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Accounting for benefits from natural capital: Applying a novel composite indicator framework to the marine environment

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ABSTRACT

Increased emphasis on the natural capital approach in the UK has led to greater demand for methods that link economic sectors with elements of natural capital, and that can provide evidence for sustainable management of the environment. However, factors describing the supply of benefits, and their links with economic sectors, are not well defined.

This study develops a novel framework that investigates how the combination of different forms of capital (natural, financial, social, manufactured or human) provide a potential supply of benefits, and how changes in quality or extent of natural capital affects supply. Factors affecting the delivery of benefits are analysed, and indicators for each factor are selected. Indicators are ranked and weighted, and benefit supply is represented as a novel, composite index. The composite supply index is then linked as an input to a related economic sector.

This framework is applied for the first time to four benefits from the marine environment in the UK: seafood, offshore wind energy, wildlife watching and water sports. The approach is compatible with national accounts, natural capital accounts, and established ecosystem service classifications. This study shows how linking economic sectors with benefits can provide new evidence in support of marine management.

1. Introduction

The importance of integrated assessments for natural capital assets, ecosystem services (ES) and the value of environmental benefits has been well established (Hooper et al., 2019). Yet, most applications have been to terrestrial environments (Liquete et al., 2013), and there are significant data gaps which hinder integrative marine natural capital assessments. The marine and coastal environment provides unique benefits to humans and society. Energy, seafood, raw materials and recreational enjoyment are all ways in which it is used and enjoyed by humans. Realisation of these benefits relies on both natural and human systems, and within these systems, living and non-living elements. Benefits from the marine and coastal environment can also contribute to the economy. However, pressure from humans is degrading the quality and quantity of assets in the natural environment (Millenium Ecosystem Assessment, 2005; UK National Ecosystem Assessment, 2011), and the complexity of interactions between human and environmental systems only increase the difficulties of managing its use. The natural capital approach can be used to overcome these difficulties; environmental extent and condition are integrated into decision-making, as are the inter-dependencies between the environment, economy and society. Worldwide efforts have therefore been made to measure natural capital assets, monitor ongoing condition, link with ES, and measure benefits they can provide.

Natural capital combines biotic organisms and non-living parts of the environment with ecological functioning, environmental processes, land mass, air and water (Costanza and Daly, 1992; Mace, 2019; Mortimer et al., 2017; Natural Capital Committee, 2014; Office for National Statistics, 2018a; United Nations, 2014a). Elements of natural capital working together supply ES, and produce benefits when combined with other forms of capital (i.e. human, social, manufactured, financial) (Costanza et al., 2014; Mace, 2019; Natural Capital Committee, 2017). Measuring assets is critical for the application of the natural capital approach (Natural Capital Committee, 2017) but no universally agreed upon methodology exists (Hooper et al., 2019).

A consistent set of rules and principles is needed for all forms of accounting, to prevent related transactions being recorded on different bases, at different times or with different values, thus making accounting

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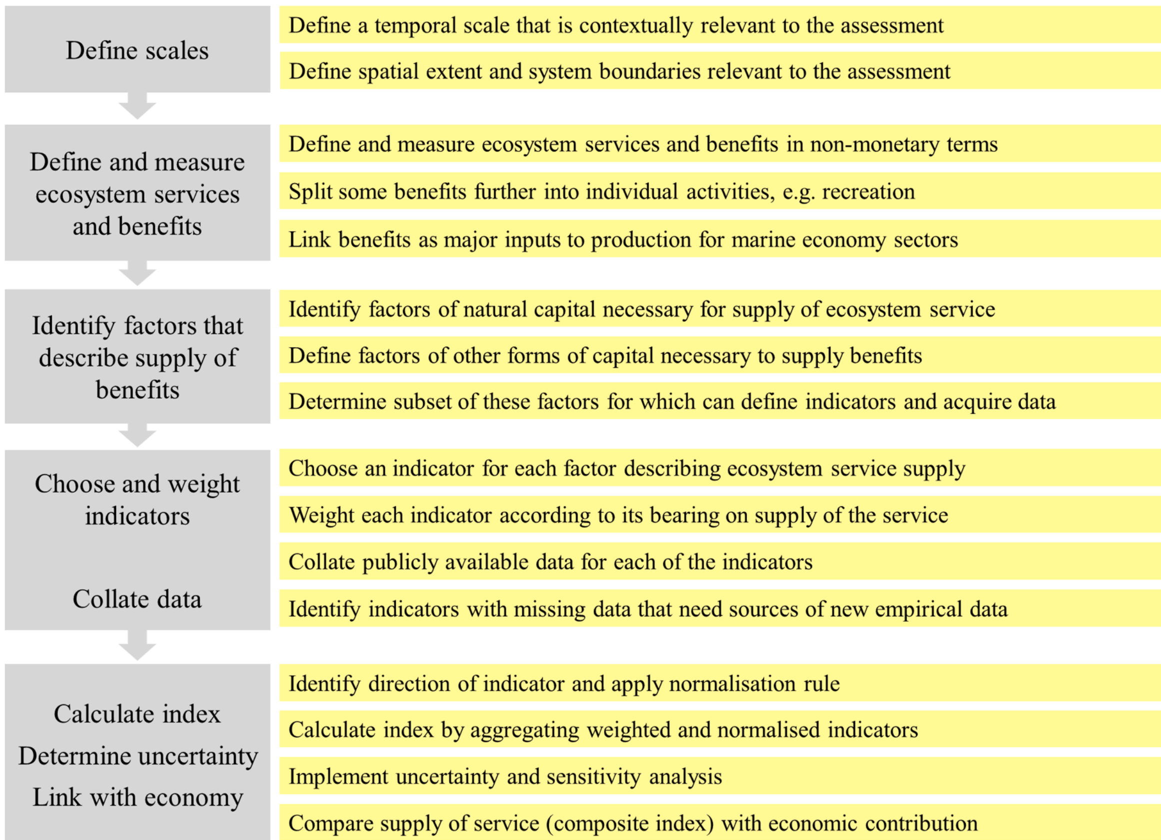

Fig. 1. A composite index for the supply of benefits.

information less useful (United Nations, 2014a). Approaches to measuring environmental assets, changes in the stock of assets, and the flow of materials or energy between the economy and the environment, are formally defined by the UN's System of Environmental-Economic Accounting Central Framework (SEEA) (United Nations, 2014a). The SEEA applies financial accounting principles (i.e. those of the System of National Accounts (United Nations, 2009)) to the environment (Dickie and Neupauer, 2019). An additional guidance framework, Experimental Ecosystem Accounting (EEA), complements the SEEA (United Nations, 2014b). The EEA includes separate accounts for ecosystem services, ecosystem extent, ecosystem condition and monetary assets. Natural capital accounts are an extension of ecosystem accounts defined under the EEA (Dickie and Neupauer, 2019; United Nations, 2014b). Assets defined within natural capital accounting have a wider scope than ecosystem assets, in that they also include environmental resources (e.g. materials and energy). Natural capital accounting (NCA), developed to measure assets and monitor change, record the condition of assets in terms of stocks and flows, in both monetary and non-monetary terms but the language and structure of NCA follows that of the SEEA, and is therefore aligned, and so fits with national accounting principles.

Marine natural capital accounts have been developed in Australia (Australian Bureau of Statistics, 2017), Sweden (Steinbach, 2017), Costa Rica (Gutiérrez-Espeleta, 2017), the Netherlands (Ruijs et al., 2018) and the UK (Office for National Statistics, 2016a; Thornton et al., 2019). However, non-natural capital involved in producing benefits is not explicitly identified in the SEEA, EEA or NCA, because their objectives are to measure environmental assets, with manufactured and human forms of capital already accounted for within the production boundary of national accounts. In addition, the capacity to supply cultural services cannot be measured in the same manner as, for example, a physical 'stock' of fish. The linkages between natural and other forms of capital that are crucial for benefit supply are therefore not reflected in NCA or in economic results, and it is difficult to make a link from assets to economic production for all types of marine benefits. Initiatives such as asset and risk registers, which use indicators to measure the condition of assets, have therefore developed alongside, rather than as an integral component of, NCA (Hooper et al., 2019).

Indices complement NCA by specifically acknowledging the contribution all forms of capital to the supply of benefits, and aid understanding of ecosystem changes (Hattam et al., 2015). Examples include the Ocean Health Index (Halpern et al., 2012), Costa Rica's Nature Index (Barton et al., 2014), and those from Canada (Alam et al., 2016) and the USA (Villamagna et al., 2014). Natural capital indicators have been particularly established in the United Kingdom (UK) (Ashley et al., 2018; Deane and Walker, 2018; McKenna et al., 2019; Scottish Natural Heritage, 2019). As a result of these applications, and the government's commitment to producing indicator sets and NCA (Curnow, 2019; HM Government, 2018; Natural Capital Committee, 2019, 2014; Office for National Statistics, 2018b, 2018c, 2015; Sunderland et al., 2019; The RSPB, 2017; Thornton et al., 2019), the UK was chosen as a case study for this research.

There are still several shortcomings of existing approaches for marine NCA. Firstly, estimates of natural capital condition at a national level refer to provisioning, regulating or cultural services in general (Scottish Natural Heritage, 2019; Tillin et al., 2019) or focus on marketable products rather than a full complement of ES (Office for National Statistics, 2018c, 2015). Secondly, many approaches have focussed on terrestrial environments, and as such existing methodologies are inadequate for marine and coastal environments (Hooper et al., 2019). In addition, a shortage of marine natural capital valuations inhibit the use of ES values to support decision making (Börger et al., 2014). Indeed, while the Natural Capital Asset Index for Scotland included four indicators for the condition of natural capital in coastal margin habitats, it has yet to be extended to the marine area (Scottish

Natural Heritage, 2019; Tillin et al., 2019). Thirdly, due to the complexities of measuring and valuing them, the supply of recreation and cultural benefits have largely been absent from valuations (Hicks, 2011; Luisetti et al., 2014). Despite the development of numerous indices that include measures of the environment as aspects of human wellbeing, wealth or social progress (European Commission, 2016; Lange et al., 2018; OECD, 2020; Social Progress Imperative, 2020), such indices are not sufficiently detailed to measure the factors of natural capital that are critical to realising environmental benefits. Lastly, few studies have linked natural capital with benefit supply and economic production. Production function methods, that recognise the role of natural resources in the production of goods, can be used to explore how changes in benefit supply affect economic output (Barbier, 2007; Cordier et al., 2014; Guerry et al., 2012; UK National Ecosystem Assessment, 2011). Although some studies have linked elements of natural capital to the economy (Klinger et al., 2018; Mancini et al., 2017), we find only two applications of natural capital integrated with economic production (Allan et al., 2019; Ochuodho and Alavalapati, 2016). In fact, there is little clarity on how this can be done on a national scale, or how it can be tailored to marine natural capital specifically. An approach linking the environment with the supply of benefits to economy would therefore improve the evidence base for marine NCA (Thornton et al., 2019, p. 55).

It is clear to us that a systematic approach is needed. This approach would describe the supply of marine benefits through the combination of all types of capital, and link them with economic production. This study therefore defined the capacity of a system to supply specific benefits through a combination of natural and human factors. To the authors' knowledge, this is the first work that attempts to measure different environmental benefits by describing them as the product of different forms of capital: natural capital as described in environmental accounts or NCA, as well as inputs from within the production boundary of national accounts. Indicators were chosen for each of the factors and a composite index was calculated that described the capacity to supply benefits (Section 2). The related economic sectors were identified, and the economic contribution of these benefits was estimated. The application of this approach was demonstrated with case studies from the UK for four marine benefits (Section 3). The implications and limitations of this approach are discussed in section 4.

2. Methodology

The approach used in this work was based on multi-criteria assessment and composite indicators. It integrated the approaches used by three previous works: a composite natural capital index (Scottish Natural Heritage, 2019), an approach to determining supply-side indicators for cultural ES (Tratalos et al., 2016), and a multi-indicator framework for cultural benefits (Villamagna et al., 2014). Several major modifications were then made. Firstly, the marine and coastal environment was made a specific focus, as there are few examples of integrative frameworks in the literature. Secondly, a detailed assessment of individual ES was undertaken (i.e. this approach went beyond the high-level categories of provisioning, cultural or regulatory services), with recreation and leisure services sub-divided into specific activities. Finally, it specifically included a mixture of financial, social, manufactured and human capital as well as natural capital. The approach is summarised in Fig. 1, and further details of each step are provided in the remainder of this section.

2.1. Define temporal and spatial scales

The temporal scale over which natural capital and ES are to be measured was defined, and set the context for the estimates. Temporal scale considered both the relevant time frame for the ES and benefits being measured, as well as the availability of economic data. Seasonal effects may also be a factor in defining temporal context; commercial fisheries may operate in all seasons, while some recreation activities occur primarily during the summer months. Most data sources used for this work were available annually, with the most recent and complete grey literature being from 2018. The estimates focused on changes in supply and economic contribution in the short-term in order to demonstrate the approach and to maximise the likelihood of available indicator data. Results for the case studies were therefore estimated annually over five years, from 2013 to 2018. The frequency with which data was available for specific indicators is discussed further in Section 2.4.

The system boundary within which services were supplied was also defined. This was broadly determined in relation to the ES being measured and relevant economic results. For example, international agreements and other management measures restrict the area of UK waters over which fish can be caught. However, a spatial boundary based on management or human-related factors does not necessarily match the boundary of a particular ecosystem or habitat type (i.e. fish prefer certain habitats, and are not restricted to the area over which the UK can fish). A well-defined inland boundary was also necessary, because some benefits also take place on the coastal margins but rely on aspects of marine natural capital in order to take place (e.g. surfing) (Eftec, 2015; United Nations, 2014b). Consequently, an approach that combined different habitat types with management, economic and legislative boundaries was required.

Spatial limits therefore combined; the outer limit of the UK Exclusive Economic Zone (EEZ) (Hooper et al., 2018) and the landside limit of coastal margin habitats based on UK National Ecosystem Assessment (2014) habitat classifications. This approach was chosen for its alignment with NCA (Office for National Statistics, 2016b), and because coastal margin habitats, such as sand dunes and cliffs, are included. Although coastal and marine habitats were included here, it was important be able to separate them because the coastal margins can be used for both marine and terrestrial activities (Natural Capital Committee, 2019). Marine and coastal margin NCA are also measured separately (Office for National Statistics, 2018b, 2016b; Thornton et al., 2019). One shortcoming of the UKNEA typology is that it includes only two littoral habitats, while also combining splash-zone and intertidal habitats, which leads to a lack of clarity as to which are marine habitats and which are coastal (Hooper et al., 2019). The UKNEA typology was used for this study because coastal and marine habitats can be delineated, while acknowledging that a different typology might be more appropriate if an analysis was required by habitat type.

2.2. Define and measure ecosystem services and benefits

When combined with other forms of capital, ES can provide benefits to people. For example, the presence of dolphins is an ES, but wildlife watching (which requires other capital inputs) is a benefit. Since the intention of this work was to link the environment to the economy, the first step was to define and characterise ES. The resulting benefits were then defined and linked from ES to production by an economic sector.

Both abiotic and biotic components of natural capital supply ES (Culhane et al., 2018b; Haines-Young and Potschin-Young, 2018), and use of the environment for economic production can be extractive or non-extractive (den Butter and Hofkes, 1995; Klinger et al., 2018; Ruiz-Frau et al., 2015). This categorisation of ES by their biotic or abiotic nature and their extractive or non-extractive use was used as the initial framework by which to characterise them. Specific ES within these four categories were defined using the marine-adapted Common International Classification for Ecosystem Services (CICES) typology (Culhane et al., 2018a, 2018b; Haines-Young and Potschin-Young, 2018). Though many classification systems for ES have been developed, the CICES framework is comprehensive in that it includes both biotic and abiotic services, and the work by Culhane et al. (2018a, 2018b) adapted the biotic side of the typology specifically for marine ecosystems.

Recreation and leisure activities have been customarily grouped

Table 1
Marine ecosystem services categorised by extractive and biotic properties. Adapted from (Culhane et al., 2018b, 2018a; Haines-Young and Potschin-Young, 2018; Klinger et al., 2018; Marine Management Organisation, 2014, 2012).

	Extractive	Non-extractive
Biotic	Wild caught seafood[1] Seafood from wild plants and algae Cultured seafood[2] Plants and algal seafood from aquaculture Biotic raw materials[3] Genetic material Biofuels[4]	Biotic regulatory & maintenance services[6] Recreation & leisure; Angling Wildlife watching Scuba diving and snorkelling Rock-pooling Wildfowling Other biotic recreation & leisure services Cultural, spiritual & historic appreciation involving interaction with marine or coastal biota [7]
Abiotic	Raw materials for construction (aggregates) Abiotic raw materials[5] Offshore oil and gas reserves Offshore wind energy Tidal and wave energy	Abiotic regulatory & maintenance services Recreation & leisure; Beach pastimes and sports Coasteering and coastal walking Sport on water (paddle sports, sailing, wind-surfing, surfing or bodyboarding) Motor-boating and recreational boat trips Sea swimming Other abiotic recreation & leisure services Cultural, spiritual & historic appreciation involving interaction with marine and coastal landscape

Notes
[1] Seafood from wild animals.
2 Animal seafood from aquaculture.
3 Raw materials and materials for agriculture and aquaculture.
4 Plant, animal and algal based biofuels.
5 Raw materials for food – e.g. salt, fresh water.
[6] Service numbers 10–23 (Culhane et al., 2018b).
[7] Service numbers 25–33 (Culhane et al., 2018b).

together as a single service in generic classification frameworks for ES, as it is impractical to list the many different recreation activities possible. However, the intention of this approach was to identify the factors needed for benefits from these services to be delivered, and link them with economic production. Therefore, detailed categorisation of recreation was required, that matched the same level of disaggregation as their related economic sectors. Recreation and leisure services were therefore split using the recreation activity types defined by the MMO (Marine Management Organisation, 2014, 2012). Marine ES are thus defined and categorised in Table 1. Since there was a direct link from the services, this classification was also applicable for the resulting benefits.

The way each benefit was measured was defined, initially in non-monetary terms. In most cases, the units of the benefit will be similar, if not the same, as those used to measure the ES itself. For example, the stock biomass of wild fish (the ES) and the quantity of landed fish harvested from that stock (the benefit) can both be measured in tonnes. Following Barbier (2007) and Cordier et al. (2014), benefits were then identified as inputs to production for relevant economic products. A select number of benefits were linked as major inputs to production for economic sectors in the UK (Fig. 2) with their UK Standard Industrial Classification (SIC) code. The SIC code, synonymous with NACE codes in the EU, identifies and categorises economic activities in the UK economy. For example, participation in water sports could be linked to the production of 'Sports, amusement and recreation services' in the market economy (SIC 93).

In determining the links between benefits from marine ES and economic sectors, both direct and indirect economic contributions were considered. For example, tourists may pay for surfing lessons while visiting the coast, whereas residents may participate in surfing for free because they own their own equipment, but might contribute to the economy in other ways while taking a surfing trip (for example, through parking payments or food and drink purchase). Marine ES also provide non-market benefits that cannot be linked directly to production in the economy (for example, the health and wellbeing benefits of recreational activities) and these non-market benefits are not considered within this study. However, by initially defining non-monetary measures of the benefits (e.g. total numbers of participants in surfing) data relevant to wider non-market benefits was also captured.

2.3. Identify factors that describe the flow of benefits

. The next stage was to identify factors that describe a supply of environmental benefits. Factors are defined here as the attributes of natural, financial, social, manufactured or human capital that, in combination, lead to the supply of environmental benefits. Factors can include habitat types, ecosystem conditions, ecological processes, climatic conditions, geographical features, infrastructure, manufactured goods or human behaviours. This definition is approximately equivalent to the 'factors determining supply' defined in the draft revision of the EEA (United Nations Statistics Division, 2020), and is compatible with the approach applied in NCA for the UK marine and coastal environment (Thornton et al., 2019). For example, the principal natural characteristic in the provision of tidal energy is a tidal flow of sufficient strength. In addition, human, financial, social and manufactured capital are required for any benefits to be realised, which, to continue the tidal energy example, include the design, construction and operation of a tidal energy installation. Here, capacity derived from non-natural forms of capital that act as direct inputs necessary for the service to occur, or are necessary to realise the benefits, are referred to as human-derived capital (HDC) (Jones et al., 2016). For example, one form of HDC for the production of marine energy is the capacity of the tidal barrage or wave energy device. Thus, the factors for the supply of benefits from tidal energy are sufficient tidal speed or volume (natural capital) and the operational and planned capacity of marine energy installations (HDC).

The factors of natural capital and HDC required in order to deliver specific benefits were identified from peer-reviewed research and grey literature. For example, surveys of visitors to beaches (Peña-Alonso et al., 2017), coastal environments (Elliott et al., 2018) or other habitats (Avila-Foucat et al., 2017; Jobstvogt et al., 2014) are available in the ecosystem services literature. Research describing the supply of commercial or recreational fishing are also not uncommon. However, identifying factors of supply for some other benefits may be more difficult, and literature may need to be sourced more creatively. For example, geophysical attributes (Possner and Caldeira, 2017) and maintenance (Faulstich et al., 2011) are described in engineering publications as important factors in the operation of wind turbines, and were interpreted in this work as factors of benefit supply. However, data was not available for all of these factors (e.g. number of people that take part in recreation due to good weather), so a sub-set of measurable factors were selected for which data is available, and these were used to apply the method outlined below. The factors were linked to indicators and form the basis of the composite supply index.

2.4. Select indicators, collect data and determine indicator weights

Indicators for each factors were determined through a literature review. Good quality indicators demonstrate a clear cause and effect relationship, while being measurable, understandable and linked to decision making (Kandziora et al., 2013; Niemeijer and de Groot, 2008). For example, fishing yield is an appropriate indicator for benefits from provisioning services provided by the marine ecosystem because it can be directly linked to human consumption (Maes et al., 2016). Conversely, phytoplankton biomass density is unsuitable as an indicator for food provision because it is not directly linked to human consumption. However, it does support this ES and therefore indirectly contributes to the supply of food (Broszeit et al., 2017), and thus provides relevant information on the status of the underlying natural capital asset. Data availability is also an important factor when selecting

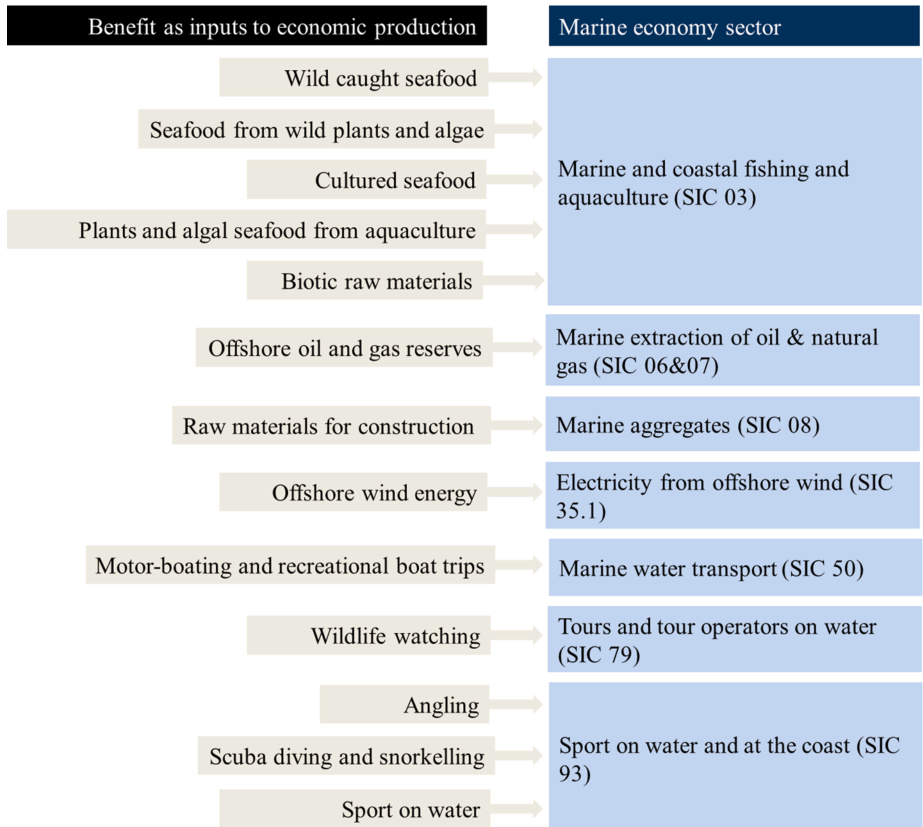

Fig. 2. Selected marine benefits as direct inputs to sectors of the marine economy. Source: Stebbings et al (2020). Adapted from Klinger et al. (2018) & Haines-Young & Potschin (2010).

indicators. For example, detailed landing data for fish are collected regularly, but information about fish stocks (which bring about the supply of fish) are compiled at a much higher spatial scale and gathered less often.

Where possible, existing national and international targets were used to guide the selection of indicators and to obtain the relevant data (Broszeit et al., 2017). For example, the Ocean Health Index and the OSPAR Commission monitor indicators relevant to the delivery of marine ES (Halpern et al., 2012; OSPAR, 2017a). Grey literature was also used (e.g. modelling environmental attributes for marine recreation (Marine Management Organisation, 2014)). Published economic data was used to determine the sectors that use marine ES, and to quantify the contribution of that sector to the wider economy. However, sources of data are likely to vary in timeliness; weather data can be available hourly, while macroeconomic indicators are published quarterly or annually in the UK by the Office for National Statistics (ONS). Some analyses are published with a lag of up to four years. In practice, there is likely to be an iterative process between the choice of indicator and assessment of data that is available to support it.

Each indicator was weighted in terms of importance. The weighting of indicators in a composite index should be carefully considered, because the resulting index will be highly sensitive to the weights used. However, despite the wide application of composite indicators, no agreed methodology exists (Nardo et al., 2005). Indicators can be selected and weighted in a variety of ways; either informed by a survey, by statistical analysis (where relationships between the factors are analysed for their relationship with the composite outcome), or by using participatory approaches (using expert or stakeholder opinion to link factors with the composite outcome) (Alam et al., 2016; Blanc et al., 2008; Nardo et al., 2005). Weighting derived using survey data or statistical analysis, such as regression analysis, factor analysis or principal component analysis, was preferred because observed data was used to relate factors to the outcome. In the absence of data that supports an alternative weighting, or if all indicators can be considered equally important, they can be given equal weighting. However, equal weighting can obscure an absence of empirical data (Dobbie and Dail, 2013), and introduce statistical bias (Blanc et al., 2008).

In this application, indicators were weighted using principal component analysis (PCA) (Gómez-Limón and Riesgo, 2009; Kotzee and Reyers, 2016; Nardo et al., 2005), using R Studio (version 1.456) and Stata 15 (RStudio, 2019; StataCorp, 2017). The indicators were inspected for normality using Lilliefors (Kolmogorov-Smirnov) test, and their suitability for PCA were tested using Barlett's Test and the Kaiser-Meyer Olkin (KMO) statistic. The correlation between indicators was analysed using a correlation matrix. Components with an eigenvalue higher than 1 were retained. Factor loadings were generated for each of the indicators on the components, and a Kaiser Varimax rotation was used to reduce the number of highly loaded individual variables in the analysis (further details given in the Supplementary Material).

2.5. Calculate composite index and link with economic contribution

Once data for the indicators for each ES were obtained and the weightings determined, the composite ES index was calculated. The approach to calculating the composite index was based on guidance published by the OECD and the European Commission (Nardo et al., 2005; OECD, 2008). This study applied normalisation and aggregation methods developed for sustainability indices (Gan et al., 2017; Luzzati and Gucciardi, 2015), because although there was no similar study available for applications to ES or natural capital, the indices for sustainability similarly combine environmental and human-orientated indicators. The methods used to calculate this index are also broadly similar to those used to compute the Social Progress Index, in that indicators are first standardised, then weighted using principal component analysis (PCA) (Social Progress Imperative, 2020; Stern et al., 2020). The direction of each indicator was first defined. For example, building developments might have an inverse (negative) effect on the supply of cultural ES and cause the composite index to decrease, whereas safe levels of pollutants might induce a positive effect and cause it to increase (Scottish Government, 2019). The indicators were then normalised, to convert them to the same unit and scale before they were aggregated (Dobbie and Dail, 2013; Luzzati and Gucciardi, 2015). Indicators were normalised using the percentage of annual differences over consecutive years from a base year (Nardo et al., 2005; Scottish Government, 2019); each individual indicator was transformed according to Equation (1), with normalised indicator N_i^t, absolute measurement of the indicator i^t at time t and absolute measurement in the base year i^0;

$$N_i^t = \frac{i^t - i^0}{i^0} \times 100 \tag{1}$$

The resulting normalised indicator was dimensionless, and showed the annual change for each indicator rather than its absolute value. The base year was 2013, except for indicators with high levels of annual variability (e.g. weather variables) for which the 20 year average was used. In the case of existing international indicators (e.g. OSPAR targets) the closest baseline year to 2013 was used.

Finally, the indices were aggregated. There are two broad approaches to aggregation for composite indices; (i) compensatory methods, where decline of one indicator can be compensated by the increase of another so that indicators for different factors can be substituted, or (ii) non-compensatory methods where substitution is unacceptable (Gan et al., 2017; Nardo et al., 2005). A non-compensatory approach was applied, under the assumption that natural capital indicators were non-substitutable. For example, supply of seafood relies on stock biomass and fishing pressure, but if stock declines, an increase in effort might temporarily maintain supply, but an increase of the number of fishing boats cannot compensate for reduced environmental capacity in the long term. The indicators were aggregated to form the composite supply index S^t using a method adapted from the non-compensatory aggregation function by Pollesch & Dale (2015), given in the Supplementary Material. The sensitivity of the composite index to the weighting of each indicator was then tested by comparing the results under different aggregation scenarios; under equal weighing and under compensatory aggregation.

Resource rent was chosen as the measure of economic contribution for benefits supplied by ES. Resource rent measures the gross return on an environmental asset based on the gross operating surplus of a related industry (United Nations, 2014a). The disaggregated sectors of the economy that specifically related to activity within the extent of marine and coastal natural capital were used as the basis of the analysis (Stebbings et al., 2020). The residual value resource rent was then estimated for each related sector using the gross operating surplus from published and disaggregated input–output tables, following the approach outlined by the ONS (Office for National Statistics, 2018c; Thornton et al., 2019). The composite index was reported for each year and compared to the resource rent.

3. Case study

A case study of four benefits was carried out to explore how they might be linked to economic sectors. Four benefits from Table 1 were chosen to represent each of the ES categories (biotic, abiotic, extractive, and non-extractive). These were, respectively, wild caught sea fish and shellfish[1], offshore wind energy, wildlife watching and water sports. These benefits were chosen as case studies because of their links to tourism, fisheries and energy, which represent important sectors in the marine economy (Stebbings et al., 2020).

[1] Hereafter referred to as ‘seafood’

Table 2
Non-monetary flow of benefits from the marine and coastal environment. Source: (Hattam et al., 2015; Mills and Cummins, 2013; Office for National Statistics, 2019h; Peña-Alonso et al., 2017; Ryan et al., 2018; Whiteley et al., 2016; Arkenford, 2018; Arkenford, 2017; Department for Business Energy Industrial Strategy, 2019; Dunne, 2019; ICES, 2019; Marine Management Organisation, 2019; Office for National Statistics, 2019g; Radford et al., 2019; Sport England, 2017; England, 2016; Sport England, 2013; VisitBritain, 2019; VisitBritain, 2018a; VisitBritain, 2018b; VisitBritain, 2015)

Benefit	2013	2014	2015	2016	2017	2018
Wild caught seafood (Thousand tonnes)	688	818	769	762	788	760
Offshore wind energy (TWh)	11.5	13.4	17.4	16.4	20.9	26.7
Water sports (Million days of participation)	36.6	36.6	41.3	44.0	38.6	50.4
Wildlife watching (Million days of participation)	5.0	12.7	9.3	9.2	8.1	7.2

Table 3
Resource rent for selected marine and coastal benefits.

Resource rent (£m, 2018 prices)	2013	2014	2015	2016	2017	2018
Wild caught seafood	286	309	280	259	263	253
Offshore wind energy	344	689	921	866	1,075	1,365
Water sports	193	218	254	253	255	256
Wildlife watching	60	59	65	65	65	65

3.1. Non-monetary measures of benefit flow

The non-monetary flow of the four benefits is given in Table 2, with further details given in the Supplementary Material.

3.2. Resource rent

The supply of non-monetary benefits act as inputs to economic sectors, based on those in Fig. 2. Although numerous benefits from ES contribute in a minor way to the marine seafood sector, it was assumed that seafood, electricity generated from offshore wind, water sports participation, and wildlife watching participation provide the most significant inputs to production of the marine seafood (SIC 03), offshore wind (SIC 35.1), sports services (SIC 93) and tour services (SIC 79) sectors respectively.

The resource rents of the seafood, offshore wind, water sports and wildlife watching sectors were estimated by disaggregating them from other activities in the economic results. Seafood was disaggregated from fisheries and aquaculture, offshore wind from total electricity, water sports from sports services, and wildlife watching (in the marine and coastal environment) from tours and tour operators. The gross operating surplus of these sectors was estimated using the input–output table for 2013–2015 (Office for National Statistics, 2017a, 2018d, 2019a) and used additional sectoral information from the Annual Business Survey (Office for National Statistics, 2017b). There were no published input–output tables available for 2016 to 2018, so the results were estimated using the annual sectoral growth for agriculture, production, and services (Office for National Statistics, 2019b, 2019c, 2019d). Resource rent was expressed in 2018 real terms using the gross operating surplus & mixed income deflator time series (Office for National Statistics, 2019e). The resource rent for each case study benefit is given in Table 3.

Table 4
Factors and indicators defining the capacity to supply seafood.

Indicators that were not quantified, and hence have no weighting, are shown *in italics*

#	Type	Factors	Indicator	Weight
1	Natural	Species biomass	Main commercial species biomass ('000 tonnes). Based on MSFD 3.2.1	4%
2	Natural	Secondary production	Other fish species biomass ('000 tonnes). Based on MSFD 1.2.1	4%
3	Natural	Primary production	Variation in PP (gC per m^2 per year)	4%
4	Natural	Habitat condition - pelagic	Change in plankton biomass and abundance. Based on MSFD 1.6.1	13%
5	Natural	Productive capacity of commercial stock	Percentage of stocks above levels of productive capacity. Based on JNCC Biodiversity Indicators B2	53%
6	Natural	Biomass stock quality	Percentage of large fish (Exceeding 50 cm). Based on MSFD 4.2.1	5%
7	Natural	Water quality	Inverse: Input of hazardous substances to the UK marine environment, as an index of estimated weight of substances per year. Based on JNCC Biodiversity Indicators B5b	6%
8	HDC	Fishing effort	Number of fishermen in the UK	12%
9	Natural	Habitat condition - benthic	*Inverse: Percentage of area with physical damage to predominant and special benthic habitats. Based on MSFD 1.6.1*	*
10	Natural	Nursery and spawning grounds	*Condition of nursery and spawning grounds for commercial fisheries*	
11	HDC	Accessibility	*Amount of fishing habitat that is accessible with fishing gear*	
12	HDC	Weather perception	*Decision to fish based on weather perception*	
13	HDC	Vessel technology	*Proportion of shipping vessels that have advanced navigational equipment*	

3.3. Composite index of supply

3.3.1. Capacity to supply wild caught seafood

The production of seafood was found to rely upon a number of factors that relate to natural capital; primary and secondary productivity (Eftec, 2015), pelagic and benthic habitat condition (Atkins et al., 2015; Eftec, 2015; Villamagna et al., 2014), species biomass (Atkins et al., 2015; Eftec, 2015), water quality (Eftec, 2015; Villamagna et al., 2014), condition of spawning or nursery grounds (Eftec, 2015; Hooper et al., 2017), and stock quality (Broszeit et al., 2017; Hattam et al., 2015; Villamagna et al., 2014). HDC that drive fishing pressure (i.e. the supply of seafood) include fishing effort (either vessel capacity or the number of fishers), vessel technology, access and the decision to fish as a result of the weather (Stephenson et al., 2018).

A further review of the literature informed the selection of indicators that relate to natural factors (Broszeit et al., 2017; Capuzzo et al., 2018; DEFRA, 2019a; Hattam et al., 2015; CEFAS, 2019; DEFRA, 2019b; DEFRA, 2019c; ICES, 2019; McQuatters-Gollop et al., 2018; OSPAR, 2017b; Radford et al., 2019). An inverse indicator was used to estimate the effect of water quality, i.e. the lower the input of hazardous substances into the marine environment, the higher the water quality. There was insufficient data available on the condition of spawning or nursery grounds, so no indicator for this factor was included. Likewise, there was insufficient data to populate a time series indicator for benthic habitat condition using the available EU Marine Strategy Framework Directive (MSFD) indicators, because only one measurement has been made for the period 2010–2015. Of the HDC indicators, only fishing effort could be estimated, based on the number of fishermen in the UK (Marine Management Organisation, 2019). The capacity to supply seafood was therefore measured by estimating eight indicators. The factors and indicators, along with their calculated weights, are summarised in Table 4.

The composite supply index for seafood was then calculated, and is shown with resource rent in Fig. 3.

3.3.2. Capacity to supply offshore wind energy

Wind speed is a critical requirement for supply of energy, but the

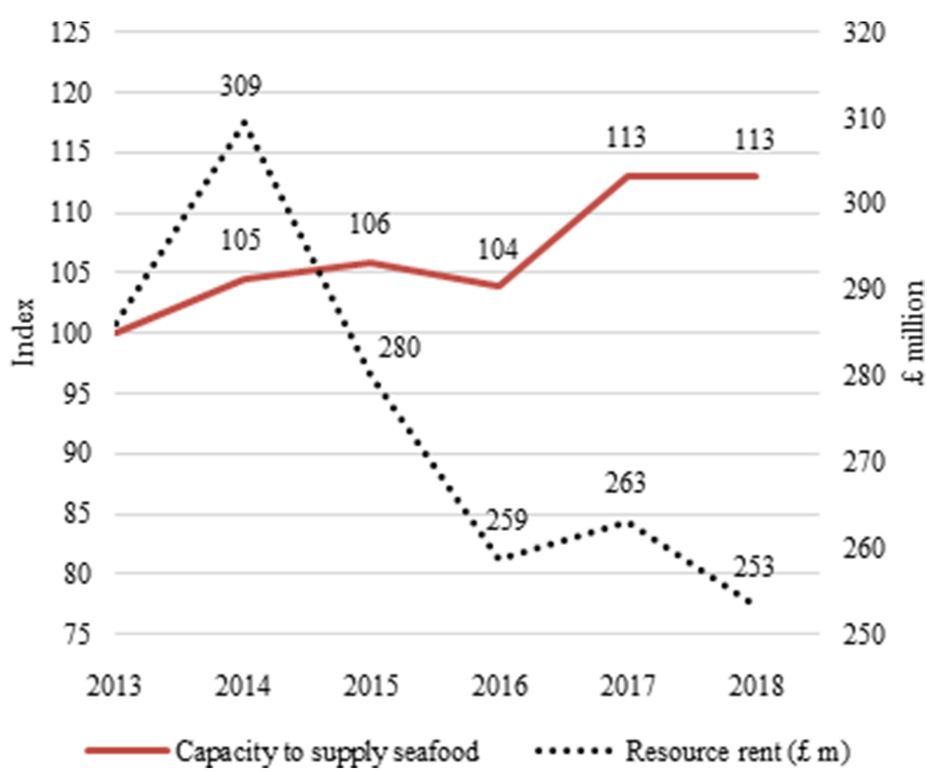

Fig. 3. Composite supply index and resource rent for seafood.

Table 5
Factors and indicators defining the capacity to supply offshore wind energy.

Indicators that were not quantified, and hence have no weighting, are shown *in italics*

#	Type	Factors	Indicator	Weight
1	HDC	Operational capacity	Capacity of installed offshore wind turbines	39%
2	HDC	Productivity	Mean capacity factor for operational turbines	24%
3	HDC	Availability and reliability	Functional availability of turbines	23%
4	Natural	Weather	Number of hours of suitable wind	14%
5	Natural	Wind quality and congestion	*Reduced wind due to high density of turbines*	

energy of the wind is not realised without turbine construction and submarine cable laying, i.e. HDC is crucial in realising the benefits of wind energy. Factors and indicators for the supply for offshore wind energy were determined from wind energy literature: operational capacity, capacity factor, availability, wind quality and wind speed (Faulstich et al., 2011; Feng et al., 2010; Miller and Kleidon, 2016; Sedaghat et al., 2017). Data to measure these indicators were sourced from the UK government reports (Department for Business Energy Industrial Strategy, 2019; The Crown Estate, 2019; The Crown Estate, 2018; The Crown Estate, 2016; The Crown Estate, 2015). An indicator for wind speed that was suitable for energy generation was calculated based on a cut in-speed of 2.5 m/s and a cut-out speed of 25 m/s (Sedaghat et al., 2017), using wind data for the Boulmer weather station on the North East coast of England (Met Office, 2019a). This station was chosen for its proximity to wind farms in the North Sea, and because the highly detailed wind records for the region were otherwise challenging to aggregate or average. The effect of reduced energy generation for turbines at high density was not estimated because calculations indicated that offshore wind farms were not close to the calculated threshold (Miller and Kleidon, 2016). The factors and indicators, and their calculated weightings, are summarised in Table 5.

The composite supply index for offshore wind energy was then calculated, and is shown with resource rent in Fig. 4.

3.3.3. Capacity to supply water sports

The factors relevant for the supply of wildlife watching activities were found to include accessibility, water quality, safety (including availability of rescue services), scenic quality, visitor congestion and environmental suitability (i.e. water depth, temperature, or wave height) (Bujosa et al., 2015; Paker and Vural, 2016; Paracchini et al., 2014; Peña-Alonso et al., 2017; Portman et al., 2016; Vallecillo et al., 2019; Villamagna et al., 2014). Data were not available to measure indicators for visitor congestion, environmental suitability to water sports or for scenic quality. The remaining 4 indicators were determined from the literature and measured using publicly available data (DEFRA, 2019a; European Environment Agency, 2014, 2015, 2016, 2017, 2018, 2019; Met Office, 2019b; RNLI, 2015, 2018b, 2018a, 2019). Although the weather was found to be an important consideration in nature-based

Table 6
Factors and indicators describing the capacity to supply water sports.

Indicators that were not quantified, and hence have no weighting, are shown *in italics*

#	Type	Factors	Indicator	Weight
1	HDC	Access	Number of bathing water sites, marinas and yacht clubs	13%
2	HDC	Safety and rescue	Number of RNLI launch and beach stations	22%
3	Natural	Bathing water quality	Coastal bathing water quality	24%
4	Natural	Water safety	Inverse: Hazardous substances in the marine environment	29%
5	Natural	Adverse weather	Inverse: Number of days of gales per year	12%
6	HDC	Visitor congestion	*Number of visitors per unit area of coastal sites*	
7	Natural	Environmental suitability	*Marine environment suitable for water sports*	
8	Natural	Scenic quality	*Tourism attracted by scenic quality of the area*	

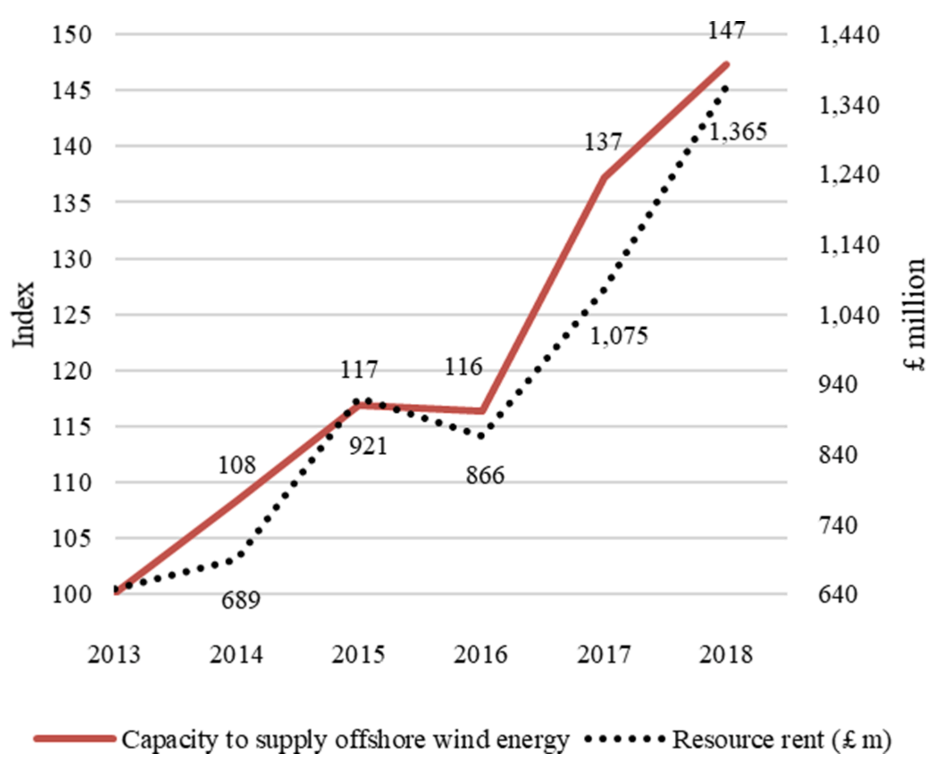

Fig. 4. Composite supply index and resource rent for offshore wind energy.

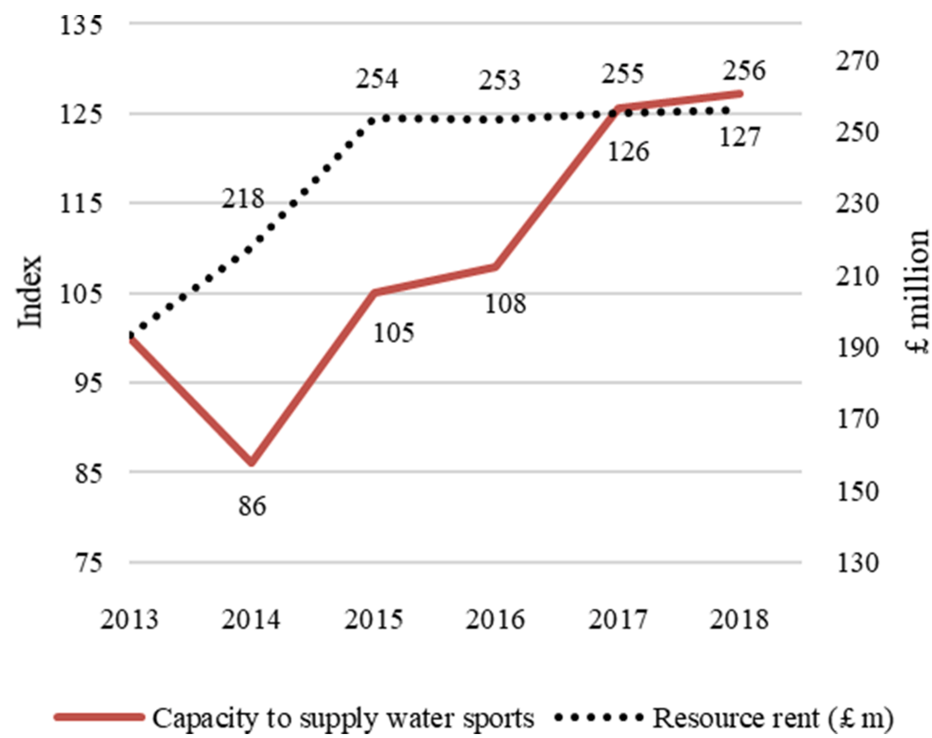

Fig. 5. Composite supply index and resource rent for water sports.

tourism (Verbos et al., 2018) and beach tourism (Peña-Alonso et al., 2017), no specific studies were found to exist on the impact of weather on water sports. The effect of good weather was therefore excluded as an indicator, but an indicator for the number of days of storms in the UK each year was included (Haigh et al., 2016; McCarthy et al., 2016; Met Office, 2019b; Muchan et al., 2015; Sibley et al., 2015), based on the assumption that high winds would prevent recreation by boat. The number of water sports clubs could also have been included, but no data was found to support this indicator. The factors and indicators, along with their weights, are summarised in Table 6.

The composite supply index for water sports was then calculated, and is shown with the resource rent in Fig. 5.

3.3.4. *Capacity to supply of wildlife watching benefits*

The factors relevant for the supply of wildlife watching activities were found to include site accessibility, safety, scenic quality, presence of birds and marine megafauna, visitor congestion and environmental suitability (in terms of water depth, temperature, or wave height) (Bentz et al., 2016; Marine Management Organisation, 2014, 2012; Paker and Vural, 2016; Paracchini et al., 2014; Peña-Alonso et al., 2017; Portman et al., 2016; Ryan et al., 2018; Villamagna et al., 2014). Indicators for wildlife watching were then determined based on the literature (Broszeit et al., 2017; DEFRA, 2019d; JNCC, 2020, 2019a, 2019b; Organisation, 2014; McCarthy et al., 2016; OSPAR, 2017a; Pinn et al., 2018; RAMSAR and JNCC, 2015). Although the weather was found to be an important consideration in nature-based tourism (Verbos et al., 2018), no specific studies were found to exist on the impact of weather perception on wildlife watching. Similarly as for water sports, the effect of good weather was not included, but an indicator of adverse weather was estimated, based on the assumption that high winds would prevent wildlife watching trips by boat.

Of these indicators, data was available to estimate the abundance of seabirds, days of adverse weather, marine mammal distribution, number of access points with rescue services, and number of cetacean encounters in the UK (DEFRA, 2019d; Hammond et al., 2018, 2013; Met Office, 2019b; Muchan et al., 2015; ORCA, 2018, 2016; OSPAR, 2017a; Pinn et al., 2018; RNLI, 2018b, 2018a, 2015, 2019; Russell et al., 2019; Sibley et al., 2015; Thompson et al., 2019). A significant data gap was that there are an unknown number of wildlife tour operators, and this indicator was excluded. No data was available for the effect of visitor congestion, accessibility, or scenic quality, and these were also excluded from further analysis. The factors and indicators, along with their calculated weights, are given in Table 7.

The composite supply index wildlife watching was then calculated, and is shown with the resource rent in Fig. 6.

Table 7
Factors and indicators describing the capacity to supply wildlife watching.

#	Type	Factors	Indicator	Weight
1	Natural	Abundance of birdlife	Abundance and distribution of seabirds	17%
2	Natural	Adverse weather	Inverse: Number of days of gales	45%
3	Natural	Abundance of sea life	Abundance and distribution of marine mammals	5%
4	Natural	Environmental suitability	Area of protected marine and coastal habitat	10%
5	HDC	Chance of seeing wildlife	Cetacean sightings over effort	11%
6	HDC	Safety and rescue	Number of RNLI launch and beach stations	11%
7	Natural	Scenic quality	*Tourism attracted by scenic quality of the area*	
8	HDC	Congestion	*Inverse: Congestion of boats or viewing areas*	
9	HDC	Accessibility	*Number of wildlife tour operators*	

Indicators that were not quantified, and hence have no weighting, are shown *in italics*.

3.4. *Sensitivity analysis*

A sensitivity analysis was carried out to investigate the effect of different weighting and aggregation methods on the composite supply index. The results for each case study compared with both compensatory aggregation and equal weighted aggregation are given in Fig. 7.

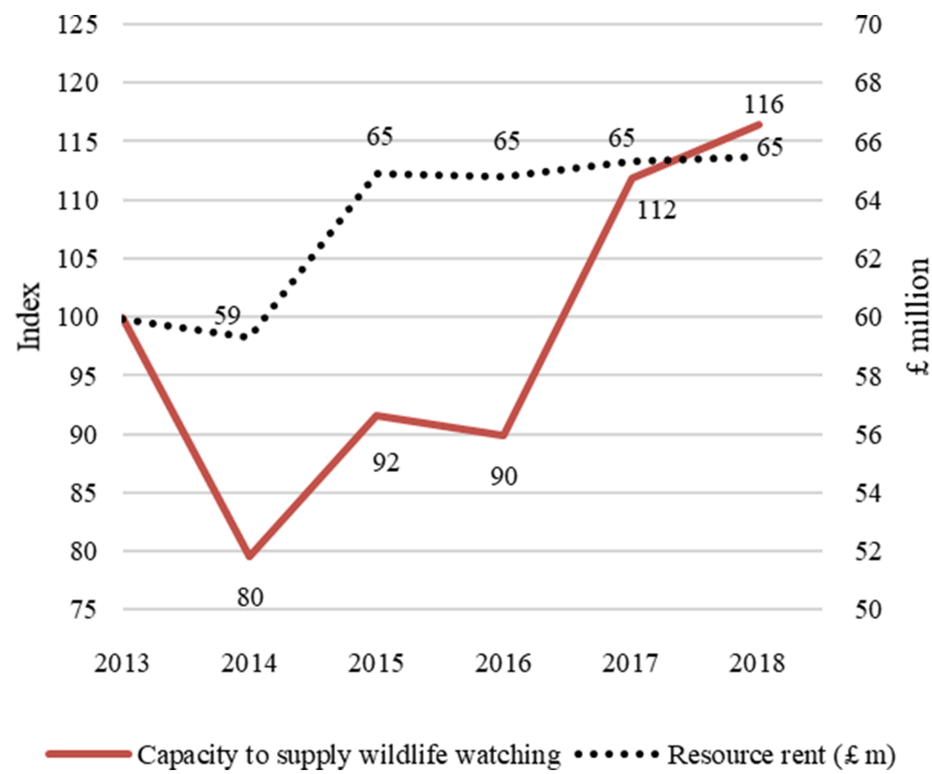

Fig. 6. Composite supply index and resource rent for wildlife watching.

Compensatory aggregation of seafood, offshore wind energy and water sports produced the same results as for non-compensatory aggregation (7a, 7b & 7c), because the natural elements of the composite indicator did not limit the HDC elements. For example, the indicators for natural capital on the seafood index (indicators 1 to 7 in Table 4) increased between 2014 and 2018, and so did not limit the index result. However, had the measures of these indicators decreased over this time period, the index would have been limited by this factor. The opposite was true for wildlife watching (7d); weather factors and the absence of wildlife were not compensated for by HDC elements of benefit supply (e.g. the number of access points). Aggregation using equal weighting gave very different results depending on the data inputs, which highlighted further the problems of an equal weighting approach. Further explanation is given in the Supplementary Material.

4. Discussion

4.1. Case study results

The supply all four of the benefits in this study increased between 2013 and 2018. As we anticipated, the results are highly sensitive to the index weightings and aggregation approaches. Although differing approaches to weighting and aggregation changed the rate of increased supply, the overall trend (i.e. positive) was the same for all aggregation approaches. The results for each benefit are now discussed in more detail.

Capacity to supply seafood increased, but the resource rent for seafood declined in real terms, despite an increase in the volume of seafood being caught. The resource rent for each kilogram of seafood has therefore declined from £0.42 in 2013 to £0.33 in 2018. These differ from the results reported by CEFAS in the Initial NCA for the marine and coastal environment (Thornton et al., 2019), in both trend and magnitude. The resource rent estimated in this study was higher than that of Thornton *et al.*, and our results show resource rent declining rather than increasing. There are two reasons for these differences; Firstly, the seafood sector was disaggregated from aquaculture and freshwater fisheries and found to have a higher operating surplus than the sector average (Office for National Statistics, 2019f), whereas previous approaches were based upon the fisheries and aquaculture sector as a whole. Secondly, additional 'ecosystem costs' specific to use of the resource had been previously included (Eftec, 2015; Thornton et al., 2019), but data for these could not be estimated in this application.

Capacity to supply wind energy increased due to the rapid expansion of offshore wind farm construction. The resource rent for offshore wind energy similarly increased, to £1,365 million in 2018. This result was lower than estimated by CEFAS (Thornton et al., 2019); our study used national accounts to estimate the contribution of offshore wind, as opposed to company level financial data used by CEFAS. Offshore wind energy has not been well defined in the ES literature with research efforts concentrated on the effects that construction of turbines have on other services. This is possibly because no biotic element is involved in delivery of wind energy. However, abiotic ES provide important benefits, and an approach to quantifying benefits from the marine area should not omit those flowing from abiotic resources. This case study could be further developed for other renewable energy technologies, and used to estimate the feedback effects of energy developments.

Participation in water sports increased from 37 million days in 2013 to 50 million days in 2018. The resource rent of this benefit also increased, along with the capacity for the benefit to be supplied. Combining the results for resource rent and non-monetary flows indicated that on average, water sports participation generated £8 of resource rent per person per visit, with little inter-annual variation between 2013 and 2018. Participation in wildlife watching similarly increased: from 60 million days in 2013 to 65 million days in 2018. Resource rent for wildlife watching increased over this time but resource rent for wildlife watching declined from £12 per person per visit in 2013 to £9 in 2018. The non-monetary flow and resource rent for these benefits had not been estimated before, and highlights the importance of both natural capital and HDC to the realisation of recreation activities. However, the composite supply index for wildlife watching omits an

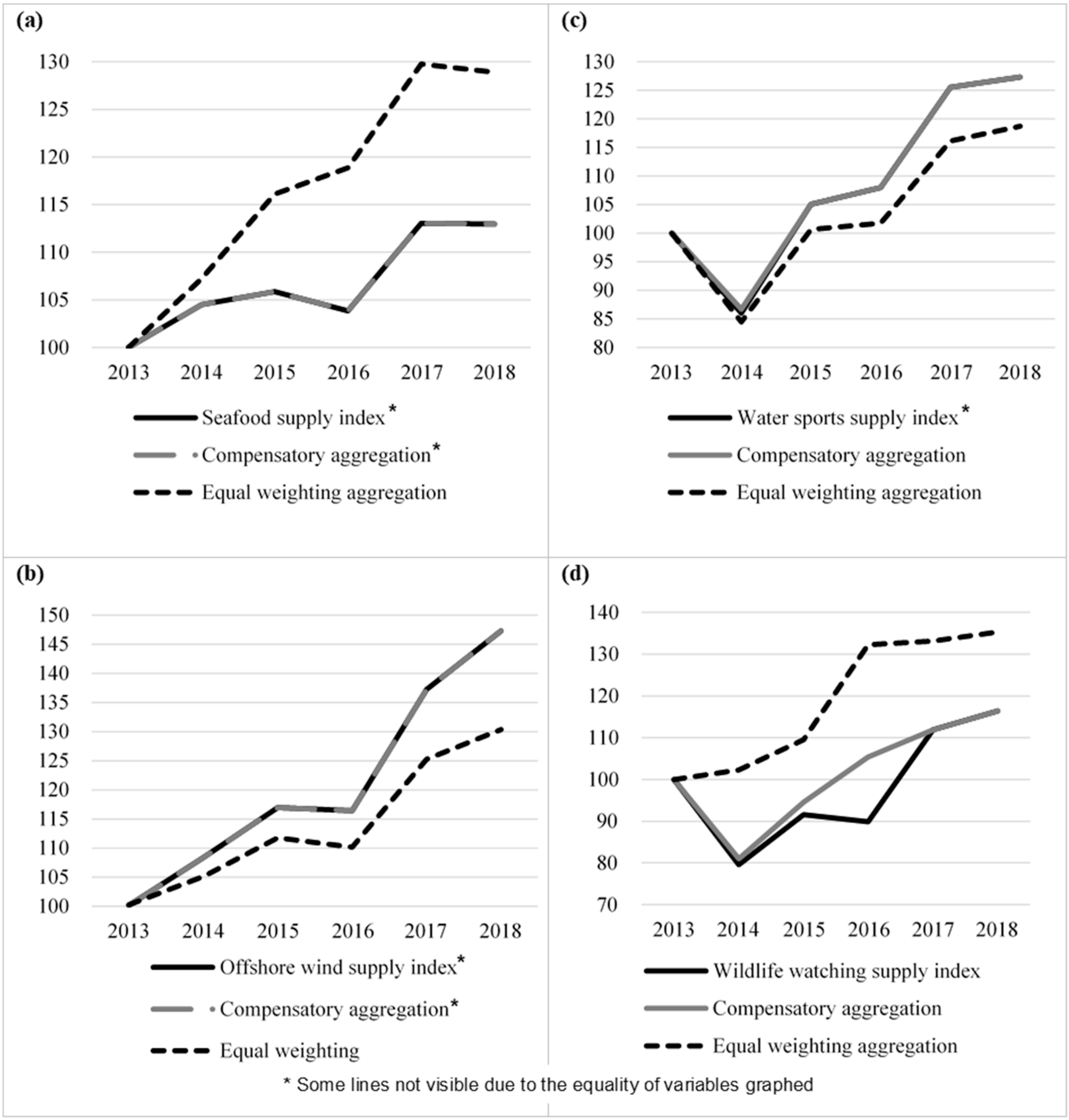

Fig. 7. Sensitivity analysis for (a) seafood, (b) offshore wind energy, (c) water sports, and (d) wildlife watching.

indicator for the number of wildlife tour operators, which is likely to be important in describing the supply of benefits, and its omission likely made the result less robust.

It was clear from examining even only four marine benefits that the way they are measured and supplied to people varied greatly. For example, seafood, a biotic resource that depends on food webs to replenish stock, required a number of indicators to represent the condition of the natural system. Other biotic resources that are extracted for use, such as kelp, may be represented using similar indicators. In contrast, recreational benefits required supporting activities (e.g. rescue and lifeguarding) in order for benefits to be supplied. Notably, the composite index demonstrated here can describe supply to all (market and non-market) users, whereas resource rent can only estimate market activity (as it is calculated from gross operating surplus).

4.2. Limitations and improvements

The statistical analysis used to calculate indicator weights was based on applications to sustainability indices rather than for benefit supply, and so was intended to weight indicators according to their variance from one another rather than for their importance to the benefit being supplied. Although PCA is a well-established method, multivariate regression would instead enable a causal relationship between actual flow of benefits and the supply factors. This analysis would therefore have been significantly improved by using a multivariate regression approach to weight the importance of the indicators to benefit supply (i.e. identify the variables most appropriate to describe the flow of benefits). However, data availability already limited the estimates within this research, and regression analysis would require many more years of data in order to be robust. Though used less frequently, participatory approaches could also have been used to estimate or validate the statistical weightings. For example, expert panel has been used to determine weights for a drinking water quality index (Scheili et al., 2020). A further limitation of the approach was that although the supply of benefits varies seasonally and spatially, adjustments for this was not included in the analysis. In addition, data scarcity limited the resulting indices. For example, of the HDC factors describing seafood, only the number of fishermen could be quantified. Furthermore, if the maximum and minimum values were known for each indicator, then the composite index could have been normalised with respect to ecological limits rather than to a baseline. Finally, the assumption that benefits contributed as major inputs to production of certain economic sectors may have understated or overstated the resource rent for that benefit, because other 'minor' sectors were ignored. For example, algae could be used in terrestrial agriculture as well as in marine aquaculture. The contribution of these benefits to other (intermediary) sectors was also ignored and understated. For example, seafood products are also used in animal feed, but this effect was not estimated.

Other general improvements to the approach would be to further include measures of natural capital asset quality within the indices, because the supply of a benefit is dependent not just on quantity but quality. For example, the carrying capacity of recreation areas could be quantified and its effect on cultural services could be estimated (Gonson et al., 2018; Peña-Alonso et al., 2017; Tian et al., 2018). Some ES could also have been disaggregated further; recreation activities were split from one another, but the contribution of geodiversity to specific abiotic ES and benefits would be an important extension (Gordon and Barron, 2013; Gray, 2011; Gray et al., 2013). Finally, the supply of ES and benefits from specific habitats could be measured, similar to the approach in Scotland (Scottish Natural Heritage, 2019). This would be important because different habitats contribute to different services. However, subdividing to habitat at the national scale is too resource intensive and currently lacks data of sufficient resolution, so would be more appropriate at a regional level.

5. Conclusion

This research systematically linked natural capital, human-derived capital, ecosystem services benefits and the economy in an 'end-to-end' supply chain; although a need for such an approach is well documented, there have been few attempts to do so. Aspects of capital were used to estimate the supply of benefits in a way that has not been previously attempted; an approach of this sort could improve upon existing approaches within natural capital accounting (NCA) (Thornton et al., 2019, p. 55). Our approach explored the factors and indicators required for the supply of benefits, rather than focussing on habitat and asset extent, and could be scaled to national or local levels. Benefits were also identified as inputs to specific sectors in the marine economy, which has previously been carried out for the fishing sector, but not for recreation. Case studies were specifically drawn from the marine environment, as although examples of natural capital indicators exist for terrestrial and coastal habitats, no similar approach has been made for the marine area. This approach went beyond the current thinking of homogenous cultural services, and particularly demonstrated how the capacity to supply recreation benefits could be measured in a way that is compatible with NCA. It also emphasised the sensitivity of composite indices to indicator weightings. This analysis therefore highlighted the knowledge gaps that would need to be filled when developing indicator sets, as the UK government intends (DEFRA, 2019e). Use of this method within a wider decision tool or indicator set (e.g. NCAI for Scotland (McKenna et al., 2019; Tillin et al., 2019)) would therefore facilitate broader links with NCA. The approach may also be of interest to practitioners of ocean accounting, particularly since this approach is still relatively effective with limited data, and where data scarcity may otherwise limit the establishment of environmental accounts under the SEAA or EEA frameworks. The methodology could be extended to other benefits and to estimate the feedback effects of one industry upon another. To conclude, despite the suggested improvements, we feel the work presented here contributes to current thinking on the natural capital approach. We encourage the development of similar and related methods so that decision making tools, particularly in the marine area, can be continually improved upon.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoser.2021.101308.

References

Alam, M., Dupras, J., Messier, C., 2016. A framework towards a composite indicator for urban ecosystem services. Ecol. Indic. 60, 38–44. https://doi.org/10.1016/j.ecolind.2015.05.035.

Allan, G., Comerford, D., Connolly, K., McGregor, P., 2019. Incorporating Natural Capital into A Computable General Equilibrium Model for Scotland, in: 27th International Input-Output Association Conference. Glasgow, UK.

Arkenford, 2018. Watersports Participation Survey 2018.

Arkenford, 2017. Watersport Participation Survey: Summary Report 2017.

Ashley, M., Rees, S.E., Cameron, A., 2018. North Devon Marine Pioneer Part 1 : State of the art report of the links between the ecosystem and ecosystem services in the North Devon Marine Pioneer 97.

Atkins, J.P., Burdon, D., Elliott, M., 2015. Identification of a Practicable Set of Ecosystem Indicators for Coastal and Marine Ecosystem Services, in: Turner, R.K., Schaafsma, M. (Eds.), Coastal Zones Ecosystem Services: From Science to Values and Decision Making. Springer, Switzerland, pp. 79–102. 10.1007/978-3-319-17214-9_5.

Australian Bureau of Statistics, 2017. Information Paper: An Experimental Ecosystem Account for the Great Barrier Reef Region, 2015. Canberra, Australia.

Avila-Foucat, V.S., Gendron, D., Revollo-Fernandez, D., Popoca, E.I., Ramírez, A., 2017. Determinants of the potential demand for whale watching in Loreto Bay National Park. Mar. Policy 81, 37–44. https://doi.org/10.1016/j.marpol.2017.03.006.

Barbier, E.B., 2007. Valuing ecosystem services as productive inputs. Econ. Policy 22, 179–229. https://doi.org/10.1111/j.1468-0327.2007.00174.x.

Barton, D.N., Certain, G., Chaves, S., Cuadra Álvaro Herrera, J., Kvaløy, P., Auxiliadora Mora, M., Nybø Vilma Obando, S., Pedersen, B., Ugalde, J., Vargas, M., 2014. Nature Index Costa Rica: An IPBES pilot project.

Bentz, J., Lopes, F., Calado, H., Dearden, P., 2016. Managing marine wildlife tourism activities: Analysis of motivations and specialization levels of divers and whale watchers. Tour. Manage. Perspect. 18, 74–83. https://doi.org/10.1016/j.tmp.2016.01.004.

Blanc, I., Friot, D., Margni, M., Jolliet, O., 2008. Towards a new index for environmental sustainability based on a DALY weighting approach. Sustain. Dev. 16, 251–260. https://doi.org/10.1002/sd.

Börger, T., Beaumont, N.J., Pendleton, L., Boyle, K.J., Cooper, P., Fletcher, S., Haab, T., Hanemann, M., Hooper, T., Hussain, S.S., Portela, R., Stithou, M., Stockill, J., Taylor, T., Austen, M.C., 2014. Incorporating ecosystem services in marine planning: The role of valuation. Mar. Policy 46, 161–170. https://doi.org/10.1016/j.marpol.2014.01.019.

Broszeit, S., Beaumont, N.J., Uyarra, M.C., Heiskanen, A.S., Frost, M., Somerfield, P.J., Rossberg, A.G., Teixeira, H., Austen, M.C., 2017. What can indicators of good environmental status tell us about ecosystem services?: Reducing efforts and increasing cost-effectiveness by reapplying biodiversity indicator data. Ecol. Indic. 81, 409–442. https://doi.org/10.1016/j.ecolind.2017.05.057.

Bujosa, A., Riera, A., Hicks, R.L., McConnell, K.E., 2015. Densities rather than shares: improving the measurement of congestion in recreation demand models. Environ. Resour. Econ. 61, 127–140. https://doi.org/10.1007/s10640-014-9785-9.

Capuzzo, E., Lynam, C.P., Barry, J., Stephens, D., Forster, R.M., Greenwood, N., McQuatters-Gollop, A., Silva, T., van Leeuwen, S.M., Engelhard, G.H., 2018. A decline in primary production in the North Sea over 25 years, associated with reductions in zooplankton abundance and fish stock recruitment. Glob. Chang. Biol. 24, e352–e364. https://doi.org/10.1111/gcb.13916.

CEFAS, 2019. Assessment of progress towards the achievement of Good Environmental Status for benthic biodiversity and seafloor habitats [WWW Document]. URL https://moat.cefas.co.uk/biodiversity-food-webs-and-marine-protected-areas/benthic-habitats/ (accessed 11.14.19).
Cordier, M., Pérez Agúndez, J.A., Hecq, W., Hamaide, B., 2014. A guiding framework for ecosystem services monetization in ecological-economic modeling. Ecosyst. Serv. 8, 86–96. https://doi.org/10.1016/j.ecoser.2014.03.003.
Costanza, R., Daly, H.E., 1992. Natural Capital and Sustainable Development. Conserv. Biol. 10.1046/j.1523-1739.1992.610037.x.
Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S.J., Kubiszewski, I., Farber, S.C., Turner, R.K., 2014. Changes in the global value of ecosystem services. Glob. Environ. Change. 26, 152–158. https://doi.org/10.1016/j.gloenvcha.2014.04.002.
Culhane, F., Frid, C., Royo Gelabert, E., White, L., Robinson, L., 2018a. Linkage matrix of marine ecosystem services and ecosystem components, Version 1.0. 10.17638/datacat.liverpool.ac.uk/496.
Culhane, F., Frid, C.L.J., Royo Gelabert, E., White, L., Robinson, L.A., 2018b. Linking marine ecosystems with the services they supply: what are the relevant service providing units? Ecol. Appl. 28, 1740–1751. https://doi.org/10.1002/eap.1779.
Curnow, J., 2019. Environmental economics in UK environmental policy: Defra's 25 Year Environment Plan. J. Environ. Econ. Policy 8, 353–358. https://doi.org/10.1080/21606544.2019.1601135.
Deane, R., Walker, A., 2018. Towards a Register of Exmoor's Natural Capital.
DEFRA, 2019a. UK Biodiversity Indicators 2019, B5b Marine pollution.
DEFRA, 2019b. UK Biodiversity Indicators 2014: B2 Sustainable Fisheries.
DEFRA, 2019c. UK Biodiversity Indicators 2014: D1a Fish Size Classes.
DEFRA, 2019d. Wild Bird Populations in the UK: 1970 to 2018.
DEFRA, 2019e. Measuring environmental change: outcome indicator framework for the 25 Year Environment Plan.
den Butter, F.A.G., Hofkes, M.W., 1995. Sustainable development with extractive and non-extractive use of the environment in production. Environ. Resour. Econ. 6, 341–358. https://doi.org/10.1007/BF00691818.
Department for Business Energy & Industrial Strategy, 2019. Digest of United Kingdom Energy Statistics (DUKES) 2019.
Dickie, I., Neupauer, S., 2019. Natural capital accounts: nations and organizations. J. Environ. Econ. Policy 8, 379–393. https://doi.org/10.1080/21606544.2019.1639219.
Dobbie, M.J., Dail, D., 2013. Robustness and sensitivity of weighting and aggregation in constructing composite indices. Ecol. Indic. 29, 270–277. https://doi.org/10.1016/j.ecolind.2012.12.025.
Dunne, S., 2019. Northern Ireland Annual Tourism Statistics 2018.
Eftec, 2015. Developing UK Natural Capital Accounts: Marine.
Elliott, L.R., White, M.P., Grellier, J., Rees, S.E., Waters, R.D., Fleming, L.E., 2018. Recreational visits to marine and coastal environments in England: Where, what, who, why, and when? Mar. Policy 97, 305–314. https://doi.org/10.1016/j.marpol.2018.03.013.
European Commission, 2016. European Social Progress Index 2016.
European Environment Agency, 2019. European bathing water quality in 2018.
European Environment Agency, 2018. European bathing water quality in 2017.
European Environment Agency, 2017. European bathing water quality in 2016.
European Environment Agency, 2016. European bathing water quality in 2015. 10.2800/599721.
European Environment Agency, 2015. European bathing water quality in 2014.
European Environment Agency, 2014. European bathing water quality in 2013.
Faulstich, S., Hahn, B., Tavner, P.J., 2011. Wind turbine downtime and its importance for offshore deployment. Wind Energy 14, 327–337. https://doi.org/10.1002/we.421.
Feng, Y., Tavner, P.J., Long, H., 2010. Early experiences with UK round 1 offshore wind farms. Proc. Inst. Civ. Eng. - Energy 163, 167–181. https://doi.org/10.1680/ener.2010.163.4.167.
Gan, X., Fernandez, I.C., Guo, J., Wilson, M., Zhao, Y., Zhou, B., Wu, J., 2017. When to use what: Methods for weighting and aggregating sustainability indicators. Ecol. Indic. 81, 491–502. https://doi.org/10.1016/j.ecolind.2017.05.068.
Gómez-Limón, J.A., Riesgo, L., 2009. Alternative approaches to the construction of a composite indicator of agricultural sustainability: an application to irrigated agriculture in the Duero basin in Spain. J. Environ. Manage. 90, 3345–3362. https://doi.org/10.1016/j.jenvman.2009.05.023.
Gonson, C., Pelletier, D., Alban, F., 2018. Social carrying capacity assessment from questionnaire and counts survey: insights for recreational settings management in coastal areas. Mar. Policy 98, 146–157. https://doi.org/10.1016/j.marpol.2018.08.016.
Gordon, J.E., Barron, H.F., 2013. The role of geodiversity in delivering ecosystem services and benefits in Scotland. Scottish J. Geol. 49, 41–58. https://doi.org/10.1144/sjg2011-465.
Gray, M., 2011. Other nature: geodiversity and geosystem services. Environ. Conserv. 38, 271–274. 10.1017/S0376892911000117.
Gray, M., Gordon, J.E., Brown, E.J., 2013. Geodiversity and the ecosystem approach: the contribution of geoscience in delivering integrated environmental management. Proc. Geol. Assoc. 124, 659–673. https://doi.org/10.1016/j.pgeola.2013.01.003.
Guerry, A.D., Ruckelshaus, M.H., Arkema, K.K., Bernhardt, J.R., Guannel, G., Kim, C.-K., Marsik, M., Papenfus, M., Toft, J.E., Verutes, G.M., Wood, S.A., Beck, M.W., Chan, F., Chan, K.M. a., Gelfenbaum, G., Gold, B.D., Halpern, B.S., Labiosa, W.B., Lester, S.E., Levin, P.S., McField, M., Pinsky, M.L., Plummer, M., Polasky, S., Ruggiero, P., Sutherland, D. a., Tallis, H., Day, A., Spencer, J., 2012. Modeling benefits from nature: using ecosystem services to inform coastal and marine spatial planning. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 8, 107–121. 10.1080/21513732.2011.647835.
Gutiérrez-Espeleta, E.E., 2017. Costa Rica: influence of natural resources accounting on public policy and decision-making. In: Vardon, M., Bass, S., Ahlroth, S., Ruijs, A. (Eds.), Forum on Natural Capital Accounting for Better Policy Decisions: Taking Stock and Moving Forward. World Bank, Wealth Accounting and the Valuation of Ecosystem Services, Washington D.C.
Haigh, I.D., Wadey, M.P., Wahl, T., Ozsoy, O., Nicholls, R.J., Brown, J.M., Horsburgh, K., Gouldby, B., 2016. Spatial and temporal analysis of extreme sea level and storm surge events around the coastline of the UK. Sci. Data 3, 160107. https://doi.org/10.1038/sdata.2016.107.
Haines-Young, R., Potschin-Young, M., 2018. Revision of the common international classification for ecosystem services (CICES V5.1): A policy brief. One Ecosyst. 3, e27108 https://doi.org/10.3897/oneeco.3.e27108.
Haines-Young, R., Potschin, M., 2010. Proposal for a Common International Classification of Ecosystem Goods and Services (CICES) for Integrated Environmental and Economic Accounting.
Halpern, B.S., Longo, C., Hardy, D., McLeod, K.L., Samhouri, J.F., Katona, S.K., Kleisner, K., Lester, S.E., Oleary, J., Ranelletti, M., Rosenberg, A.A., Scarborough, C., Selig, E.R., Best, B.D., Brumbaugh, D.R., Chapin, F.S., Crowder, L.B., Daly, K.L., Doney, S.C., Elfes, C., Fogarty, M.J., Gaines, S.D., Jacobsen, K.I., Karrer, L.B., Leslie, H.M., Neeley, E., Pauly, D., Polasky, S., Ris, B., St Martin, K., Stone, G.S., Rashid Sumaila, U., Zeller, D., 2012. An index to assess the health and benefits of the global ocean. Nature 488, 615–620. https://doi.org/10.1038/nature11397.
Hammond, P.S., Lacey, C., Gilles, A., Viquerat, S., Börjesson, P., Herr, H., Macleod, K., Ridoux, V., Santos, M.B., Scheidat, M., Teilmann, J., Vingada, J., Øien, N., 2018. Estimates of cetacean abundance in European Atlantic waters in summer 2016 from the SCANS-III aerial and shipboard surveys. 10.1016/j.ijppaw.2016.02.001.
Hammond, P.S., Macleod, K., Berggren, P., Borchers, D.L., Burt, L., Cañadas, A., Desportes, G., Donovan, G.P., Gilles, A., Gillespie, D., Gordon, J., Hiby, L., Kuklik, I., Leaper, R., Lehnert, K., Leopold, M., Lovell, P., Øien, N., Paxton, C.G.M., Ridoux, V., Rogan, E., Samarra, F., Scheidat, M., Sequeira, M., Siebert, U., Skov, H., Swift, R., Tasker, M.L., Teilmann, J., Van Canneyt, O., Vázquez, J.A., 2013. Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. Biol. Conserv. 164, 107–122. https://doi.org/10.1016/j.biocon.2013.04.010.
Hattam, C., Atkins, J.P., Beaumont, N., Börger, T., Böhnke-Henrichs, A., Burdon, D., De Groot, R., Hoefnagel, E., Nunes, P.A.L.D., Piwowarczyk, J., Sastre, S., Austen, M.C., 2015. Marine ecosystem services: Linking indicators to their classification. Ecol. Indic. 49, 61–75. https://doi.org/10.1016/j.ecolind.2014.09.026.
Hicks, C.C., 2011. How Do we value our reefs? Risks and tradeoffs across scales in "biomass-based" economies. Coast. Manage. 39, 358–376. https://doi.org/10.1080/08920753.2011.589219.
HM Government, 2018. A Green Future: Our 25 Year Plan to Improve the Environment.
Hooper, T., Ashley, M., Börger, T., Langmead, O., Marcone, O., Rees, S.E., Rendon, O., Beaumont, N.J., Attrill, M.J., Austen, M.C., 2018. Application of the natural capital approach to the marine environment to aid decision - making.
Hooper, T., Beaumont, N.J., Griffiths, C., Langmead, O., Somerfield, P.J., 2017. Assessing the sensitivity of ecosystem services to changing pressures. Ecosyst. Serv. 24, 160–169. https://doi.org/10.1016/j.ecoser.2017.02.016.
Hooper, T., Börger, T., Langmead, O., Marcone, O., Rees, S.E., Rendon, O., Beaumont, N., Attrill, M.J., Austen, M., 2019. Applying the natural capital approach to decision making for the marine environment. Ecosyst. Serv. 38, 100947 https://doi.org/10.1016/j.ecoser.2019.100947.
ICES, CIEM, 2019. Stock Assessment Graphs and Tables [WWW Document]. URL http://standardgraphs.ices.dk/stockList.aspx (accessed 9.20.19).
JNCC, 2020. Special Areas of Conservation: Harbour porpoise (Phocoena phocoena) [WWW Document]. URL https://sac.jncc.gov.uk/species/S1351/ (accessed 2.26.20).
JNCC, 2019a. Natura 2000 summary site details.
JNCC, 2019b. UK Offshore Marine Protected Areas.
Jobstvogt, N., Watson, V., Kenter, J.O., 2014. Looking below the surface: The cultural ecosystem service values of UK marine protected areas (MPAs). Ecosyst. Serv. 10, 97–110. https://doi.org/10.1016/j.ecoser.2014.09.006.
Jones, L., Norton, L., Austin, Z., Browne, A.L., Donovan, D., Emmett, B.A., Grabowski, Z. J., Howard, D.C., Jones, J.P.G., Kenter, J.O., Manley, W., Morris, C., Robinson, D.A., Short, C., Siriwardena, G.M., Stevens, C.J., Storkey, J., Waters, R.D., Willis, G.F., 2016. Stocks and flows of natural and human-derived capital in ecosystem services. Land use Policy 52, 151–162. https://doi.org/10.1016/j.landusepol.2015.12.014.
Kandziora, M., Burkhard, B., Müller, F., 2013. Interactions of ecosystem properties, ecosystem integrity and ecosystem service indicators: a theoretical matrix exercise. Ecol. Indic. 28, 54–78. https://doi.org/10.1016/j.ecolind.2012.09.006.
Klinger, D.H., Maria Eikeset, A., Davíðsdóttir, B., Winter, A.-M., Watson, J.R., 2018. The mechanics of blue growth: management of oceanic natural resource use with multiple, interacting sectors. Mar. Policy 87, 356–362. https://doi.org/10.1016/j.marpol.2017.09.025.
Kotzee, I., Reyers, B., 2016. Piloting a social-ecological index for measuring flood resilience: a composite index approach. Ecol. Indic. 60, 45–53. https://doi.org/10.1016/j.ecolind.2015.06.018.
Lange, G.-M., Wodon, Q., Carey, K., 2018. The Changing Wealth of Nations 2018: Building a Sustainable Future, The Changing Wealth of Nations 2018: Building a Sustainable Future. 10.1596/978-1-4648-1046-6.
Liquete, C., Piroddi, C., Drakou, E.G., Gurney, L., Katsanevakis, S., Charef, A., Egoh, B., 2013. Current status and future prospects for the assessment of marine and coastal ecosystem services: a systematic review. PLoS One 8. https://doi.org/10.1371/journal.pone.0067737.
Luisetti, T., Turner, R.K., Jickells, T., Andrews, J., Elliott, M., Schaafsma, M., Beaumont, N.J., Malcolm, S., Burdon, D., Adams, C., Watts, W., 2014. Coastal zone

ecosystem services: From science to values and decision making; a case study. Sci. Total Environ. 493, 682–693. https://doi.org/10.1016/j.scitotenv.2014.05.099.
Luzzati, T., Gucciardi, G., 2015. A non-simplistic approach to composite indicators and rankings: an illustration by comparing the sustainability of the EU Countries. Ecol. Econ. 113, 25–38. https://doi.org/10.1016/j.ecolecon.2015.02.018.
Mace, G.M., 2019. The ecology of natural capital accounting. Oxford Rev. Econ. Policy 35, 54–67. https://doi.org/10.1093/oxrep/gry023.
Maes, J., Liquete, C., Teller, A., Erhard, M., Paracchini, M.L., Barredo, J.I., Grizzetti, B., Cardoso, A., Somma, F., Petersen, J.-E., Meiner, A., Gelabert, E.R., Zal, N., Kristensen, P., Bastrup-Birk, A., Biala, K., Piroddi, C., Egoh, B., Degeorges, P., Fiorina, C., Santos-Martín, F., Naruševičius, V., Verboven, J., Pereira, H.M., Bengtsson, J., Gocheva, K., Marta-Pedroso, C., Snäll, T., Estreguil, C., San-Miguel-Ayanz, J., Pérez-Soba, M., Grêt-Regamey, A., Lillebø, A.I., Malak, D.A., Condé, S., Moen, J., Czúcz, B., Drakou, E.G., Zulian, G., Lavalle, C., 2016. An indicator framework for assessing ecosystem services in support of the EU Biodiversity Strategy to 2020. Ecosyst. Serv. 17, 14–23. https://doi.org/10.1016/j.ecoser.2015.10.023.
Mancini, M.S., Galli, A., Niccolucci, V., Lin, D., Hanscom, L., Wackernagel, M., Bastianoni, S., Marchettini, N., 2017. Stocks and flows of natural capital: Implications for Ecological Footprint. Ecol. Indic. 77, 123–128. https://doi.org/10.1016/j.ecolind.2017.01.033.
Marine Management Organisation, 2019. Uk Sea Fisheries Statistics 2018.
Marine Management Organisation, 2014. Modelling marine recreation potential in England (MMO 1064).
Marine Management Organisation, 2012. Compilation of spatial data on marine recreation activities (MMO 1013).
McCarthy, M., Spillane, S., Walsh, S., Kendon, M., 2016. The meteorology of the exceptional winter of 2015/2016 across the UK and Ireland. Weather 71, 305–313. https://doi.org/10.1002/wea.2823.
McKenna, T., Blaney, R., Brooker, R.W., Ewing, D.A., Pakeman, R.J., Watkinson, P., O'Brien, D., 2019. Scotland's natural capital asset index: Tracking nature's contribution to national wellbeing. Ecol. Indic. 107, 105645 https://doi.org/10.1016/j.ecolind.2019.105645.
McQuatters-Gollop, A., Atkinson, A., Bedford, J., Best, M., Bresnan, E., Cook, K., Devlin, M., Johns, D.G., Machairopoulou, M., McKinney, A., Mellor, A., Ostle, C., Tett, P., Aubert, A., 2018. Changes in plankton biomass and abundance [WWW Document]. CEFAS UK Mar. Online Assess. Tool. URL https://moat.cefas.co.uk/biodiversity-food-webs-and-marine-protected-areas/pelagic-habitats/plankton-biomass/ (accessed 11.14.19).
Met Office, 2019a. Met Office Integrated Data Archive System (MIDAS) Land and Marine Surface Stations Data (1853-current) [WWW Document]. NCAS Br. Atmos, Data Cent http://catalogue.ceda.ac.uk/uuid/220a65615218d5c9cc9e4785a3234bd0 (accessed 1.14.20).
Met Office, 2019b. UK Storm Centre [WWW Document]. URL https://www.metoffice.gov.uk/weather/warnings-and-advice/uk-storm-centre/index (accessed 11.14.19).
Millenium Ecosystem Assessment, 2005. Ecosystems and Human Wellbeing: Synthesis. Island Press, Washington D.C. 10.1196/annals.1439.003.
Miller, L.M., Kleidon, A., 2016. Wind speed reductions by large-scale wind turbine deployments lower turbine efficiencies and set low generation limits. Proc. Natl. Acad. Sci. 113, 13570–13575. https://doi.org/10.1073/pnas.1602253113.
Mills, B., Cummins, A., 2013. The economic impact of domestic surfing on the United Kingdom. Surfers Against Sew. 32.
Mortimer, D., Fleming, V., Smith, M., Stoker, B., Weighell, T., 2017. Natural capital work in JNCC.
Muchan, K., Lewis, M., Hannaford, J., Parry, S., 2015. The winter storms of 2013/2014 in the UK: hydrological responses and impacts. Weather 70, 55–61. https://doi.org/10.1002/wea.2469.
Nardo, M., Saisana, M., Saltelli, A., Tarantola, S., 2005. Tools for Composite Indicators Building.
Natural Capital Committee, 2019. Marine and the 25 year environment plan.
Natural Capital Committee, 2017. How to do it: a natural capital workbook. Version 1 31.
Natural Capital Committee, 2014. Towards a Framework for Defining and Measuring Changes in Natural Capital.
Niemeijer, D., de Groot, R.S., 2008. A conceptual framework for selecting environmental indicator sets. Ecol. Indic. 8, 14–25. https://doi.org/10.1016/j.ecolind.2006.11.012.
Ochuodho, T.O., Alavalapati, J.R.R., 2016. Integrating natural capital into system of national accounts for policy analysis: An application of a computable general equilibrium model. For. Policy Econ. 72, 99–105. https://doi.org/10.1016/j.forpol.2016.06.020.
OECD, 2020. OECD Better-Life Index.
OECD, 2008. Handbook Composite Indicators. Methodology and User Guide.
Office for National Statistics, 2019a. United Kingdom Input-Output Analytical Tables 2015.
Office for National Statistics, 2019b. Index of Service Sectors Time Series (L2NC) [WWW Document]. URL https://www.ons.gov.uk/economy/grossdomesticproductgdp/timeseries/l2nc/pn2.
Office for National Statistics, 2019c. Index of Production Time Series (DIOP) [WWW Document]. URL https://www.ons.gov.uk/economy/economicoutputandproductivity/output/datasets/indexofproduction (accessed 10.10.19).
Office for National Statistics, 2019d. Agriculture, growth annual, time series A2EQ [WWW Document]. URL https://www.ons.gov.uk/economy/grossdomesticproductgdp/timeseries/a2eq/pn2 (accessed 10.11.19).
Office for National Statistics, 2019e. Gross operating surplus & mixed income deflator [WWW Document]. URL https://www.ons.gov.uk/economy/grossdomesticproductgdp/timeseries/ybge/bb (accessed 1.31.20).
Office for National Statistics, 2019f. Annual Business Survey 2018.
Office for National Statistics, 2019g. UK natural capital: peatlands.
Office for National Statistics, 2019h. Estimates of the population for the UK. England and Wales, Scotland and Northern Ireland.
Office for National Statistics, 2018a. UK Environmental Accounts 2018.
Office for National Statistics, 2018b. UK Natural Capital: interim review and revised 2020 roadmap.
Office for National Statistics, 2018c. UK natural capital : Ecosystem service accounts, 1997 to 2015.
Office for National Statistics, 2018d. United Kingdom Input-Output Analytical Tables 2014 [WWW Document].
Office for National Statistics, 2017a. United Kingdom Input-Output Analytical Tables 2013 [WWW Document]. URL https://www.ons.gov.uk/economy/nationalaccounts/supplyandusetables/datasets/ukinputoutputanalyticaltablesdetailed (accessed 12.20.17).
Office for National Statistics, 2017b. Annual Business Survey 2015 Revised Results.
Office for National Statistics, 2016a. Methodology: Natural Capital.
Office for National Statistics, 2016b. Scoping UK coastal margin ecosystem accounts.
Office for National Statistics, 2015. Natural capital accounting 2020 roadmap: Interim review and forward look.
ORCA, 2018. The State of European Cetaceans 2018.
ORCA, 2016. The State of European Cetaceans 2006-2015.
OSPAR, 2017a. Marine bird abundance, OSPAR Intermediate Assessment 2017.
OSPAR, 2017b. Extent of Physical Damage to Predominant and Special Habitats, OSPAR Intermediate Assessment 2017.
Paker, N., Vural, C.A., 2016. Customer segmentation for marinas: evaluating marinas as destinations. Tour. Manage. 56, 156–171. https://doi.org/10.1016/j.tourman.2016.03.024.
Paracchini, M.L., Zulian, G., Kopperoinen, L., Maes, J., Schägner, J.P., Termansen, M., Zandersen, M., Perez-Soba, M., Scholefield, P.A., Bidoglio, G., 2014. Mapping cultural ecosystem services: a framework to assess the potential for outdoor recreation across the EU. Ecol. Indic. 45, 371–385. https://doi.org/10.1016/j.ecolind.2014.04.018.
Peña-Alonso, C., Hernández-Calvento, L., Pérez-Chacón, E., Ariza-Solé, E., 2017. The relationship between heritage, recreational quality and geomorphological vulnerability in the coastal zone: a case study of beach systems in the Canary Islands. Ecol. Indic. 82, 420–432. https://doi.org/10.1016/j.ecolind.2017.07.014.
Pinn, E., Mitchell, I., Hawkridge, J., Macleod, K., 2018. Cetacean abundance and distribution: wide-ranging cetaceans [WWW Document]. CEFAS UK Mar. Online Assess. Tool. URL https://moat.cefas.co.uk/biodiversity-food-webs-and-marine-protected-areas/cetaceans/abundance-and-distribution-of-cetaceans-other-than-coastal-bottlenose-dolphins/ (accessed 1.27.20).
Pollesch, N., Dale, V.H., 2015. Applications of aggregation theory to sustainability assessment. Ecol. Econ. 114, 117–127. https://doi.org/10.1016/j.ecolecon.2015.03.011.
Portman, M.E., Shabtay-Yanai, A., Zanzuri, A., 2016. Incorporation of socio-economic features' ranking in multicriteria analysis based on ecosystem services for marine protected area planning. PLoS One 11, e0154473. https://doi.org/10.1371/journal.pone.0154473.
Possner, A., Caldeira, K., 2017. Geophysical potential for wind energy over the open oceans. Proc. Natl. Acad. Sci. 114, 11338–11343. https://doi.org/10.1073/pnas.1705710114.
Radford, Z., Ribeiro, J., Lynam, C., 2019. UK ICES fish stock and shellfish stock assessment data 2017 [WWW Document]. doi.org/10.14466/CefasDataHub.74.
RAMSAR, JNCC, 2015. Spatial and summary data for UK Ramsar sites.
RNLI, 2019. RNLI Lifeguards Report 2018.
RNLI, 2018a. RNLI 2018 Operational Statistics.
RNLI, 2018b. RNLI 2017 Operational Statistics.
RNLI, 2015. RNLI 2014 Annual Report and Accounts.
RStudio, 2019. RStudio: Integrated Development for R.
Ruijs, A., Heide, M. van der, Berg, J. van den, 2018. Natural capital accounting and the Sustainable Development Goals: Current and potential uses and steps forward. PBL Netherlands Environmental Assessment Agency, The Hague.
Ruiz-Frau, A., Kaiser, M.J., Edwards-Jones, G., Klein, C.J., Segan, D., Possingham, H.P., 2015. Balancing extractive and non-extractive uses in marine conservation plans. Mar. Policy 52, 11–18. https://doi.org/10.1016/j.marpol.2014.10.017.
Russell, D.J.F., Morris, C.D., Duck, C.D., Thompson, D., Hiby, L., 2019. Monitoring long-term changes in UK grey seal pup production. Aquat. Conserv. Mar. Freshw. Ecosyst. 29, 24–39. https://doi.org/10.1002/aqc.3100.
Ryan, C., Bolin, V., Shirra, L., Garrard, P., Putsey, J., Vines, J., Hartny-Mills, L., 2018. The development and value of whale-watch tourism in the west of scotland. Tour. Mar. Environ. 13, 17–24. https://doi.org/10.3727/154427317X14964473293707.
Scheili, A., Delpla, I., Rodriguez, M.J., 2020. Development of a drinking water quality index based on a participatory procedure using mixed multicriteria methods. Environ. Monit. Assess. 192 https://doi.org/10.1007/s10661-020-08516-7.
Scottish Government, 2019. Scotland's Natural Capital Asset Index 2019 - Detailed model (data to 2017).
Scottish Natural Heritage, 2019. Scotland's Natural Capital Asset Index.
Sedaghat, A., Hassanzadeh, A., Jamali, J., Mostafaeipour, A., Chen, W.H., 2017. Determination of rated wind speed for maximum annual energy production of variable speed wind turbines. Appl. Energy 205, 781–789. https://doi.org/10.1016/j.apenergy.2017.08.079.
Sibley, A., Cox, D., Titley, H., 2015. Coastal flooding in england and wales from atlantic and north sea storms during the 2013/2014 winter. Weather 70, 62–70. https://doi.org/10.1002/wea.2471.
Social Progress Imperative, 2020. Social Progress Index 2020.

England, Sport, 2017. Active Lives Online [WWW Document]. accessed 9.6.19. https://activelives.sportengland.org/.

Sport England, 2016. Active People Survey 10.

England, Sport, 2013. Active People Interactive [WWW Document]. accessed 9.6.19. https://activepeople.sportengland.org/.

StataCorp, 2017. Stata Statistical Software: Release 15.

Stebbings, E., Papathanasopoulou, E., Hooper, T., Austen, M.C., Yan, X., 2020. The marine economy of the United Kingdom. Mar. Policy 116. https://doi.org/10.1016/j.marpol.2020.103905.

Steinbach, N., 2017. How environmental accounts in Sweden are driven by changing policy need. In: Vardon, M., Bass, S., Ahlroth, S., Ruijs, A. (Eds.), Forum on Natural Capital Accounting for Better Policy Decisions: Taking Stock and Moving Forward. World Bank, Wealth Accounting and the Valuation of Ecosystem Services, Washington D.C.

Stephenson, F., Mill, A.C., Scott, C.L., Stewart, G.B., Grainger, M.J., Polunin, N.V.C., Fitzsimmons, C., 2018. Socio-economic, technological and environmental drivers of spatio-temporal changes in fishing pressure. Mar. Policy 88, 189–203. https://doi.org/10.1016/j.marpol.2017.11.029.

Stern, S., Krylova, P., Harmacek, J., 2020. 2020 Social Progress Index Methodology Summary.

Sunderland, T., Waters, R., Marsh, D., Hudson, C., Lusardi, J., 2019. Accounting for National Nature Reserves: A natural capital account of the National Nature Reserves managed by Natural England.

The Crown Estate, 2015. Offshore Wind Operational Report 2014. 10.1016/B978-0-12-410422-8.00003-0.

The Crown Estate, 2016. Offshore Wind - Operational Report 2015. 10.1016/B978-0-12-410422-8.00003-0.

The Crown Estate, 2018. Offshore Wind - Operational Report 2017. 10.1016/B978-0-12-410422-8.00003-0.

The Crown Estate, 2019. Offshore Wind Operational Report 2018. 10.1016/B978-0-12-410422-8.00003-0.

The RSPB, 2017. Accounting for Nature: A Natural Capital Account of the RSPB's estate in England. 10.1525/bio.2009.59.3.13.

Thompson, D., Duck, C.D., Morris, C.D., Russell, D.J.F., 2019. The status of harbour seals (Phoca vitulina) in the UK. Aquat. Conserv. Mar. Freshw. Ecosyst. 29, 40–60. https://doi.org/10.1002/aqc.3110.

Thornton, A., Luisetti, T., Grilli, G., Donovan, D., Phillips, R., Hawker, J., 2019. Initial natural capital accounts for the UK marine and coastal environment. Report prepared for the Department for Environment Food and Rural Affairs.

Tian, H., Lindenmayer, D.B., Wong, G.T.W., Mao, Z., Huang, Y., Xue, X., 2018. A methodological framework for coastal development assessment: A case study of Fujian Province. China. Sci. Total Environ. 615, 572–580. https://doi.org/10.1016/j.scitotenv.2017.09.309.

Tillin, H.M., Langmead, O., Hodgson, B., Luff, A., Rees, S.E., Hooper, T., Frost, M., 2019. Feasibility study for a Marine Natural Capital Asset Index for Scotland Asset Index for Scotland.

Tratalos, J.A., Haines-Young, R., Potschin, M., Fish, R., Church, A., 2016. Cultural ecosystem services in the UK: Lessons on designing indicators to inform management and policy. Ecol. Indic. 61, 63–73. https://doi.org/10.1016/j.ecolind.2015.03.040.

UK National Ecosystem Assessment, 2014. The UK National Ecosystem Assessment Follow-on: Synthesis of the Key Findings. Cambridge.

UK National Ecosystem Assessment, 2011. UK National Ecosystem Assessment: Synthesis of the Key Findings. UNEP-WCMC, Cambridge. 10.1177/004057368303900411.

United Nations, 2014a. System of environmental-economic accounting 2012 - Central framework. ST/ESA/STAT/Ser.F/109.

United Nations, 2014b. System of Environmental- Economic Accounting 2012—Experimental Ecosystem Accounting.

United Nations, 2009. System of National Accounts 2008.

United Nations Statistics Division, 2020. Chapter 6: Ecosystem services concepts for accounting, in: System of Environmental-Economic Accounting 2012 - Experimental Ecosystem Accounting Revision 2020. pp. 1–4.

Vallecillo, S., La Notte, A., Zulian, G., Ferrini, S., Maes, J., 2019. Ecosystem services accounts: Valuing the actual flow of nature-based recreation from ecosystems to people. Ecol. Modell. 392, 196–211. https://doi.org/10.1016/j.ecolmodel.2018.09.023.

Verbos, R.I., Altschuler, B., Brownlee, M.T.J., 2018. Weather studies in outdoor recreation and nature-based tourism: A research synthesis and gap analysis. Leis. Sci. 40, 533–556. https://doi.org/10.1080/01490400.2017.1325794.

Villamagna, A.M., Mogollón, B., Angermeier, P.L., 2014. A multi-indicator framework for mapping cultural ecosystem services: The case of freshwater recreational fishing. Ecol. Indic. 45, 255–265. https://doi.org/10.1016/j.ecolind.2014.04.001.

VisitBritain, 2019. Activities in Britain's nations and regions.

VisitBritain, 2018a. The GB Tourist 2018 Annual Report.

VisitBritain, 2018b. The Great Britain Day Visitor 2018 Annual Report.

VisitBritain, 2015. Great Britain Tourism Survey (GBTS): Statistics 2014.

Whiteley, G., Shabb, Katherine, Korkeala, Outi, Alan Mccullough, R.S., Shabb, K., Korkeala, O., Mccullough, A., Smithers, R., 2016. Reviewing cultural services valuation methodology for inclusion in aggregate UK natural capital estimates. Report for Office National Statistics.