbon used is the abatement cost of non-traded carbon central value provided by BEIS (2017a). The non-traded sector includes all the emissions not covered within the EU Emission Trading System. Therefore, emissions are valued at the non-traded price of carbon (BEIS 2017b), which is more appropriate for the C flow we are estimating in this report. The only valuation option for this service is the one illustrated in Equation 1 below (cf. Defra / ONS principles and coastal margins). Since the BEIS abatement costs are in £/tCO_2e, the carbon burial rates in Table 14 were also converted into tCO_2e. The value of the carbon sequestration and storage benefit (healthy climate), *CB*, is estimated as (Luisetti et al. 2019):

$$CB_t = S_t * V_t$$

where S_t represents the carbon sequestration and storage service at time *t* and V_t is the economic value at time *t* used to measure the carbon sequestration and storage benefit in monetary units. The previous equation represents the economic value of the flow of the ecosystem service of climate regulation, specifically carbon sequestration and storage at a specific point in time. The magnitude of S_t is determined by:

$$S_t = a_t^{c,m} * Cb_t$$

that is, the total area extent of the coastal or marine ecosystem considered ($a_t^{c,m}$) (i.e., saltmarshes and/or seabed sediments) (in Table 2) multiplied by the carbon burial rate (Cb_t) measured in tonnes of carbon or tonnes of CO_2 equivalent per unit area at time *t,* (in Table 14 column *Carbon burial Tonnes/ha/yr*) Given the uncertainty surrounding the carbon burial estimates, we take a conservative approach and use the lower bound estimates per each habitat type of the marine sediments; for saltmarshes see Footnote 2*.* Cb_t can be determined by biogeochemical sampling and/or modelling.

[8] For this report, we define environmental forcing as the combination of natural and anthropogenic processes that cause environmental change.

Based on data concerning the extent of coastal and marine habitats considered available for the 2017 only, the estimate of total quantity of carbon stored in the same year is equal to 10.55 million tonnes of CO_2 equivalent, with a total monetary value of £601 million[9]. To show how the different coastal and marine ecosystems considered contribute to carbon storage and, in turn, to the monetary value of carbon stored, in Table 15 estimates for the 2017 are disaggregated by ecosystem.

Table 15. Estimated volume and value of carbon sequestration by habitat, 2017

Habitat	Extent (ha)	Carbon sequestration (million tCO2e)[a]	Monetary value (£ million)
Coastal saltmarshes	52,832	0.16	9.5
Sublittoral sand	26,484,814	7.68	437.8
Sublittoral mud	6,149,456	2.70	154.2
Total	32,687,102	10.55	601.5

[a] To obtain estimates in tCO2e, tonnes of carbon estimates are multiplied by 3.66.
Source: Defra / ONS (2016)

4.1.6 Places and seascapes for nature watching and recreation

Figure 6 shows the logic chain applied to the places and seascape cultural ecosystem service provided by coastal and marine habitats. Coastal and marine environments provide numerous recreational opportunities, e.g. walking, sport activities, nature and wildlife watching, sea angling, boating activities, etc. All these activities are enhanced by the aesthetic value of the surrounding natural environment and the man-made amenities aimed at improving the experience. Another important aspect relates to health and psychological effects of coastal and marine environments, often referred to as the “blue gym” (White et al. 2016). Moreover, cultural ecosystem services provided by coastal and marine places and seascapes embrace a much broader class of immaterial benefits as experiential interactions, intellectual and educational activities, spiritual and symbolic interactions, and so on.

The UK estimates of time spent at and total visits to beaches and the marine environment for recreation and nature watching are scaled up from the Monitor of Engagement with Natural Environment Survey (MENE) (Natural England 2019) to represent the UK population. Only the visits reportedly being at “beach” are accounted for in this report, as they provide a clearer spatial relation between the visits and the marine environment. The time spent at beaches and the marine environment in the period 2012 to 2017 has increased substantially

[9] The monetary estimates for carbon sequestration service in coastal margins reported in the study *Scoping UK coastal margin ecosystem accounts* (Defra / ONS, 2016) account for sand dune, saltmarsh, and machair habitats with a total extent of 136,005 ha. In this report, the service provided by sublittoral sand and mud is added to the service provided by saltmarsh habitats, resulting in a total extent considered of approximately 32 million ha (see Table 2 and Table 15).

from 408 million hours to 722 million hours; an increase of 77%. In the same period, the total visits have increased from 167 million in 2012 to 251 million in 2017; an increase of 49% (Figure 12). Therefore, time spent at beaches and marine habitats has grown proportionally more than the number of visits, generally signalling a more frequent and persisting use of coastal and marine, or 'blue' spaces, for outdoor recreation.

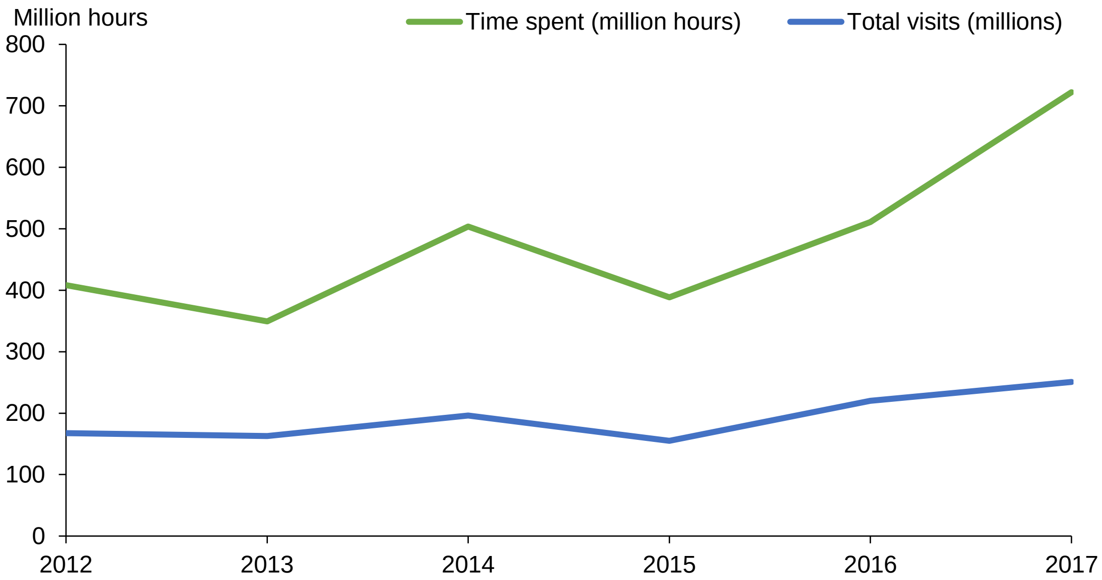

Figure 12. Outdoor recreation to beaches and marine blue spaces in the UK, 2012-2017
Source: MENE (Natural England 2019) and ONS (2019)

The economic valuation methods suitable for assessing the benefits from cultural ecosystem services can be broadly divided into revealed preferences and stated preferences approaches. The application of both these approaches is often limited by data availability, as they require statistical modelling and primary valuation studies. Also, stated preference methods provide welfare values instead of exchange values preferred for natural capital accounting. Moreover, it is very challenging to capture the whole set of cultural benefits with a single technique. Ideally, in order to account for all the cultural values, many techniques should be used concurrently.

A simple travel cost method used in the ONS natural capital accounting publications, which will be used in this report, is to consider the expenditures associated with travelling to the coastal and marine environment as a proxy for the value of the cultural ecosystem services provided. The value calculated, CES_t, represents the proxy for a general recreational value of coastal and marine environments, aggregating all the different cultural values described, and is derived as:

$$CES_t = avgcost_t * N_t$$

where $avgcost_t$ is the average cost per trip including travel, parking and admission costs, and N_t is the total number of visits to the coastal and marine environment.

The monetary value of recreation to beaches and the marine environment has increased of 23% between 2012 and 2107, from £744 million to £916 million. Expenditures reached the highest levels in 2014 and 2016, at £1.1 billion and £1.0 billion respectively (see Table 16). The average amount spent per visit, in contrast, has decreased in the same period of 18%, going from £4.40 in 2012 to £3.70 in 2017 (with a peak of £5.60 in 2014). The values in Table 16 generally follow the number of visits and time spent at habitat trends, with peaks in 2014 (196.0 million visits and 503.6 million hours, equal respectively to +2.1% and +4.8% on the average 2012-2017) and in 2016 (220.2 million visits and 510.7 million hours, equal respectively to +14.7% and +6.3% on the average 2012-2017). In contrast, the value for the 2017 is affected more by the significant decrease of per-visit expenditure than by the physical indicators of visits and time spent. Indeed, the peaking number of visits and time spent at habitat for the year (250.7 million visits and 722.2 million hours) do not translate in increasing expenditures for travel, parking and admission costs.

Table 16. Value of beaches and marine habitats recreation in the UK, 2012 to 2017 (£million, 2017 prices)

	2012	2013	2014	2015	2016	2017
Recreation	744.92	670.11	1101.36	813.90	1013.40	916.50

Note: all values expressed in 2017 real terms using GDP deflator published by ONS (2019).

Data used in this section are collected by the ONS and governmental agencies (Natural England), therefore can be considered to have a high confidence level and precision. Regardless, the data in the MENE relates only to recreational visits in England (Natural England 2019). The simple travel cost approach used in this report does not account for visits with no costs incurred, resulting in an underestimation of the total value (see Defra / ONS 2016 for a discussion). Moreover, as shown by the relationship between trends in monetary values and physical indicators, the link between the use of natural habitat and the value of human benefits stemming from it should be increasingly strengthened.

4.1.7 Renewable energy from Offshore Wind Farms

Figure 7 shows the logic chain applied to the abiotic service of renewable energy from offshore wind farms (OWFs) providing the benefit of wind-generated electrical energy. The main marine habitats enabling the provision of this service in the UK, following the EUNIS classification in Table 1, are some sublittoral sediments as shown in the 'Habitats' column. Although offshore wind farms (OWF) are not commonly located in sublittoral mud sediments, their cables are, they therefore are included in the logic chain. Also, currently, turbines are not located off the continental shelf (>200 m depth), which would correspond to A6 in the EUNIS classification in Table 1. However, turbines could be installed in those areas in the future. In the second column, spatial extent of the marine habitats identified, and wind energy are the main enabling features for the abiotic service of renewable energy (OWFs) to be provided. Energy provision and turbine construction are the two main activities related to this service, which in turn could create damages to the marine habitats affected by its provision. However, the Crown Estate lease specific zones for these activities and the Marine Management Organisation (MMO) grant licences to wind farms within these specific zones providing some human management to balance the impacts.

Energy generated by offshore wind sources is published by the Department for Business, Energy and Industrial Strategy (BEIS) in the annual Digest of United Kingdom Energy Statistics (DUKES) (BEIS 2013–2018). Figure 13 compares the time series of energy generated using main renewable sources in the UK between 2012 and 2017. Following the general trend of renewable energy use, electricity generated from OWFs has increased by 175% between 2012 and 2017, reaching 20,916 GWh generated in 2017. In the same period, offshore wind energy has constituted, on average, 20% of the total energy mix generated by renewable sources.

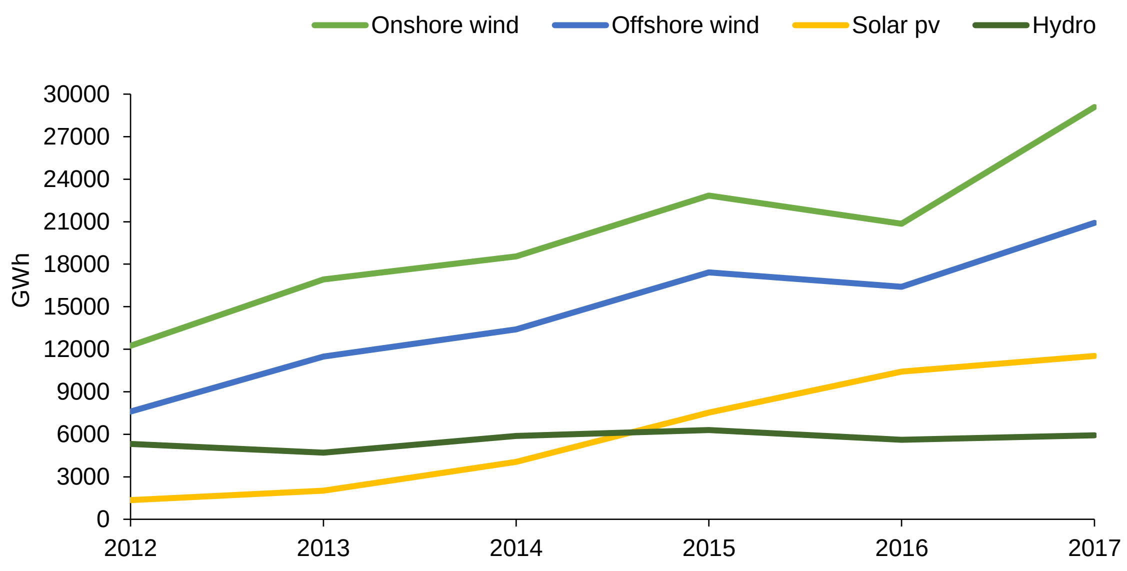

Figure 13. Energy generation from main renewable sources in the UK, 2012-2017
Source: BEIS 2013–2018

The residual value approach used to determine the resource rent as outlined in Section 3.1.1 has been used to estimate the monetary value of the ecosystem service provided by offshore wind power generation, in line with the UK natural capital monetary estimates 2016 (Defra / ONS 2016). Data are sourced from the annual financial statements of 25 major wind power producers listed in the DUKES (BEIS 2013–2018). These companies represented around 78% of total wind power production in the period considered. The resource rent is obtained using the gross operating surplus derived by subtracting the costs of sales and employment from the total turnover of the companies considered. The gross operating surplus is then corrected for the depreciation of fixed capital and the return to produced assets. The latter has been calculated using the British Government Securities 10-year Nominal Par Yield series in real terms (Bank of England 2019). Finally, unit resource rents for each year are calculated by dividing the total resource rent by the total units of wind energy generated for a given year. The unit resource rents derived for the years 2012 to 2014 are averaged and applied to the period 2015 to 2017.

Table 17 shows the total monetary value of the ecosystem service provided by the environment with offshore wind energy generation. The resource rent value of offshore power generation has increased of 152% between 2012 and 2017, when it peaked at £2.3 billion. The monetary value has experienced a substantial increase in 2015, being 103% higher than the previous year. The trend of monetary values in Table 17 generally follow the increasing offshore wind energy generation (Figure 13), that in turn mainly depends on the increased installed capacity (+15.5% yearly average between 2012 and 2017). The decline of monetary values between 2013 and 2014 and between 2015 and 2016 are partially explained, respectively, by lower industry turnover and lower wind speeds (BEIS, various years).

Table 17. Value of offshore wind generated power in the UK, 2012 to 2017 (£million, 2017 prices)

	2012	2013	2014	2015	2016	2017
Offshore wind	934.8	1077.5	963.9	1962.2	1847.7	2355.6

Note: all values expressed in 2017 real terms using GDP deflator published by ONS (2019).
Source: BEIS (2013-2018).

Data used in this section are collected by the Office for National Statistics (ONS), the relevant governmental department (BEIS) and the public financial documents compiled by energy companies, and therefore can be considered to have a high level of confidence and precision. Data used in the calculation of the resource rent relate only to those companies with financial documentation publicly available.

4.1.8 Marine Aggregates

Figure 8 shows the logic chain applied to the abiotic service of aggregates extraction, which involves the extraction of sand and gravel from the seabed. The main habitats providing this service in the UK, following the EUNIS classification in Table 1, are shown in the ‘Habitats’ column of the logic chains (Figures 2 – 8). It is worth noting that the industry targets particular deposits (e.g., drowned river bed features). Accordingly, there is a 3-dimensional element to aggregates deposits Not all areas of the seabed with sand and gravel are therefore suitable for extraction. In addition, the industry will avoid areas with appreciable silt content. The main habitat features enabling the service are spatial extent of the seabed habitats identified and depth of deposit. Since the activity of extraction itself, even when dedicated to coastal protection, can put at risk the provision of the service in the future, some management interventions, such as marine licencing, have been applied. However, the spatial extent of activity is not the same as area licensed. So, consideration is given also to the potential for recovery of the seabed. For example, although, given the non-renewable nature of the abiotic service, the deposits will not regenerate, if a surface layer of the deposit (licence conditions require at least 0.5 m) is left, this facilitates faunal recovery. We know from research that dredging can cause changes in the physical habitat and associated biology although work carried out under the Regional Seabed Monitoring scheme (Cooper and Barry 2017) aims to ensure that the affected habitat is left in a state which will enable the return of the benthic community, thus preserving the functioning of the ecosystem. For example, Cooper et al. (2019) show that seabed mapping can facilitate the management of marine habitats, but that different seabed habitat classification approaches provide different levels of information. In their paper, Cooper et al. (2019) show that biologically-based habitat maps could offer a more cost-effective basis for ecological monitoring because being able to detect ecological change, they can better inform marine spatial planning.

The quantity of aggregates provided by the coastal and marine environment is derived from the Summary Statistics of Marine Aggregates published yearly by the Crown Estate and refers to removal of sand and gravel from the seabed of English and Welsh territorial seas and continental shelf, under licence from the Crown Estate (See Figure 14).

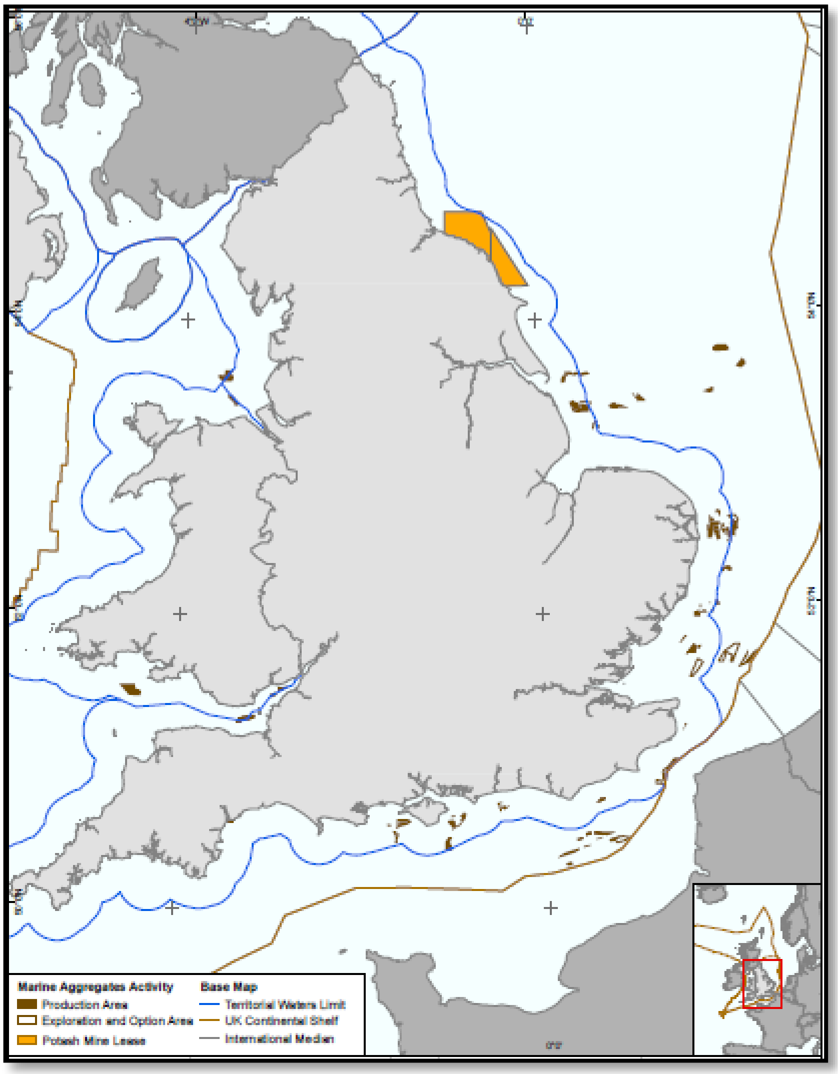

Figure 14. Areas licensed for the extraction of marine aggregates
Source: © The Crown Estate 2019

The total quantity of aggregates extracted by licensed companies has not fluctuated substantially between 2012 and 2017, averaging around 18 thousand tonnes and with an increase of 16% during the reporting period. The increased extraction of aggregates has been driven by a 41% rise in demand from 2012 - 2017. In contrast, during the same period, quantities extracted abroad have decreased by 31% and the use of aggregates for beach nourishing and filling by 4% (See Figure 15).

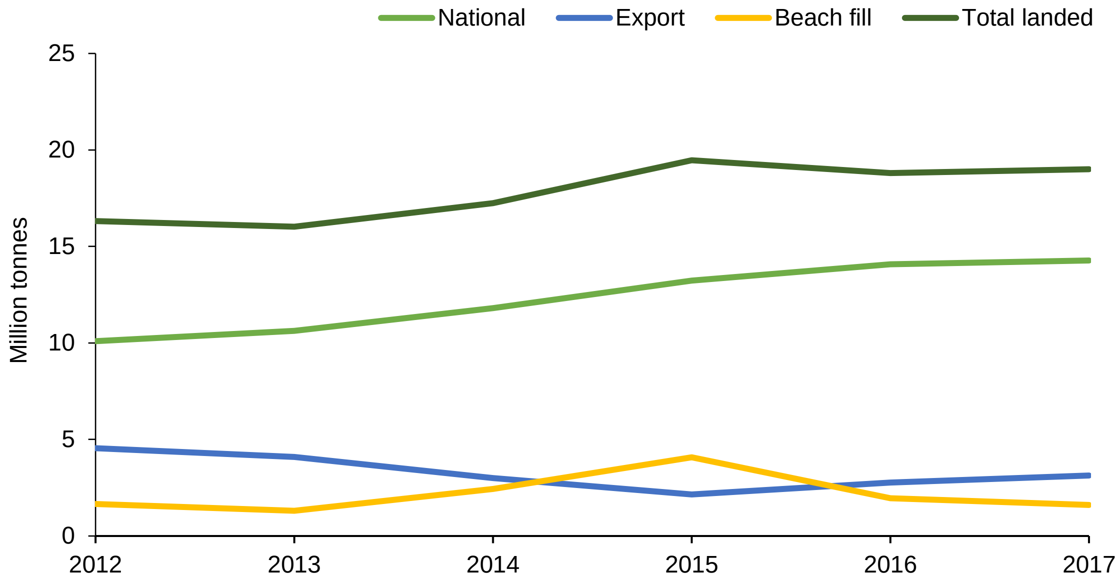

Figure 15. Extraction of marine aggregates from Crown Estate licensed areas 2012-2017
Source: The Crown Estate 2019

The monetary valuation of the ecosystem service provided by the extraction of marine aggregates is obtained using a resource rent-based approach. The total output sales of the sector are obtained multiplying the quantity of marine aggregates extracted in each year by the market price of aggregates. The market price used, £7.03 per tonne including levies and royalties, is the most recently available estimated value (ABPmer, in press). As specific data regarding production costs and capital assets of the licensed companies are not available, the resource rent is calculated using a residual value approach from the data presented in the System of National Accounts (SNA) considering the whole “mining and quarrying” sector (SIC 05-09). The aggregation of industrial sectors used in the SNA refers to different levels between the input-output tables and the tables on fixed capital consumption and production, with the former including the “other mining and quarrying” sector (SIC 08) and the latter grouping all activities under “mining and quarrying” (SIC 05-09). Therefore, to maintain consistency between the two data series, the resource-rent-to-output ratio is calculated on the whole “mining and quarrying” sector and then applied to the total value of the aggregates extracted. To derive the return on produced assets, the British Government Securities 10-year Nominal Par Yield (Bank of England 2019) series in real terms is applied to the net produced fixed capital for each relevant year.

The monetary valuation obtained for the ecosystem service provided by the marine environment through aggregates extraction shows a decreasing trend between 2012 and 2016. The value of aggregates decreased by 70% during that period with an annual average decrease of 27% (see Table 18), in contrast to the opposite, positive trend related to physical provision. A possible reason could be related to structural and financial changes faced by the industry in recent years resulting in lower operating surplus and fewer exports.

Table 18. Value of marine aggregates extracted in England and Wales, 2012 to 2016 (£million, 2017 prices)

	2012	2013	2014	2015	2016
Aggregates	46.57	38.42	29.29	19.33	14.18

Note: all values expressed in 2017 real terms using GDP deflator published by ONS (2019).

The ecosystem services of marine aggregates extraction for the UK is likely to be underestimated due to data availability limitations. The data used in this section are collected by the agency responsible for marine aggregates extraction in England and Wales (MMO), and therefore does not include quantities extracted in Scotland and Northern Ireland. Moreover, the market price is not directly disclosed by extraction companies.

5 The supply and use of marine and coastal ecosystem services in the UK: Physical and monetary ecosystem services accounts

The capacity of marine and coastal ecosystems to provide services to society depends largely on the habitat extent and condition relative to the ecosystem service of interest. In this section, for the ecosystem assets identified in Table 1, we are able to provide information on their extent (Table 2).

Indicators for condition, or the ability of the ecosystem to deliver specific ecosystem services, was set out in Tables 4 and 5. Unfortunately, the specific data required to distinguish condition of individual habitats are not available. In the absence of these data, we have assumed that all areas of habitat are in reasonable condition to provide services at the current rate. Addressing these data gaps is identified as a priority area for future research.

In the following sections, the results obtained in Section 3 regarding the calculation of ecosystem services physical and monetary supply provided by the UK coastal and marine environment will be summarised, following the accounting standard, with the development of supply and use tables.

5.1 UK marine and coastal physical ecosystem services account: Supply table

Ecosystem services flowing from marine and coastal habitats provide benefits to society. Where possible, we assess the annual contribution to natural capital of each service showing the amount of service in physical terms supplied by specific marine habitats.

Table 19. UK marine and coastal ecosystem services physical account 2012-2017

Ecosystem service	EUNIS Habitat	2012	2013	2014	2015	2016	2017
Provisioning							
Finfish and shellfish (Thousand t fish catch)	A2, A3, A4, A5, A6	581.26	583.28	696.27	644.36	656.61	673.50
Regulating							
Climate regulation (carbon) (Million tCO_2e carbon stored) [a]	A2, A5, A6	10.55	10.55	10.55	10.55	10.55	10.55
Natural hazard protection (ha habitat extent) [a]	A1, A2, B1, B2	52832.00	52832.00	52832.00	52832.00	52832.00	52832.00
Waste (nutrient) remediation (Thousand t pollutants discharged) [a]	A2, A5, A6, B1	109.10	109.10	109.10	109.10	109.10	109.10
Cultural							
Recreation (Million hours spent at habitat)	A1, A2, B1, B2, B3	408.25	349.30	503.58	388.24	510.73	722.18
Abiotic							
Renewable electrical energy from wind (GWh energy generated)	A5	7603.17	11471.78	13404.59	17422.74	16405.74	20915.91
Aggregates extracted (Million t)	A5	16.31	16.03	17.25	19.47	18.81	19.00

[a] the ecosystem service provision is assumed to be constant between 2012 and 2017 due to data limitations.
Note: For climate regulation, the physical provision of the service is considered constant but its value changes because the price of carbon is reviewed annually.

5.2 The value of services provided by marine and coastal ecosystems in the UK

The economic value of each of the key ecosystem services provided by marine and coastal ecosystems is also, whenever possible, assessed in this report for its annual contribution to Marine and Coastal Natural Capital. The economic accounts for marine and coastal ecosystems show the value of each service supplied by specific marine and coastal habitats. These are static estimates calculated at a specific point in time. All values are expressed in 2017 prices.

Table 20. UK marine and coastal ecosystem services monetary account 2012-2017(£ million, 2017 prices[a])

Ecosystem service	EUNIS Habitat	2012	2013	2014	2015	2016[b]	2017
Provisioning							
Finfish and shellfish	A2, A3, A4, A5, A6	122.76	89.72	97.29	81.37	146.61	NA
Regulating							
Climate regulation (carbon) [c]	A2, A5, A6	607.07	607.20	608.01	616.38	614.74	601.48
Natural hazard protection [c]	A1, A2, B1, B2,	5590.42	5590.42	5590.42	5590.42	5590.42	5590.42
Waste (nutrient) remediation [c]	A2, A5, A6, B1	1691.83	1691.83	1691.83	1691.83	1691.83	1691.83
Cultural							
Recreation	A1, A2, B1, B2, B3	744.92	670.11	1101.36	813.90	1013.40	916.50
Abiotic							
Renewable electrical energy from wind	A5	934.82	1077.47	963.93	1962.23	1847.69	2355.64
Abiotic products - aggregates extracted	A5	46.57	38.42	29.29	19.33	14.18	NA

[a] All values expressed in 2017 real terms using GDP deflator published by ONS (2019).
[b] Data from SNA Supply and Use tables 2012–2016 were used to calculate the resource rent.
[c] The ecosystem service provision is assumed to be constant between 2012 and 2017 due to data limitations. Also note that for climate regulation, the physical provision of the service is considered constant but its value changes because the price of carbon is reviewed annually.

5.3 The users of marine ecosystem services in the UK

The compilation of a coherent natural capital and ecosystem services accounting requires the identification of users and beneficiaries of the various marine and coastal ecosystem services in the UK. This is important for getting to grips with the political realities of managing these resources. Some ecosystem services are critical to specific sectors of national, regional or local economies. Defining who are stakeholders and who, specifically, are the end users and beneficiaries will help decision-makers, especially policy-makers but also resource managers, guide investment for the most cost-effective impact. In this section we specifically look at who could be the direct users for the key marine and coastal ecosystem services discussed in this report.

Tables 18 and 19 show the basic structure of possible physical and monetary use tables for the UK marine and coastal environments following the guidance of the SEEA (UN et al. 2014). The use table makes explicit the contribution of ecosystem services to economic and human activities, including household and government, following the convention used in the national accounts. The development of use table requires linking the ecosystem services provision with the different user categories. This poses several challenges:

1. The link between ecosystem services and users is not always directly related to a well-defined spatial area
2. The basic users' categories shown in Tables 21 and 22 could be further disaggregated, spatially or by sector, allowing wider policy use
3. The disaggregation of ecosystem services supply between different users' types is often made difficult by lack of suitable data

Due to a lack of a precise spatial link between ecosystem services provision and users, the structure of the use table has to be informed by the possible uses of ecosystem services and data availability. In this research, the initial compilation of use tables provided in Tables 21 and 22 only shows the contribution to those sectors or population segments representing the first direct users of the ecosystem services. The contribution to intermediate and final users/beneficiaries is not explicitly reported because of lacking data and missing accepted methodology. Therefore, in Tables 21 and 22 only one cell is compiled for each ecosystem service. The development of complete use tables represents one of the main challenges for future work on coastal and marine natural capital accounts.

Regardless, departing from the basic classification of users shown in Tables 21 and 22 still requires the disaggregated categories to be comparable with the national accounts. A possible further classification of industry/business users could be based on the one used in national accounts, namely Standard Industrial Classification (SIC UK) and Nomenclature of

Economic Activities (NACE EU). Moreover, for some ecosystem services (e.g. regulating and cultural) it can make sense to classify the users in terms of local, national, and global. Further possible disaggregation could concern government based on the institutional level (local, regional or central).

Finally, it is worth noting that the initial structure and compilation of Use tables developed in this report focuses on identifying possible direct users of ecosystem services provided by the UK coastal and marine environment. The choice of focusing on direct users is linked with the foundational use of accounts, namely the need to measure and track the contribution of natural assets to the production of goods and services. Identifying direct users, therefore, allows to establish at what point the contribution from natural assets enters in the production and provision process of goods and services.

Wider beneficiaries should also be accounted for in the development of Use Tables, especially focusing on social dimensions of beneficiary groups. Indeed, valuation of benefits stemming from ecosystem services provision can be assessed only if beneficiaries exist. Particularly relevant on this point is the disaggregation of households based on their socio-demographic characteristics. The SEEA tends to treat households as a single economic unit. Further work would be needed to identify and disaggregate relevant household types on the basis of income level, gender, household composition, working status, geographical position, etc. This poses several challenges adding up to those concerning direct user identification, specifically related to social and spatial heterogeneity of households and the possibility to establish links with ecosystem services provision. A possible way forward could entail trying to establish a link between ecosystem services accounts and available secondary data (e.g. census, national socio-demographic surveys, etc.). This would help to classify beneficiaries and identify how environmental interventions, management and public investments impact different groups.

Table 21. UK marine and coastal ecosystem services physical use by sector. Example for 2016.

Ecosystem service	Industry								Households	Government	Rest of the world - Exports	Flow to the environment
	Agriculture, forestry, fisheries	Mining and quarrying	Manufacturing	Electricity and gas supply	Water collection, treatment and	Construction	Tourism industry	Other industries				
Provisioning												
Finfish and shellfish (Thousand t fish catch)	656.61	-	-	-	-	-	-	-	-	-	-	-
Regulating												
Climate regulation (carbon) (Million tCO2e carbon stored)	-	-	-	-	-	-	-	-	-	10.55	-	-
Natural hazard protection (ha Habitats extent)	-	-	-	-	-	-	-	-	-	52832.0	-	-
Waste (nutrient) remediation (Thousand t pollutants discharged)	-	-	-	-	109.10	-	-	-	-	-	-	-
Cultural												
Recreation (Million hours spent at habitat)	-	-	-	-	-	-	-	-	510.73	-	-	-
Abiotic												
Renewable electrical energy (GWh generated)	-	-	-	16405.74	-	-	-	-	-	-	-	-
Aggregates extracted (Million t)	-	18.81	-	-	-	-	-	-	-	-	-	-

Note: The initial compilation of use tables only shows the contribution to those sectors or population segments representing the first direct users of the ecosystem services. In Tables 21 and 22 only one cell is compiled for each ecosystem service, as the contribution to intermediate and final users/beneficiaries is not explicitly reported because of lacking data and missing accepted methodology.

Table 22. UK marine and coastal ecosystem services monetary use by sector. Example for 2016 (£ million, 2017 prices).

Ecosystem service	Industry								Households	Government	Rest of the world - Exports	Flow to the environment
	Agriculture, forestry, fisheries	Mining and quarrying	Manufacturing	Electricity and gas supply	Water collection, treatment and	Construction	Tourism industry	Other industries				
Provisioning												
Finfish and shellfish	146.61	-	-	-	-	-	-	-	-	-	-	-
Regulating												
Climate regulation (carbon)	-	-	-	-	-	-	-	-	-	614.74	-	-
Natural hazard protection	-	-	-	-	-	-	-	-	-	5590.42	-	-
Waste (nutrient) remediation	-	-	-	-	1691.83	-	-	-	-	-	-	-
Cultural												
Recreation	-	-	-	-	-	-	-	-	1013.40	-	-	-
Abiotic												
Renewable energy	-	-	-	1847.69	-	-	-	-	-	-	-	-
Aggregates extracted	-	14.18	-	-	-	-	-	-	-	-	-	-

Note: All values expressed in 2017 real terms using GDP deflator published by ONS (2019). The initial compilation of use tables only shows the contribution to those sectors or population segments representing the first direct users of the ecosystem services. In Tables 21 and 22 only one cell is compiled for each ecosystem service, as the contribution to intermediate and final users/beneficiaries is not explicitly reported because of lacking data and missing accepted methodology.

5.4 Confidence level of the monetary estimates reported in the supply and use tables

The experimental nature of the coastal and marine natural capital accounts developed in this report requires a clarification on the confidence level of the monetary estimates calculated to populate the supply and use table. A confidence level for the values obtained allows to better understand the perspective on how the figures should be used, how to compare results in this report with other publications, and where the main gaps are. Considerations about the confidence level of data and methodology used are reported for each ecosystem service in Sections 4.1.2 to 4.1.8 and are summarised in Table 23.

Table 23. Confidence level of the monetary estimates used to compile the accounts

Ecosystem service	Confidence	Notes
Finfish and shellfish	Medium/High	Data are collected by national and international fisheries organisations (MMO, ICES) resulting in high confidence. Resource rent calculation can be improved by using more detailed sector information.
Waste (nutrient) remediation	Medium	Likely to be an underestimation of the true service value as the method accounts for a limited set of pollutants and sources.
Natural hazard protection	Medium/low	Valuation considers saltmarshes only, excluding other habitats relevant for the service provision. Strong working assumptions have been formulated to obtain the initial estimates.
Climate regulation (carbon)	Medium	Main limitation is related to scientific uncertainty around carbon burial rates estimates currently available.
Recreation	Medium/High	Data are collected by national agencies (NE, ONS) resulting in high confidence. The main limitation relates to primary data available for England only.
Offshore wind energy	Medium/High	Data are collected by government and national agencies (BEIS, ONS). Resource rent calculation can be improved by using more detailed sector information.
Abiotic products - Marine aggregates	Medium/High	Data are collected by national agencies (Crown Estate). Resource rent calculation can be improved by using more detailed sector information. The main limitation relates to primary data available for England and Wales only.

6 Challenges and potential future directions

The aim of this project was to advance the development of natural capital accounting with regard to UK marine and coastal ecosystems. Recognising that natural capital accounting for marine and coastal ecosystems is a relatively new activity and given the time constraints of this project, we were limited to the use of existing, available data and building on previous analysis, much of it very recent if not concurrent. Our investigations confirmed the unique challenges and immense scale of work to be done in this field.

Marine and coastal natural capital accounting is particularly challenging, not only because the UK marine environment is approximately three times the size of the terrestrial environment and much less accessible, but because of overlapping attribution of assets in the important coastal zone, the economic importance of mobile marine natural assets and the patchy nature of the information available. The ability of UK marine and coastal habitats to deliver ecosystem services, which provide significant and important benefits to the UK economy and society, is determined largely by the extent and ecological condition of those assets. Our knowledge of both of those aspects across the spectrum of marine and coastal ecosystems is far from complete. Indeed, the underpinning science although developing at a rapid pace remains wholly inadequate to address the global challenges that we face in this sector. Important outputs of this project therefore are not only the assessment of existing information but the identification of inconsistency in methods and gaps in our knowledge and the prioritised recommendations for the improvement of the data and asset coverage.

6.1 Methodological issues in marine and coastal accounting

There are a number of methodological issues that are still widely debated in terms of natural capital accounting. The reconciliation of the overlap between marine and coastal accounts with other UK natural capital accounts remains at the forefront. Although there may be overlap in many areas, such as grazing on salt marshes, we chose to focus on the ecosystem services that would benefit entities in the marine environment and the maritime economy. A more in-depth look at, and analysis of, these types of activities would aid in giving a truer assessment of the benefits of these ecosystems as well as the sustainability of these ecosystem services.

The appropriate level of spatial disaggregation also remains a challenge. What is useful at the national level may be much less useful at the country level and, as much of the data is collected by country agencies with responsibility for environmental conservation and management, better coordination and collaboration may be needed to develop a system that serves the several potential beneficiaries better.

Adjustments to the analysis to reflect changing ecosystem conditions, whether due to extreme weather events, ecological recovery, restoration or an altered management regime, may also need further consideration.

Analyses of the delivery of ecosystem services are generally considered by individual service when, in reality, they generally come in 'baskets', that is, multiples. Moreover, it is recognised that the production of certain ecosystem services is synergistic while others are antagonistic, so considering them in isolation may not give the best estimate, especially of long-term deliveries.

The suitability of information developed from survey data as opposed to modelling is another issue that may need to be addressed with implications for the direction of funds for filling data gaps. The development of new technology and software in this area should be explored to the fullest.

6.2 Data and information gaps, their priority and possible means of solution

It almost goes without saying that the data gaps faced by this project were enormous, in nearly every context. Whether map images, or data on length and width of coastline, the condition of protective ecosystems, with and without seawalls, the measurable health benefits of seaside recreation, quantifiable value of cultural importance of maritime heritage, the paucity of useful data and information for natural capital accounting was striking. For the data that we were able to access much was incomplete or inconsistent, often lacking the benefit of regular monitoring. It should be recognised that nearly 36% of UK marine and coastal environment falls in the category of just 'Seabed' or 'Known unknown'. Habitat designation for these areas must be the priority. The variety of new technology available, particularly in the field of earth observation and remote sensing, must be applied to this challenge. Public and private sector cooperation on data collection could benefit all. Extent of our broad-scale habitats must be the first level of understanding.

The implementation of reporting under the MSFD will contribute to the provision of data on marine ecosystems. In particular, this should provide better data on water quality for marine waters that will complement assessments currently carried out for transitional and coastal waters under the WFD. These data will enable a better understanding of the capacity of coastal habitats to deal with various pollutant and nutrient loads. It must be recognised that incoming flows reflect not only current but past activities on land as it can take decades for seepage to groundwater to re-emerge carrying with it a signature of historical activities within a watershed.

We need a better understanding of the various ecological processes associated with the delivery of key ecosystem services, such as related to heavy metal movement in the marine environment, nutrient remediation in the deep-sea, effectiveness of natural hazards protection of complex sea margins (e.g. seawalls plus saltmarsh), effectiveness of different species in influencing the delivery of various ecosystem services and in terms of resilience after intervention. A better scientific understanding will enable more appropriate indicators to be developed.

Overall ecosystem condition may not be a good indicator of whether or not a specific habitat can continue to deliver a particular ecosystem service on a sustainable basis. Although some recommend the use of MSFD and OSPAR ecological indicators for condition assessment for natural capital accounting, the ‘ecological score’ which is reported to MSFD or OSPAR is actually an ‘index’, that is, the result of combining several indicator scores. Many of these indicators may have little, if any, relevance to the provision of the target ecosystem service, e.g. aggregates. Moreover, the score would relate to the UK as a whole and not to a particular habitat as broken down for the purposes of natural capital accounting. Overall ecosystem condition and what sustainable ecosystem services are the priority is a different debate that needs to be addressed in the policy arena.

The rising importance of climate regulation highlighted questions regarding the effectiveness of different marine and coastal habitats, especially saltmarsh and seagrass, in sequestering carbon, and the impact of changing environmental parameters on the effectiveness of these processes. We know that similar sediment types in different regions of the deep sea can store different amounts of C, however, the boundaries relevant for carbon processing do not match well to EUNIS habitat categories.

Finally, a better method for collecting and analysing data on cultural values associated with the marine and coastal environment is needed. The UK is a ‘maritime nation’ and much of its history is associated with the seas, not only locally but globally. More effort needs to be given to developing a better understanding of society’s appreciation for marine and coastal habitats in the UK.

6.3 Potential refinement of marine natural capital accounts

The feasibility of developing further accounts will depend on the benefit provided by the additional information as compared to the cost. Innovative ways should be sought to develop information of benefit to both public and private sector activities. New technologies for remote data collection and automated data analysis offer potential for significant advances in information flows. With devolved responsibility for environmental management, a priority area of research going forward should be to increase the granularity and ability to produce local or sectoral accounts. Understanding better the potential users and beneficiaries could open opportunities for joint working and civil society involvement. New economic endeavours, such as deep-sea mining, especially in the high seas, should be monitored closely to ensure that the rise of one marine-related industry does not predispose the collapse of another. More attention needs to be given to what are called ‘Inter-services’ (see SEEA central framework), which addresses inter-sectoral flows with environmental impacts, such as finfish discards as a result of the discards ban, or the release of stored carbon following fishing or aggregates extraction. Social and economic trends need to be monitored with an assessment of their impact on marine and coastal environments.

Climate change in particular will put increasing stress on our environment. Marine, like terrestrial habitats, will face warming temperatures but marine species will be facing the increasing threat of ocean acidification. Coastal ecosystems will be under particular stress as the projected increased frequency and intensity of storms will batter coastal habitats threatening sea cliffs and coastal habitation. Rising sea levels threaten to transform significant areas of the UK coastline. The UK is among the most vulnerable of the European countries subject to sea level rise with some 78% of the population living within 50 km of the coast. The defence of near-coast infrastructure is of concern to a number of sectors and government departments, including transport, energy, agriculture and defence. Understanding the ecosystem services associated with coastal habitats will be important for developing policy and management plans to deal with climate change stress and enhance the resilience of these areas. This relatively little understood environment with strong economic links deserves far more attention for both ecological as well as economic reasons.

6.4 Potential use of marine natural capital accounts in policy and management

Natural capital accounting follows on from the UK's ground-breaking work on the UK National Ecosystem Assessment (UK NEA) (2011). The ecosystem services delivered by the wide spectrum of UK marine and coastal habitats are many and varied and have yet to be fully delineated, certainly not by habitat but neither overall. The natural capital approach is raising our awareness of marine and coastal ecosystem services, recognising the cross-cutting nature of many services, carbon for instance, and demanding consideration of the underpinning science in the attempt to estimate the value and understand the sustainability of ecosystem services from an environment about which we are still know very little. Following UN guidelines (SEEA) natural capital accounting responds to the broad call for a national accounting approach that takes into consideration the sustainability of economic activity in terms of environmental context.

Natural capital accounting will also help us appreciate the contribution of marine and marine related sectors to national economy and societal well-being. This will facilitate the incorporation of marine and maritime sector considerations, such as fisheries, MPAs, and offshore energy development, into key government policy documents. Based on the best available science and accessible data it supports the government objective of science-based policy and taking an ecosystem approach to natural resource management. Better understanding of the relationship between ecological processes and economic outcomes should aid decision-making in marine spatial planning and licensing needed for the 'proportionate management' of the marine environment with consideration for the development of the blue economy.

In addition, natural capital accounting supports the objectives of HMG 25 Year Environmental Plan (2018), especially Chapter 5: *"Securing clean, healthy productive and biologically diverse seas and oceans."* The information developed will assist the UK in meeting the goals of natural capital accounting for marine and coastal habitats and provide useful information for the design of fisheries policy, which in turn will underpin trade negotiations. Information developed through natural capital accounting will strengthen UK contribution and reporting to a number of multilateral agreements including the Marine Strategy Framework Directive (MFSD), OSPAR and the Water Framework Directive (WFD).

A natural capital approach also supports the UK government's commitment to several international, multilateral environmental agreements including the UN brokered Sustainable Development Goal (SDG) 14: "*Conserve and sustainably use oceans, seas, and marine* *resources for sustainable development*, and the Commonwealth Blue Charter. This approach will help UK deliver on ‘Because the Ocean’ which supports the Paris Climate Agreement. A better understanding of ecosystem services of the coastal margins, which will be under increasing threat as climate change proceeds, will enable us to tackle the development of joint land/marine approaches to address key issues such as climate change, waste, pollution and natural hazards.

Natural capital accounting by its very nature, i.e. the stepwise progression of the analysis, the link between biophysical and social science, the need for regular monitoring, will help place science and economic evidence at the forefront of decision-making. Identifying gaps in monitoring and evaluation will need to be addressed to enable policy to be more securely based on a sound understanding of the costs and benefits, including biophysical trade-offs, of different policy and development options.

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8 Supplementary Material

8.1 Spatial distribution of EUNIS marine habitats

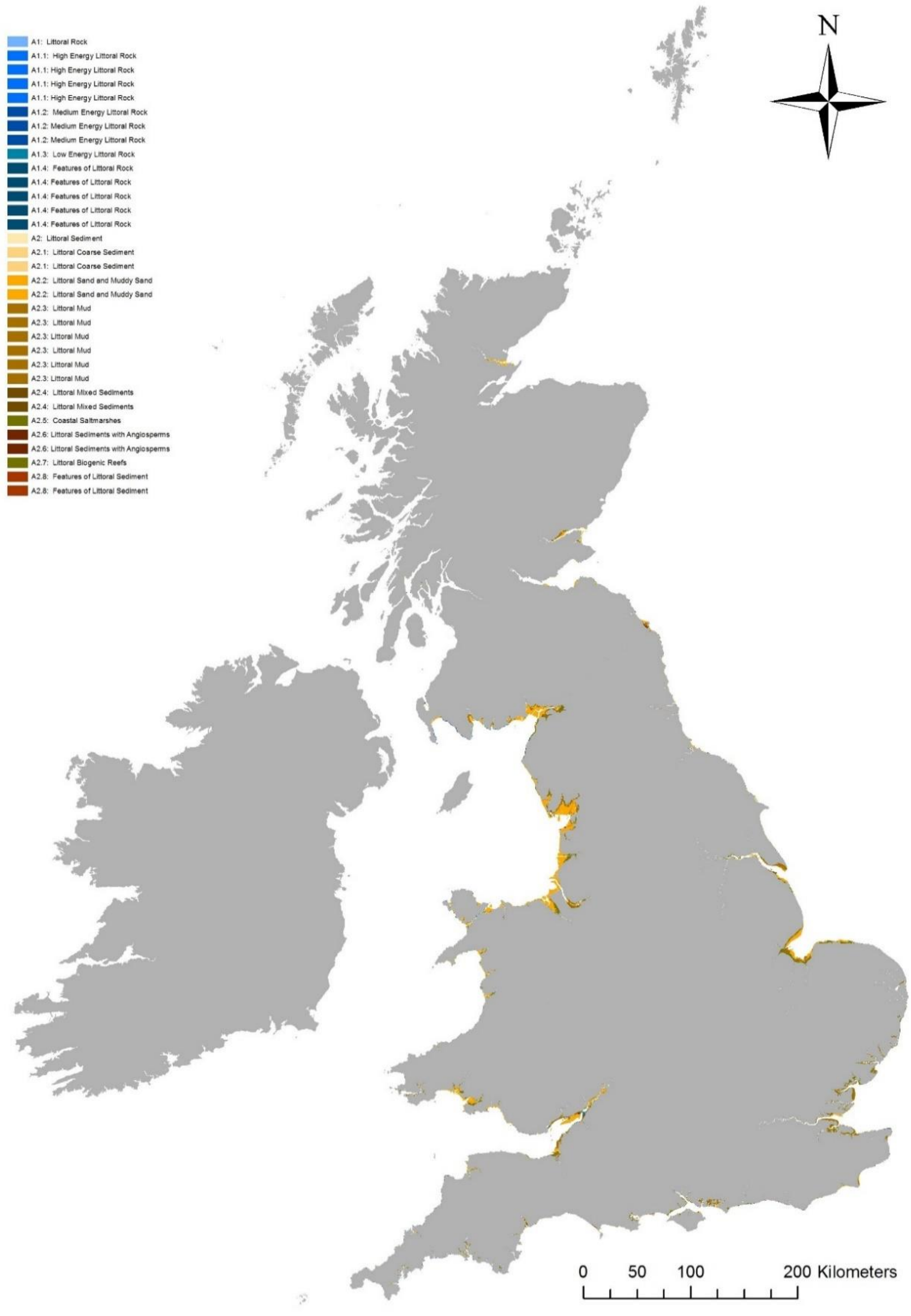

Figure S1. Distribution of littoral habitats (EUNIS categories A1 and A2) around the UK (JNCC 2017)

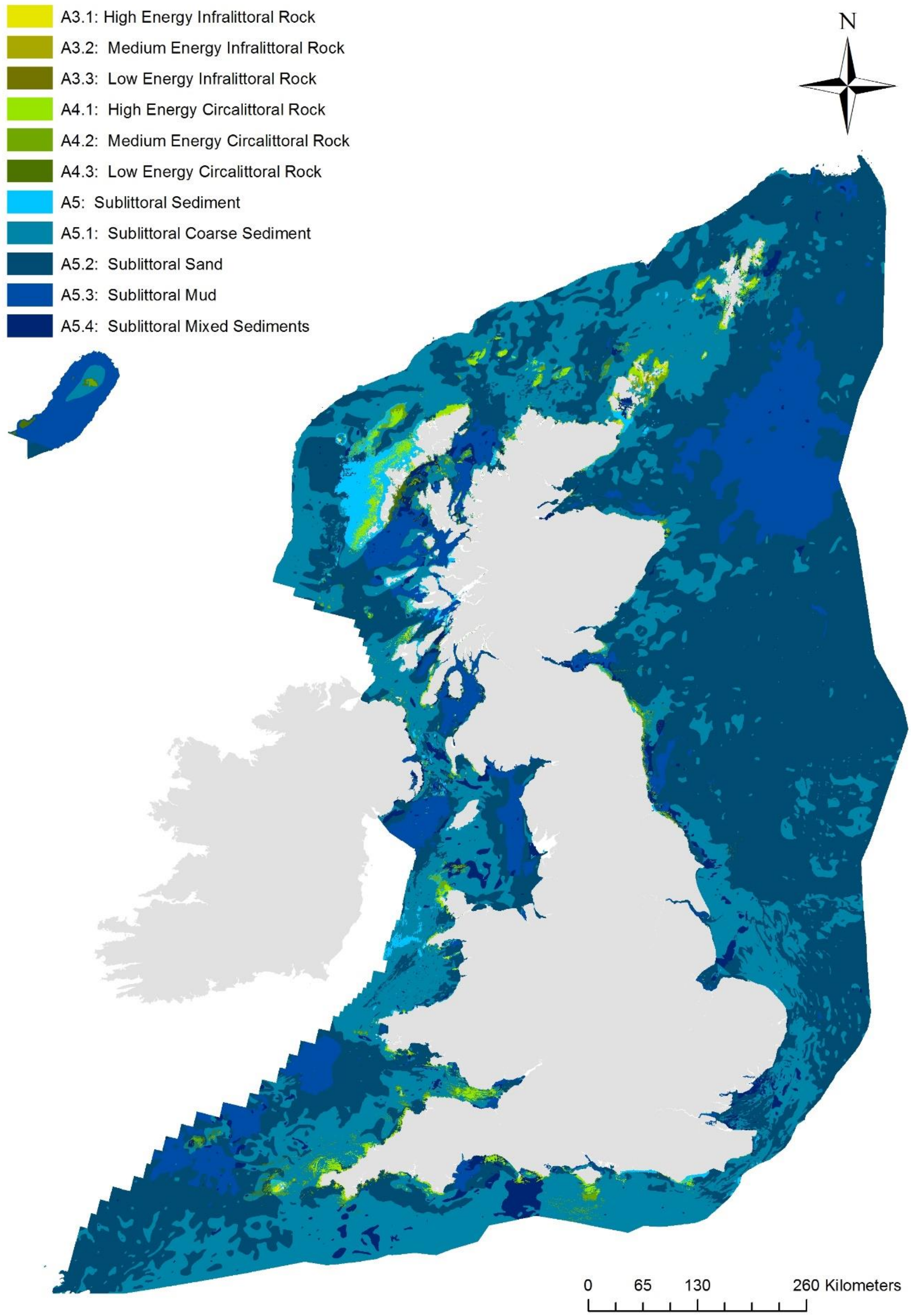

Figure S2. Distribution of infralittoral, circalittoral and sublittoral habitats (EUNIS categories A3, A4 and A5) around the UK (JNCC 2019)

Figure S3. Distribution of deep-sea habitats (EUNIS category A6) around the UK (JNCC 2019)

8.2 Alternative ecosystem services valuation methods and data

8.2.1 Finfish and shellfish

The use of a residual value resource rent approach is considered coherent with SEEA guidelines and suitable for this initial coastal and marine natural capital accounting. Nevertheless, it is recognised that this approach could be potentially more precise and that other approaches could perform better in assessing the value of the provisioning service and the capital stock. The approach for estimating the value of the provisioning service could be perfected in several areas. The first issue relates to the data limitation concerning the costs of the fishing industry. Indeed, costs (operating, employment, taxation, etc.) derived directly from the national fleet through nation-wide surveys would allow better construction of the resource rent value. Moreover, the catch of different finfish species is related to different levels of cost for fishers. If such data were available, the costs linked to catch of each species would be directly linked with the ex-vessel price landings for the same species, allowing a more reliable resource rent value to be calculated. The second issue relates to the assessment of discards, which are not readily accounted for in the official statistics available. To solve this, a bio-economic modelling approach would be advised. The third issue relates to the role of recreational fisheries. Concerning the former, some benefits can be captured in accounting for cultural ecosystem services, potentially through using travel cost or stated preferences approaches. Regardless, the role of recreational finfish catch on the capital stock of finfish biomass should be further investigated through the use of bio-economic modelling and sectoral surveys. Finally, the fisheries sector is highly relevant from a supra-national perspective, as it is the result of complex policy and institutional interactions between different countries using the same scarce resource. This policy perspective could be further considered through the inclusion of fishing quotas and international agreements within the valuation method. This is particularly relevant to precisely account for the contribution to national economies of finfish catches outside of the national EEZs. Also, the accounting and reporting perspective employed in different countries should be harmonised in order to avoid double-counting. In this regard, there is a specific need to define the fleet on which to focus on in national accounts (only the national fleet, or both national and foreign fleets actively fishing in national boundaries) and the landing sites to be considered for calculating the profits of the industry (only the national landing sites, or both national and foreign sites).

8.2.2 Waste (nutrient) remediation

Evidence supporting the valuation of waste (nutrient) remediation in marine and coastal environments includes the use of a production cost approach, replacement cost approach, cost avoided approach, payment for ecosystem services, and revealed and stated preferences primary valuation studies. As in the case of natural hazard protection, a replacement cost approach is considered more coherent with natural capital accounting principles. Regardless, the same modelling effort and further research would be required as the waste and nutrients flowing into marine and coastal habitats are the effects of complex interactions between natural and human activities, mostly occurring on land. Therefore, to apply a replacement cost to account for coastal and marine habitats waste breakdown, detoxification and burial/removal/neutralisation, the following datasets would be needed:

- Data related to streams of waste and nutrients from different point and diffuse sources. This information would be necessary to assess and spatially track the quantity and characteristics of waste and nutrient loads reaching marine and coastal environments.
- Data on the economic and social activities generating the flow of waste and nutrients. This information would be relevant to explore whether to use different monetary proxies to value different types of waste and nutrients discharges, as the remediation treatments might differ between pollutant and sources types.
- Data on economic costs of man-made alternative options to remediate waste and nutrient loadings, e.g. the costs of wastewater treatment.
- Data on the retention rates of different habitats for different waste and nutrient components and models on the diffusion of waste and nutrients across coastal and marine habitats. Different habitats have different retention rates and thresholds. Therefore, the ecosystem service of remediation is provided concurrently by bundles of natural habitats and the interactions between them need to be physically and spatially defined.

This information would feed into an integrated bio-economic model able to assess the service provision at different geographical scales.

It is worth noting that the replacement cost approach has been used in the literature (see La Notte et al. 2017) to account for remediation of nitrogen considering the cost needed to replace natural habitats providing the service with constructed wetlands. Also, shadow prices for waste and nutrient remediation could be directly applied to the quantities obtained using remediation rates of the different habitats. Regarding remediation rates, the high scientific uncertainty should be highlighted. It is therefore recognised that some form of value transfer method could be partially or totally employed. Finally, ideally the levelised cost approach could be applied in the case of specific waste and nutrient treatment (e.g. wastewater treatment plants).

8.2.3 Natural hazard protection

More accurate valuation should be grounded on complex bio-economic modelling, including several characteristics of marine and coastal environments coupled with social and economic attributes. A replacement cost approach is generally considered more in line with natural capital accounting requirements, as it values the cost of replacing natural habitats with man-made equivalent protection. Regardless, it is demanding in terms of data needs and modelling efforts, and further research is required. In particular, to apply this approach to the accounting of the service in UK marine and coastal and marine environments, the following sets of data would be needed:

- Disaggregated data related to the extent and position of different natural habitats. Indeed, habitats provide the service with differentiated intensity.
- Suitable data on condition indicators specific to different natural habitats. The capacity of providing natural hazard protection is strictly linked with different biophysical characteristics of natural habitats, and each habitat requires a suitable set of condition indicators allowing the estimation of service provision intensity. The indicators should be also suitable to link physical assessment to monetary valuation of the service.
- Disaggregated data related to the position and extent of existing man-made alternatives. This would be needed to isolate the service provision from natural habitats.
- Disaggregated data on the economic activities and infrastructures existing on land protected by natural habitats. This would be essential to estimate the monetary value of the service, as it would allow an estimate of the economic and social damages that would occur if natural habitats did not provide protection. Data granularity would be relevant to spatially link the value of land protected to different natural habitats.
- Biophysical models to assess to the risk and vulnerability to natural hazard. Risk and vulnerability considerations would be relevant to precisely appraise and calibrate the assessment of natural hazard protection provision. In particular, data on flooding risk, water catchment areas, and climate conditions would be needed.

This information should be linked through the development of an integrated bio-economic model able to assess the service provision at different geographical scales.

8.2.4 Renewable energy – offshore wind farms

The use of data sourced directly from financial documents provided by wind energy generation companies enables more precise estimates of resource rent than using aggregate data from the SNA, which nevertheless remains a suitable option in the absence of detailed data. Indeed, in the SNA, the electricity generation sector is not disaggregated by sources and by industrial activities. Regardless, not all the companies operating in the sector have financial documents readily available and considering only the major producers might slightly bias the resource rent calculation. Also, the production of wind energy by terrestrial and offshore farms might incur different costs. Therefore, a disaggregation allowing the isolation of companies operating only offshore or allowing the isolation of the quantity and costs of electrical energy generated offshore, would be advised. Alternative methods for valuing the ecosystem service of offshore wind energy provided by marine and coastal environments could include isolating potential royalties or levies paid directly by extractors to the government or using a so-called “levelised cost” approach. Regarding royalties and levies, it has proved difficult to disentangle those from general taxation and subsidies, therefore a more disaggregated reporting by the industry would be needed. Regarding the levelised cost approach, as suggested by the ONS, it requires the calculation of an average cost over the lifetime of the plants per unit of electricity generated, reflecting the cost of building, operating and decommissioning the plants. In addition, another suitable option for accounting purposes would be to source the output sold by the sector through multiplying energy generation data by the price of energy on the market. This approach could be perfected by the availability of data specifically related to offshore wind energy prices and by disaggregating the different users (in terms of industrial vs non-industrial, and quantity of energy used). Finally, there is some evidence regarding the interaction between wind farms and the provision of other ecosystem services (e.g. finfish and shellfish, carbon storage) which should be further explored from an accounting perspective.

8.2.5 Aggregate extraction

A resource rent approach could return more precise estimates if data on output and revenues could be sourced directly from the financial documents of companies operating in marine aggregates extraction. Moreover, market prices are often not disclosed by the companies. Regardless, the same alternative methods considered for wind energy generation could be applied to accounting for marine aggregates extraction. These royalties are separately accounted for but disentangling those from general taxation would require further investigation. Also, a levelised cost approach could be potentially explored for aggregates extraction companies. Finally, it is worth noting that some of the aggregates extracted are used for beach replenishment, therefore interacting with the ecosystem service provision of coastal habitats.

8.2.6 Places and Seascapes for recreation and nature watching

The value of recreational opportunities is used in this report as a proxy for the valuation of cultural ecosystem services provided by marine and coastal environments. The simple travel cost approach is based on data collected in the Monitor of Engagement with Natural Environment Survey (MENE) which includes only England (Natural England 2019). Therefore, estimates for the UK are obtained scaling up results from the MENE.

The value of cultural ecosystem services provided by coastal and marine environment can be better elicited considering two main points:

1. **Obtain a more accurate estimation of recreational values**

   The simple travel cost method could be expanded by considering more sophisticated modelling approach including spatial components and control variables. This would potentially allow an estimate of the value of all visits to marine and coastal environments, including visits with no expenditure reported.

   The potential to integrate different UK surveys should be further explored. In particular, Scotland's People and Nature Survey (SPNS), the Outdoor Recreation section of the National Survey for Wales (NSW), the Great Britain Day Visits Survey (GBDVS), and the Watersports Participation Survey (WPS) contain information similar to the MENE (Natural England 2019). Integrating information from those surveys would allow better assessment of UK level hours spent and number of visits to marine and coastal environments, but also more precisely disaggregate the type and the reason of visits. It is worth noting that the definition of what is considered a recreational visit can differ across surveys, therefore integration is not straightforward.

   Finally, the possibility to spatially disaggregate recreational visits between different habitats should be further explored as it would allow to attach recreational values to specific marine and coastal habitats.

2. **Include other cultural ecosystem services**

   Cultural ecosystem services provided by coastal and marine places and seascapes embrace a much broader class of immaterial benefits as aesthetic value, health and psychological wellbeing effects, sense of place, experiential interactions, intellectual and educational activities, spiritual and symbolic interactions. These benefits constitute an important part of the cultural ecosystem services provided but are difficult to quantify (both in physical and monetary terms) in a natural capital accounting framework. Indeed, it is difficult to disentangle these more immaterial benefits from general recreational benefits, and it is complex to attach a monetary value due to the lack of exchange value proxies.

For some ecosystem services listed (e.g. intellectual and educational, health effects), information sourced from education and research institutions and national health institutions could help to quantify the benefits provided, and ideally elicit a monetary value through travel cost, cost avoided and replacement cost approaches. Another possible approach could refer to the integration of information sourced from different recreational surveys in order to disaggregate the different types of visits and model the share of benefits attributable to different ecosystem service types. Moreover, the possibility to use methodologies eliciting welfare values (e.g. stated preferences techniques) that are more suitable for capturing the value of immaterial benefits should be further explored in order to make it coherent with natural capital accounting principles. Finally, a suitable approach could consider expanding natural capital accounts with satellite accounts containing non-monetary indicators and information able to capture immaterial cultural ecosystem services and health related benefits.