Valuing the Solent Marine Sites Habitats and Species: A Natural Capital Study of Benthic Ecosystem Services and how they Contribute to Water Quality Regulation.

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Executive Summary

Excessive nutrient inputs (principally **nitrogen [N] and phosphorous [P]**) in the **Solent Marine Sites (SEMS)** are causing eutrophication, leading to a decline in water quality and an increase in the growth of green macroalgae on intertidal mudflats. These impacts can have adverse effects on the ecology and species within the UK nature conservation designation sites (e.g. overwintering birds) in and around the Solent, to which the Habitat Regulations apply. The impact on the condition of the sites is relevant in the context of meeting: legislative requirements (e.g. Water Framework Directive); protecting these habitats (e.g. Special Areas of Conservation, Special Protection Areas, Marine Conservation Zones); improving public health (e.g. reducing shellfish and bathing water contamination), but also for a viable and productive marine economy (e.g. sustainable aquaculture & fisheries and tourism).

As such the University of Portsmouth in March 2019 was commissioned by the Environment Agency (EA) to provide this strategic update of the natural capital value of habitats and species in the context of water quality for the Solent and Isle of Wight area. The overall aim of this study is to provide evidence to help value the changes in the level of ecosystem service that could result from changes in the quality of benthic habitats as a result of increasing or reducing nutrients such as N and P. This critical evidence base will enable the EA and other decision-makers (at both a national government and local authority level) to ensure natural resources are given the appropriate level of protection, whilst supporting sustainable economic growth. Together they will support the delivery of the UK government's 25-year plan for the environment, especially objectives set in Chapter 5: "*Securing clean, healthy productive and biologically diverse seas and oceans*".

The project was divided into three phases which are summarised below.

Part A Mapping natural capital stocks and estimating their capacity to remove nutrients

Following Office for National Statistics (ONS) guidance we assessed the extent of the SEMS marine natural capital stocks (habitats) defined using the **European Nature Information System** (**EUNIS**) habitat classification system. Baseline habitat assessments (in ha) have been made for: **littoral sediments (including with green algal mats), coastal saltmarsh, seagrass beds, reedbed (*Phragmites australis*) and subtidal sediments** from the Solent (Lymington Harbour to Pagham Harbour) and several inshore areas around the Isle of Wight (Yar estuary to Bembridge harbour). Due to the growing recognition of the ecosystem services provided by suspension-feeding bivalves (such as oysters) commercial **Native Oyster (*Ostrea edulis*)**, shellfish beds in the SEMS were also mapped. The habitat map created for the SEMS represents the 'best available evidence' at the time of writing this report in September 2020.

Using literature data from other temperate coastal systems we then assessed the N and P removal potential (*via* burial of N and P in underlying sediments and loss of N to the atmosphere [denitrification]) for each of the aforementioned habitats and species to determine annual ecosystem service flows. As N and P are also accumulated into biogenic material (e.g. the shell of bivalves), we have also incorporated this in to our calculations for native oyster beds. Results indicate that existing habitats in the SEMS could remove **3,590** tonnes of N yr^{-1} and **811** tonnes of P yr^{-1} based on each habitat's current Water Framework Directive (WFD) condition status. To represent how the local condition of habitats may influence ecosystem service flows, we used the 2016 WFD (cycle 2) waterbody summary condition assessments, together with their qualifying sub-feature assessments, for each of the habitats in each region of the SEMS. The level of baseline ecosystem service and that would be expected if all biotopes were adjusted based on their regional WFD condition classification data is presented in the ExSummary table 1 below.

ExSummary Table 1

Total Nutrient Removal	Baseline (Extent Only) (Tonnes yr^{-1})	Baseline (with Extent and WFD Condition) (Tonnes yr^{-1})	All Good WFD Condition (Tonnes yr^{-1})	All High WFD Condition (Tonnes yr^{-1})
Nitrogen	3264	**3590**	4487	5712
Phosphorus	636	**811**	915	1451
Total	**3900**	**4474**	**5402**	**7162**

Regional adjustments based on WFD condition suggest including local condition assessment data increased the overall amount of nutrients removed by regulatory services. However, many biotopes in the Solent were delivering at a higher (e.g. littoral sediments and reedbeds) or lower capacity (macroalgal mats and native oysters) than if we had only examined the baseline extent data. This is an important consideration, because condition assessments are often omitted when creating natural capital or ecosystem service accounts, potentially leading to an undervaluation of the UK's ecosystems services. Similarly, we estimated substantial annual uplifts in nutrient removal potential if prescribed policy goals (e.g. recovery of all habitats to good or high ecological status) were met in the near future.

At the WFD catchment level —**littoral sediments** were the largest contributing habitat for **N removal** in many regions including: Lymington Estuary, Beaulieu Estuary, Portsmouth Harbour, Langstone Harbour and Bembridge Harbour reflecting their large habitat area. However, **littoral sediments (overlain with macroalgal mats)** remain a large contributor to the potential N and P removal budget, particularly in Portsmouth and Chichester Harbours. The high efficiency of N and P removal by macroalgal mats most likely represents these algae forming thick canopies, which act as natural physical barriers, reducing N (and P) fluxes from the sediment to the overlying water, while at the same time the physical structure and growth stage of algal mats may play an important role in the regulation of N removal by denitrification. This capacity is likely to reduce as the Solent recovers from eutrophication. **On a relative per m^2 basis**—**Saltmarsh communities** are the most important habitat for N removal (see ExSummary table 2 below) and are the most important habitat for **overall N removal** in many of the smaller estuaries including; Pagham Harbour, the Yar Estuary, Newton Harbour and the Hamble Estuary. **Reedbed, littoral sediments (with macroalgal mats)** and **native oyster beds** all had high N removal potential (**on a per m^2 basis)**, while P removal is highest in **littoral sediments (overlain with macroalgal mats)** and **reedbed** habitats (**on a per m^2 basis**). However, at the catchment level many of these habitats' contributions to the **overall nutrient budget is often small because of the limited extent of each habitat**. Based on the median seagrass remediation rates we estimate a **negative efflux of P** from this habitat. It is important to note that the values calculated here are average annual values and if we had used the higher range biophysical values to represent P burial or considered seasonal patterns of seagrass burial then the values may be positive.

ExSummary Table 2

Nitrogen and phosphorus median removal rates	Coastal saltmarshes	Seagrass beds	Reedbeds	Littoral sediment	Littoral sediment (macroalgal mats)	Sublittoral sediment	Native oyster beds
Nitrogen (g N m^{-2} yr^{-1})	37.7	18.2	7.6	13.3	25	6.6	11.04
Phosphorous (g N m^{-2} yr^{-1})	4.7	-4.3	7.6	0.6	29.7	0.2	0.99

Habitat removal vs nutrient loads into the SEMS

To investigate the potential for different habitats to impact regional-scale water quality, we then combined the previously calculated annual N and P removal rates with the estimates of nutrient loading. EA modelling output undertaken for the Partnership for Urban South Hampshire (PUSH) was used for a catchment-level analysis. In the Solent, pollution from nutrients, mainly comes from agriculture (50%) and coastal background sources (40%) with only 10% estimated to originate from urban discharges. Each year, it is estimated that about **5,016 tonnes of N and 602 tonnes of P** enters the SEMS; with the largest loadings entering the highly urbanised Southampton Water catchment (approximately 30% and 40% of total N and P loads respectively). Budget calculations show that on an annual basis, estuarine habitats retain and **remove 35% of total N loading** and **91% of total P** that would otherwise pass into the Solent channel. These results suggest most of the estuarine P is tied up in the biota of the estuary, especially in **macroalgal mat and saltmarsh** habitats. Regional estimates for the proportion of the N load that could potentially be removed by habitats varied widely between the catchments, ranging from **11% in Bembridge Harbour to 92% in the case of Langstone Harbour**. The comparatively high remediation capacity of habitats in Langstone harbour is a result of diversions of nutrient inputs away from the harbour in recent years. However, our analysis does highlight that a relatively large proportion of N removed in Langstone Harbour (~25%) is still achieved *via* the growth of green macroalgal mats which are often associated with detrimental effects on infaunal species and wading bird populations (Thornton 2016). Overall, these results serve as an important baseline for the current removal capacity of habitats in the SEMS region as well as a framework for considering the potential use of any habitat or species to ameliorate or remediate eutrophic conditions.

Economic valuation of nutrient removal ecosystem services

To generate relevant estimates of economic value associated with natural N and P removal, we rely on actual mitigation costs of nutrient reduction measures undertaken on the UK's southeast coast. Replacement costs for removing a kilogram of N vary substantially so were extracted from a combination of nutrient management and planning documents for Poole Harbour (Bryan *et al.,* 2013; RSPB 2013; BPPDC 2017), which together provide some of the most comprehensive regionally-focussed valuation estimates for N in the UK. Mitigating costs for additional P loads to achieve neutral development were taken from an interim (2019-2025) plan for the River Avon (RAWG, 2019) another neighbouring catchment of the Solent. Based on the average replacement costs, total N and P removal by the SEMS biotopes are estimated to be worth in the region of: **£962 million (N)** and **£179 million (P)**. More refined estimates, factoring in the condition of the biotopes in the SEMS, would generate values of just over **£1.1 billion (N)** and **£228 million (P),** respectively. These valuations have also been disaggregated to provide nutrient reduction values (£) specific to different regions in the SEMS. We also acknowledge that the cost of providing alternative mitigation measures is likely to vary significantly by watershed, depending on what is feasible given the extent of nutrient pollution. Thus, we believe that the average replacement cost price for N and P used in this study provides a good approximation of a typical cost of interventions to improve local water quality.

Part B Additional ecosystem services relating to water quality

The analysis of additional individual ecosystem services with potential links to water quality were also considered in this report. These included:

- Climate regulation (carbon sequestration and storage)
- Commercial, recreational and subsistence fisheries.
- Nursery function and supporting the existence of biodiversity.
- Recreation, tourism and leisure.

The table below shows the basic structure of possible monetary use tables for the SEMS marine and coastal environments following ONS guidance of building UK natural capital accounts.

ExSummary Table 3

Ecosystem service	Current Status (Tonnes yr^{-1})	All Good Condition (Tonnes yr^{-1})	All High Condition (Tonnes yr^{-1})
Waste remediation (nitrogen stored t yr^{-1})	£1,059.16 M	£1,323.67 M	£1,685.04 M
Waste remediation (phosphorus stored t yr^{-1})	£228.63 M	£258.03 M	£409.18 M
Climate regulation (carbon stored t yr^{-1})	£2.30 M	£2.97 M	£4.87 M
Commercial, recreational and subsistence fisheries. (catch t yr^{-1})	£13.85 M	£13.98 M	£14.31 M
Nursery function and supporting the existence of biodiversity.	Not assessed/limited data	Not assessed/limited data	Not assessed/limited data
Recreation, leisure and tourism	£2.73 M	£3.18 M	£3.59 M
Total	**£1,304.38 M**	**£1,601.82 M**	**£2,116.99 M**

Even with the consideration of only six ecosystems services, some narrowly described, the current annual value of the flow of goods and services from SEMS marine and coastal ecosystems is impressive, estimated here at over **£1.3 billion per annum.** The harvest of finfish and shellfish is the ecosystem service that is perhaps the most familiar to people for several reasons ---employment, cultural history, culinary tradition--- but in the larger context its economic value appears relatively minor. The relative importance of regulating services, namely the removal of N and P, compared to the other services is most notable. Supporting, habitat and cultural ecosystem services such as **nursery function and recreation, tourism and leisure are considered to be underestimated**, due to insufficient data or limited to willingness to pay values which exclude on-the-water experience, direct revenues from tourism as well as the broader cultural appreciation of Solent maritime heritage. We also estimate that if water quality improvements were implemented to improve all habitats to "Good" or "High" WFD status approximately **£298- 812 M yr^{-1}** of additional monetary benefits could be accrued.

Part C Assessing multiple stressors and impacts on benthic habitats and ecosystem services

The final section of this report assessed negative changes in the level of ecosystem service(s) that could result from changes in the quality of benthic biotopes, in the context of current and future anthropogenic stressors. We considered biotope impacts associated with four important stressors that directly or indirectly impact on water quality:

- Physical abrasion from mobile fishing gears,
- Introduction of microbial pathogens (*Escherichia coli*),
- Increase N and P inputs (eutrophication),
- Climate change driven activities (sea level rise)

Sensitivity assessments following the Marine Evidence-based Sensitivity Assessment (MarESA) methodology have been undertaken for each of the stressors relating to biotopes at Levels 4 to 5 of the EUNIS classification system. These assessments, which are available on the Marine Life Information Network (MarLIN) website, are updated by the Marine Biological Association of the United Kingdom (MBA) using best available evidence and peer review. These contain the aggregated sensitivity, resistance and resilience outputs for benthic habitats. The work utilises the MarESA sensitivity assessments as well as the outputs from previous projects, including fishing vessel positional monitoring system (VMS) data, modelled sea level rise data (LiDAR flooding) from the Solent Dynamic Coast Project [SDCP] and habitat compensation targets from the Solent Regional Habitat Compensation Programme (RHCP). The outputs from the MarESA-stressor modelling concluded that potential water quality related replacement cost savings from reducing all four stressors is estimated to be **£516.25 million per year.** Available evidence also suggests that if projected saltmarsh habitat loss due to sea level rise continues then **£5.5 billion per year**. **(3.5% discount rate)** in terms of regulatory service benefits will not be realised over the next century. Current thresholds modelled here are largely precautionary as potential options for meeting future habitat compensation targets are likely to change and evolve as more locally specific policy thresholds are designed that can support a 'net gain' approach for the marine systems.

Conclusions and recommendations

The enhanced significance of taking a natural capital approach in this report is that for the first time a comprehensive and consistent list of indicators for assessing and valuing water quality in the Solent are collected. The indicators can be used to map and assess ecosystem service flows based on extent and condition per habitat type. The framework also allows a relative ecosystem "value" to be placed on key habitats and species in a consistent manner that can then be applied in horizontal assessments across different ecosystems. The estimated value of habitat-mediated N and P removal, for example can help policymakers move from a position of taking no monetary account of issues relating to 'likely' significant adverse effects caused by eutrophication resulting from increasing nutrient levels to demonstrating indicative monetary values in environmental impact appraisals. Similarly, the non-monetary results can help guide where to focus sub-national policy making efforts e.g. direct estimates of mitigation of nutrients by habitats in tonnes yr^{-1} can be useful to compare whether new housing and development growth can be accommodated without having a detrimental effect upon the coastal environment. In conclusion, a number of recommendations are outlined in the final section of this report (Pages 81-84). These are summarised here and include:

- **Gaps in mapping and valuation evidence:** Key areas that may require further mapping effort to support the use of indicative values include seagrass and native oyster beds. Similarly, several EUNIS habitats (e.g. kelp beds, polychaete reefs, maerl beds, epiphyte and sponge communities) only comprised small areas in the SEMS but could potentially be important contributors to N and P removal. Littoral and subtidal sediments (including those overlain with macroalgal mats) are often overlooked in a nutrient removal context yet provided substantial N and P removal services. There is clear evidence from EA data that macroalgal mats are beginning to decline across several areas of the Solent. More research into the different ecosystem services (and disservices) provided by macroalgal mat assemblages would also be important to allow management trade-offs to be made.

- **Nitrate neutrality and improving water quality in the Solent:** From a management perspective our N and P loading vs habitat uptake analyses suggest that even if “nutrient neutrality” is achieved for new developments in the SEMS (e.g. *via* nitrogen credits or offsetting), greater nutrient reductions (i.e. not just maintaining the status quo) will be required if habitats and species in the Solent are to fully recover from the impacts of eutrophication. The results of the assessment which was undertaken at a waterbody level identified that strong N limitation was only found in Langstone Harbour (watershed N bioremediation rate of 92%). The best available evidence is for a focus across all Solent estuaries and harbours to be on N reductions and for developments that are impacting on Southampton Water, Chichester Harbour and Pagham Harbour, a P budget may be required. Given the continuing need to reverse historic environmental declines and prepare for new developments and climate change, this report shows where cost-effective investment will enhance the nutrient removal services provided by the Solent’s waterbodies; and thus, increase the Solent’s (and indeed the UK’s) overall natural capital.

- **Including ecosystem condition and other ecosystem services in water quality assessments:** Overall, the conclusion from the initial natural capital accounts is clear that restoring and/or improving the existing condition of biotopes should be seen as a major consideration for management in the Solent. Water quality assessments may best inform policy if modelled changes are presented not just as concentrations of N or P removed but also in terms of bundles of other ecosystem services. We have addressed this knowledge gap by introducing a generalisable framework for the assessment and valuation of water quality services. However, greater understanding of economic valuations of individual and bundled services is also needed. Specific ecosystem services not addressed here but, that would be useful to value in a water quality context include: fresh water provisioning, sediment stabilisation, natural hazard protection (e.g. floods, storms), raw materials (e.g. biofuels) and other marine “wastes” (e.g. heavy metals, persistent pollutants, microplastics, radioactive substances.

- **Policy relevance of cumulative impacts on water quality related ecosystem services**: There are significant gaps in the scientific understanding of how ecosystem services flowing from habitats change in response to multiple stressors. The conclusions reached through this part of the study are caveated due to uncertainties in the scientific data and modelling (such as inaccuracies in benthic habitat and fisheries abrasion data). Nevertheless, these values are indicative of how stressors such as sea level rise are affecting the productivity and health of the marine environment, and the water quality derived services society receives from it. The information developed here can be used to better understand the potential value (£) risk of losing ecosystem services of the coastal margins, which will be under increasing threat as climate change proceeds. Including biophysical trade-offs, of different habitat compensation requirements will also enable environmental net gain approaches to address key issues such as climate change, waste, nutrient pollution and natural hazards. A valuation study looking more explicitly at the future recovery of marine habitats and by extension ecosystem services would be worthwhile.

List of acronyms

AIH	Available Intertidal Habitat
B	Billion
BAP	Biodiversity Action Plan
Biotope/habitat	In this report the term habitat and biotope are used interchangeably
BPPDC	Borough of Poole and Purbeck District Council
C	Carbon
CBA	Cost-Benefit Analysis
CBT	the Convention of Biological Diversity
CCO	Coastal Channel Observatory
CEFAS	Centre for Environment, Fisheries and Aquaculture Science
CICES	Common International Classification of Ecosystem Services
CIS	Common Implementation Strategy
CSF	Catchment Sensitive Farming
DECC	Department of Energy and Climate Change
Defra	Department for Environment, Food and Rural Affairs
DIN	Dissolved Inorganic Nitrogen
DIP	Dissolved Inorganic, Phosphorus
DON	Dissolved, Organic Nitrogen
DOP	Dissolved Organic Phosphorus
EA	Environment Agency
EEA	European Economic Area
EMODnet	The European Marine Observation and Data Network
ENCA	see DEFRA Enabling a Natural Capital Approach
EO	Earth Observation
EQS	Environmental Quality Standards
ES	Ecosystem Services
EUNIS	European Nature Information Systems
EUSeaMap	EMODnet broad-scale seabed habitat map for Europe
FSA	Food Standards Agency
GDP	Gross Domestic Product
GES	Good Ecological Status
GIS	Geographic Information System
GVA	Gross Value Added
HIWW	Hampshire & Isle of Wight Wildlife Trust
ICES	International Council for the Exploration of the Sea
IWMS	Integrated Water Management Study
JNCC	Joint Nature Conservation Committee
LiDAR	Light Detection and Ranging Data
LS	Littoral Sediment
LSMA	Littoral Sediment with Macroalgae
M	Million
MarESA	Marine Evidence-based Sensitivity Assessment
MarLIN	Marine Life Information Network
MESH	Mapping European Seabed Habitats project
MMO	Marine Management Organisation
MPA	Marine Protected Area
MSFD	Marine Strategy Framework Directive

N	**Nitrogen**
NBA	**National Biodiversity Atlas**
NC	**Natural Capital**
NDPB	**Non-Departmental Public Body**
NE	**Natural England**
NERC	**Natural Environment Research Council**
NPV	**Net Present Value**
NTZ	**No Take Zones**
NWEBS	**National Water Environment Benefits Survey**
ONS	**Office for National Statistics**
OSPAR	**The Oslo-Paris Convention for the Protection of the Marine Environment of the North-East Atlantic**
OY	**Native Oysters**
P	**Phosphorus**
PES	**Payments for Ecosystem Services Schemes**
PN	**Particulate Nitrogen**
PP	**Particulate, Phosphorus**
PSG	**Project Steering Group**
PUSH	**The Partnership for Urban South Hampshire**
RAWG	**River Avon SAC Working Group**
RE	**Reedbeds**
RHCP	**Regional Habitat Compensation Programme**
RSPB	**The Royal Society for the Protection of Birds**
SAC	**Special Areas of Conservation**
SDCP	**Solent Dynamic Coast Project**
SEEA	**United Nations System of Environmental Economic Accounting**
SEMS	**Solent Marine Sites (formerly the Solent European Marine Sites)**
SG	**Seagrass**
SIFCA	**The Southern Inshore Fisheries and Conservation Authority**
SM	**Saltmarsh**
SNPs	**Shoreline Management Plans**
SPA	**Special Protected Area**
SS	**Subtidal Sediment**
SSD	**Solent and South Downs**
TEEB	**The Economics of Ecosystems and Biodiversity**
UK NEAFO	**UK National Ecosystem Assessment Follow-On**
VMS	**Vessel Positional Monitoring System**
WFD	**Water Framework Directive**
WTP	**Willingness to Pay**

Table of Contents

1. Introduction

This is the final report for project ENV6003066R - ***Valuing the Solent Marine Sites Habitats and Species: A Natural Capital Study of Benthic Ecosystem Services and How they Contribute to Water Quality Regulation***. The overall aim of this project is to investigate and assess the natural capital value of the Solent Marine Sites (SEMS) in terms of the function of coastal habitats and key species controlling water quality. More specifically the project considers the changes in ecosystem service provision that could result from changes in the extent and quality of benthic (seabed) habitats. The total quantity (area) of benthic habitats is fixed (as the area of the Solent marine environment), and the changes in the relative quantities (areas) and qualities (condition) of different habitats are considered in the context of current and future stressors on water quality. Improving the evidence base regarding the value of Nitrogen (N) and Phosphorous (P) inputs is of crucial importance to marine and coastal regulators such as the Environment Agency (EA) and other stakeholders interested in pursuing improvements in water quality. The evidence and model predictions provided in this report are relevant to different marine policy contexts and can be integrated with decision support tools such as Cost-Benefit Analysis (CBA) when considering possible regulatory and policy interventions, as well as making the case for investment in natural systems.

The project was divided into three key phases:

Part A. Assess the current natural capital value of the SEMS in terms of the function of habitats and key species controlling water quality (particularly relating to N and P pollution).

- **Section 2**: Provides an overview of the methodology that has been developed and employed in this project including a definition of the boundary of the SEMS marine ecosystem and identifies which marine habitats to differentiate as sub-assets.
- **Section 3**: Identifies the capacity of habitats to provide N and P removal ecosystem services based on their current extent and condition.
- **Section 4**: Describes the method adopted to investigate the economic value of the N and P removal ecosystem services and sets out illustrative accounts, on the basis of the data obtained so far.
- **Section 5** Draws conclusions and discusses limitations from the evidence identified in part A and discusses data gaps, uncertainty in the N and P accounts

Part B. Extend the assessment to include the wider natural capital benefits of water quality improvements.

- **Section 6** Builds on the analysis in Part A by quantifying and valuing a number of other water quality-related ecosystem goods and services including; carbon sequestration and storage, fisheries, nursey function and recreation, tourism and leisure services in the SEMS.

Part C. Model ecosystem services changes from benthic biotopes in the context of current and future anthropogenic stressors.

- **Section 7** Outlines the general principles that has allowed benthic habitat sensitivities and their risk of impact to be used to estimate a change in the level of ecosystem service as a result of a change in multiple stressors exerted on benthic habitats.

Part A: Using Marine Biotope Data to Link Natural Capital with Improved Water Quality Outcomes

2. Methodology

This section focuses on how marine natural capital accounts might be developed for the SEMS in the context of mitigating N and P pollution. It also defines the boundary of the marine ecosystem and identifies which marine habitats to differentiate as sub-assets.

2.1 Overall Approach

The method that has been developed takes into account the unique characteristics of the marine environment, building on the principles from the conventional stock and flow accounts (e.g. Environmental Accounts produced by the Office for National Statistics (2017) and the importance of natural assets to society. The principles of stocks and flows are the framework for organising what is being measured from the marine ecosystem. This is consistent with conventional concepts of natural (ecosystem) capital which define it as something that is productively valuable. Key aspects of the approach are that accounts are measured in both physical and monetary units (wherever possible), and that the value of the stock of a natural capital asset is determined through the value of the flows of services it is expected to provide over time. The conceptual framework that underpins ecosystem asset accounting, linking ecological condition with economic output, is described in Figure 1. Broadly, the accounting framework includes assessment of both stocks and flows, in monetary and non-monetary terms. The non-monetary accounts consider the extent and quality of stocks, and quantities (rather than values) of ecosystem services and thus overlaps with concepts of a natural capital asset registers and condition assessments, with the expectation that accounts and associated parameters would be quantified and recorded at regular intervals (usually annually). In this report defining the pathways between ecosystem stocks and ecosystem assist values has been undertaken through a staged approach:

- **Define the boundary to assess the Solent's marine natural capital assets and identify which marine habitats to differentiate as sub-assets –** Thirteen WFD catchment areas within and adjacent to the SEMS area were mapped (more details below Section 2.2.) using a European Nature Information System (EUNIS) approach. Seven main EUNIS habitats were mapped including; reedbeds (*Phragmites australis*), saltmarsh, seagrass (*Zostera* spp.), intertidal sediments, subtidal sediments and native oyster beds (*Ostrea edulis*). For the first time we also calculate the contributions of littoral sediments overlain with green macroalgal mats separately within an EUNIS framework.

- **Identify relevant indicators, data sources and proxies that represent the biophysical rates that contribute toward the long-term storage/removal of nutrients in the Solent –** We then collated from published sources biophysical N and P removal rates as a function of coupled nitrification-denitrification (N only) and long-term burial in sediments (N & P) for each of the key habitats linked to the ecosystem service of waste (nutrient) remediation.

- **Explore the options for the valuation of the service flows (and hence the asset value relating to those services) –** The above data were combined with avoided wastewater treatment costs to produce robust economic values. This provides conservative valuation estimates (UK £) for offsetting N and P in northern European coasts, but we also consider a range of remediation options with the lowest (Catchment Sensitive Farming [CSF]) and highest (water treatment works from upstream point sources) costs.

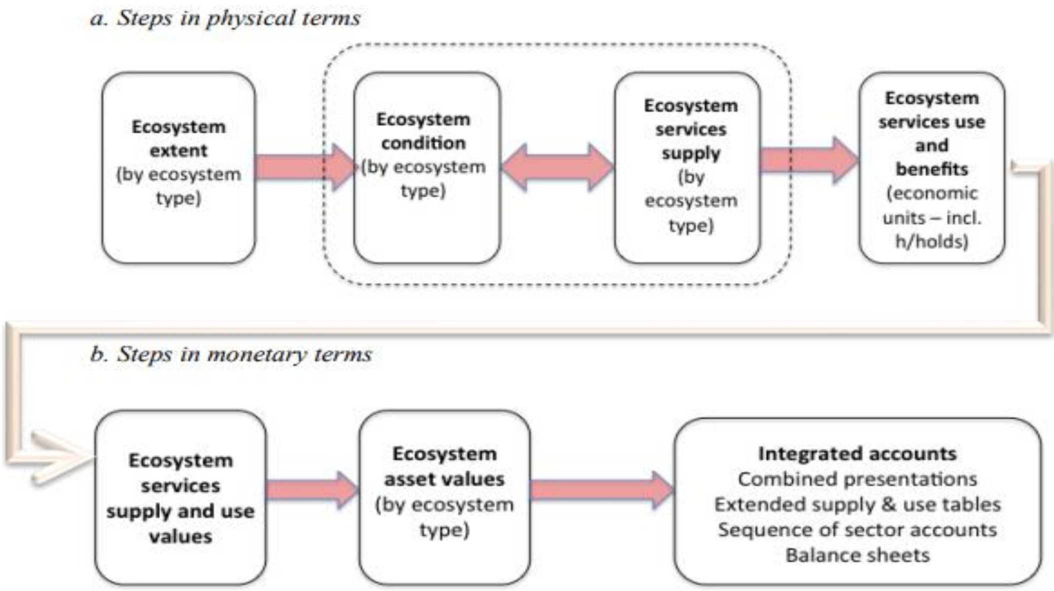
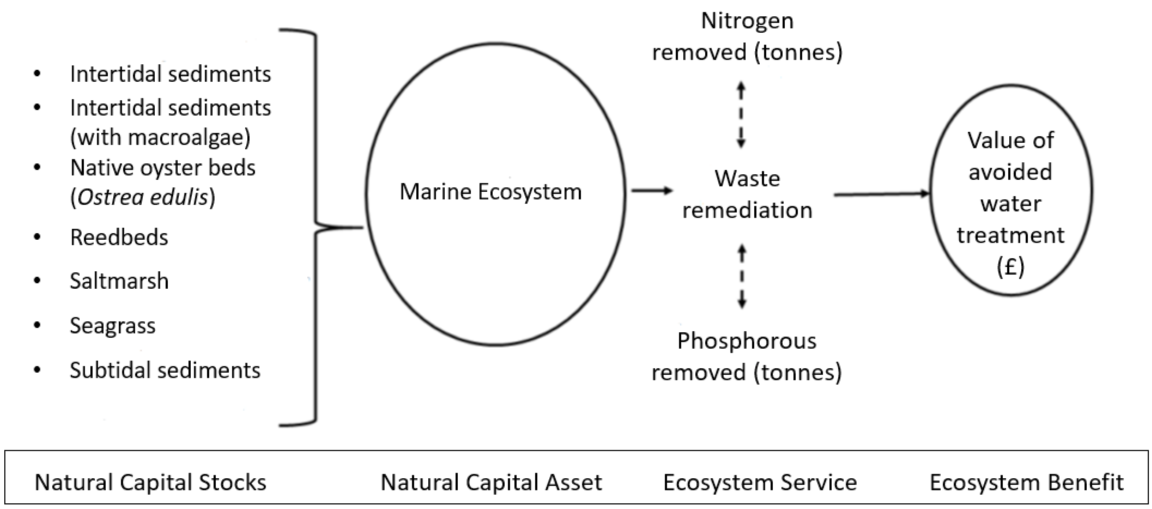


Figure 1 Broad steps in ecosystem accounting. The dotted line around the boxes for ecosystem condition and ecosystem services supply indicates that measurement of these aspects may often be completed in parallel, and iteration between them is appropriate in developing a single best picture. Also, while the figure indicates a progression from physical to monetary, for some provisioning services direct estimation of monetary values may be undertaken, or estimates for the accounts may be taken from existing studies. Experimental Ecosystem Accounts (SEEA-EEA 2014).)

The logic chain for assessing the Solent's natural capital stocks with regards to water quality are depicted in Figure 2.

Figure 2 Logic chain for assessing the provision of waste (nutrient) remediation from the marine environment.



2.2 Overview of the Solent Marine Sites (SEMS)

The SEMS is one of a number of Marine Sites in the UK which are designated as internationally important sites for their habitats and species. The SEMS covers the harbours, estuaries, areas of open coast and inshore water around the Solent. The site stretches from Hurst Spit in the west to Chichester Harbour in the east and includes areas along the north coast of the Isle of Wight from Yarmouth to Bembridge Harbour, as well as the mainland shores; the SEMS is 781.1km^2 in size (Figure 3). Much of the SEMS is of high nature conservation value, however these important sites are under pressure or impacted as a result of the highly populated, urbanized coast leading to significant exploitation. For example, it is estimated 65% of saltmarsh habitat has been lost in the SEMS between 1946 and 2013 and an estimated 98% reduction in native oyster landings has been recorded between 1978 and 2013 (Watson *et al.,* 2020).

There are eleven designated Water Framework Directive (WFD) transitional and coastal assessment units within the SEMS which were used here to delineate catchment assessment boundaries. Although not formally within the SEMS, we also included the WFD catchment of Pagham Harbour, a site located at the east end of the Solent which is a local nature reserve and has Special Protected Area (SPA) status. In addition, although the Hamble Estuary forms a subsection of the Southampton Water WFD waterbody, here we separated this catchment into a separate WFD assessment unit. The Solent strait WFD boundary was also extended at its east most edge to capture seagrass habitats that might have been missed using the original classification. The SEMS itself contains several international nature conservation sites including: two maritime Special Area of Conservation (SAC); Sites of Scientific Interest (SSSI), five SPAs, four Ramsar sites and five Marine Conservation Zones within the complex. These designations have varying levels of overlap illustrated in Figure 4. As a network of sites, these zones contribute to fulfilling the United Kingdom's obligations under the Convention of Biological Diversity (CBD) as well as non-binding instruments such as the recommended coherent network of marine protected areas (MPA's) under the OSPAR (Oslo and Paris Conventions) Recommendation 2003/3.

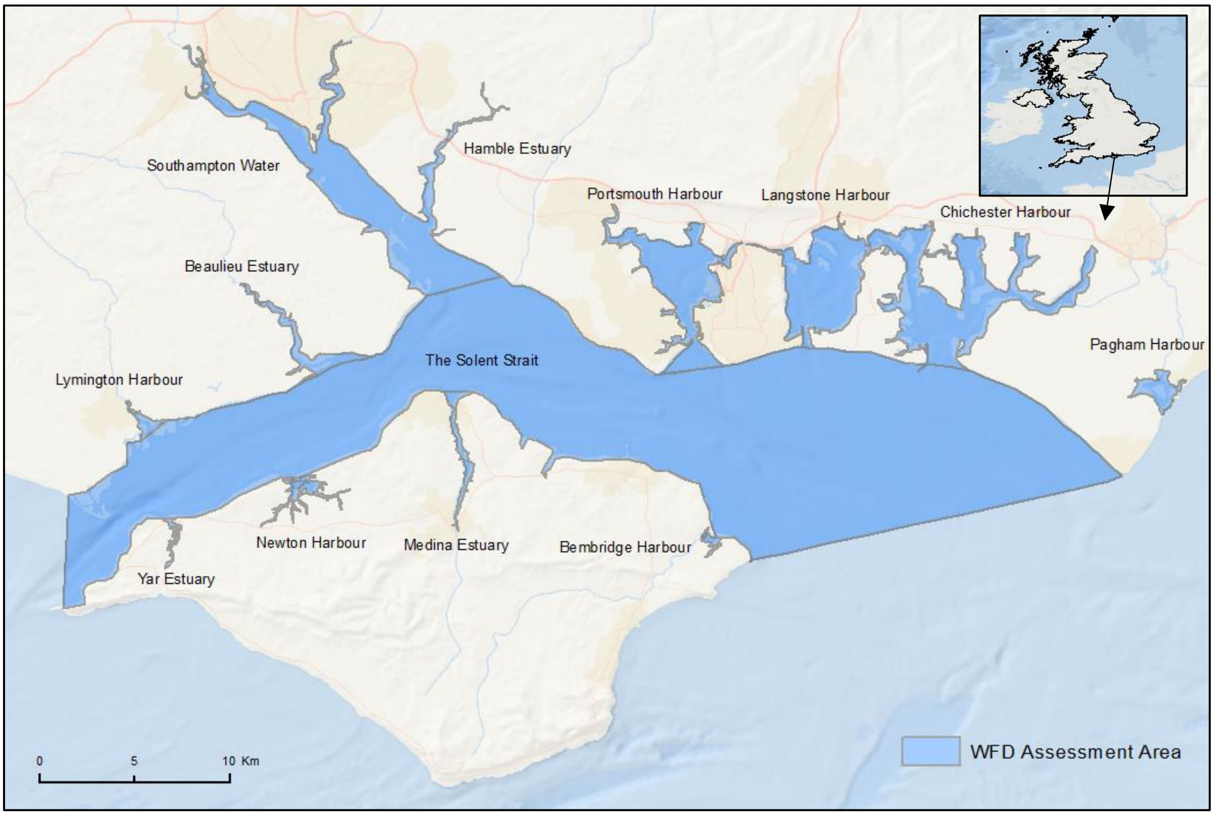


Figure 4 The assessment area (dark blue) for natural capital services and value relating to water quality in the Solent. The geographic units (i.e. individual polygons) chosen for sub-dividing and mapping the area are the WFD transitional and coastal assessment units.

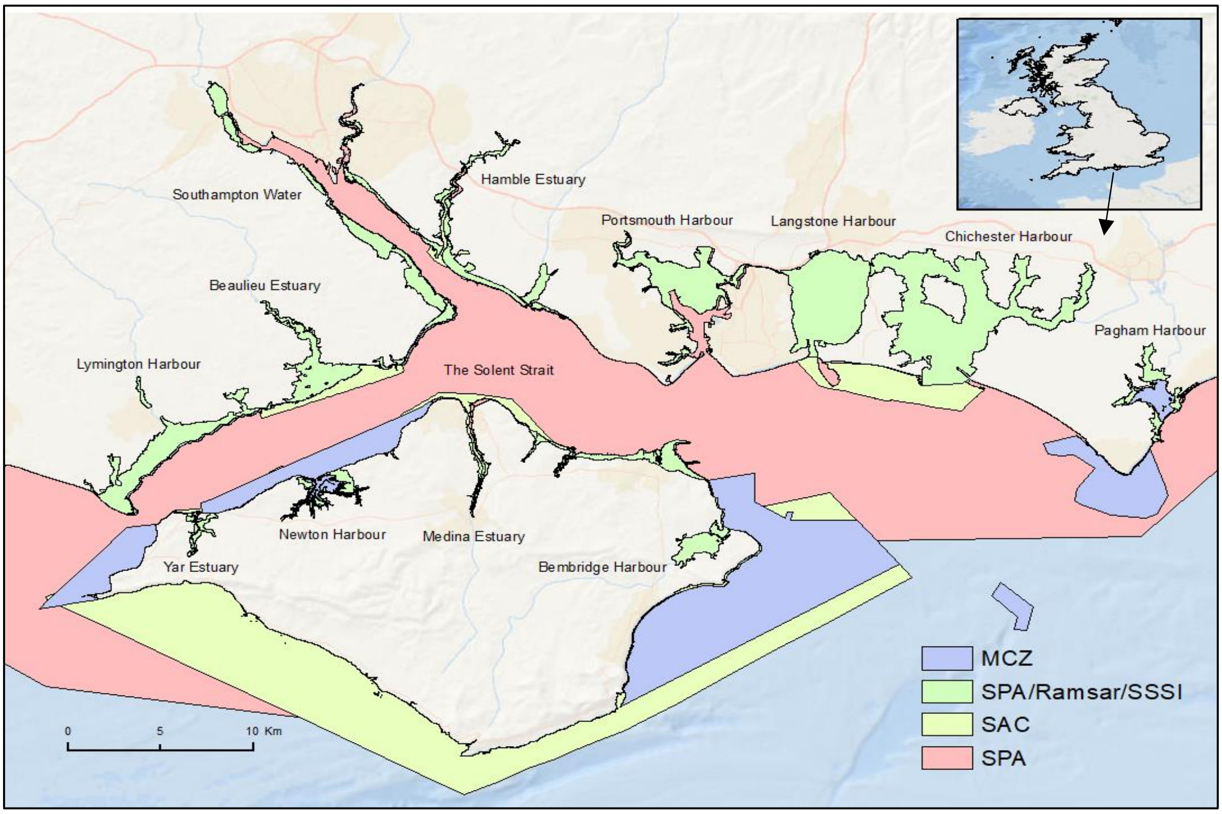


Figure 3 Designated marine and intertidal conservation sites within the SEMS (as of January 2020).

2.2.1 Mapping Natural Capital Stocks

The environmental features, and habitats present within the SEMS, up to the mean high water (MHW), were derived from 'best available (as of September 2020)' habitat map data available for the region. A range of GIS broad-scale SEMS habitat maps and datasets were utilised (Table 1) and combined using Arc GIS (version 10.7). A composite habitat map was generated that combined the available spatial data sets (Figure 5). Data were accessed through three sources 1) Internal EA habitat datasets, compiled from best available survey maps 2) Coastal Channel Observatory habitat survey data and 3) Modelled data from EMODnet/EUSeaMap (2018). The map aims to provide habitat data, where possible at EUNIS levels 3-5. As, by definition, habitat types at EUNIS level-4 and above are determined by both their biotic and abiotic features, the underlying assessment units (habitats) are hereafter addressed as 'biotopes'(Salomidi *et al.,* 2012). EUNIS classifications were aggregated for systems with similar or highly comparable mechanisms of nutrient exchange (e.g. A.2 littoral sediments and A.5 subtidal sediments). Ancillary non-spatial datasets on macroalgal mat communities and native oyster (*Ostrea edulis*) beds were also sourced from local monitoring programmes (Appendix 1). To estimate the native oyster biotope area, locations of commercial beds, active oyster dredge areas and numbers of individuals caught yr^{-1} in key sections of the SEMS were combined. Seagrass (*Zostera* spp.), and green macroalgal mat areas are based on the largest estimates of coverage within the SEMS using surveys conducted from 2006-2018. A disaggregated summary of biotopes present in the 12 Water Framework Directive (WFD) transitional and coastal assessment units (including the Hamble estuary) of the SEMS are also provided in Appendix 1.

A confidence assessment for classified seabed maps, was also produced (Table 1), indicating the likelihood of a particular biotope being correctly mapped within a study area. Confidence was based on JNCC confidence scores (Lillis, 2016). The JNCC Confidence Assessment Scheme is a systematic approach using a multi-criteria questionnaire to score biotope maps derived from survey data according to three key aspects: remote sensing, distinctness of class boundaries and amount of sampling. The scoring framework assigns each biotope map with a score between 0 (Low) and 4 (High; Figure 6). This enables end-users to determine the adequacy of the data-layers for decision-making, and future survey effort can be directed to low-scoring areas. The use of the UKSeamap, returns a high JNCC confidence score (4) because it is accompanied by a Mapping European Seabed Habitats project (MESH) confidence map. Frequent aerial imagery collection in combination with satellite and ground truthing are used by the Environment Agency for macroalgal mat extent and density and, therefore, scores highly (4). In contrast, low JNCC confidence scores (1-2) for seagrass and native oyster beds reflect significant spatial and temporal data gaps in the region (e.g. Portsmouth and Chichester Harbours for native oyster beds). In the case of seagrass, available datasets did not allow for comparisons in seagrass distribution between years, due to differing seagrass beds surveyed each year highlighting a lack of consistent annual sampling across the whole of the SEMS. This is partly because the subtidal components of these biotopes cannot be easily delineated from other biotopes using earth observation imagery, meaning they have to be sampled directly (by boat or walkover); inevitably reducing the availability and accuracy of coverage data.

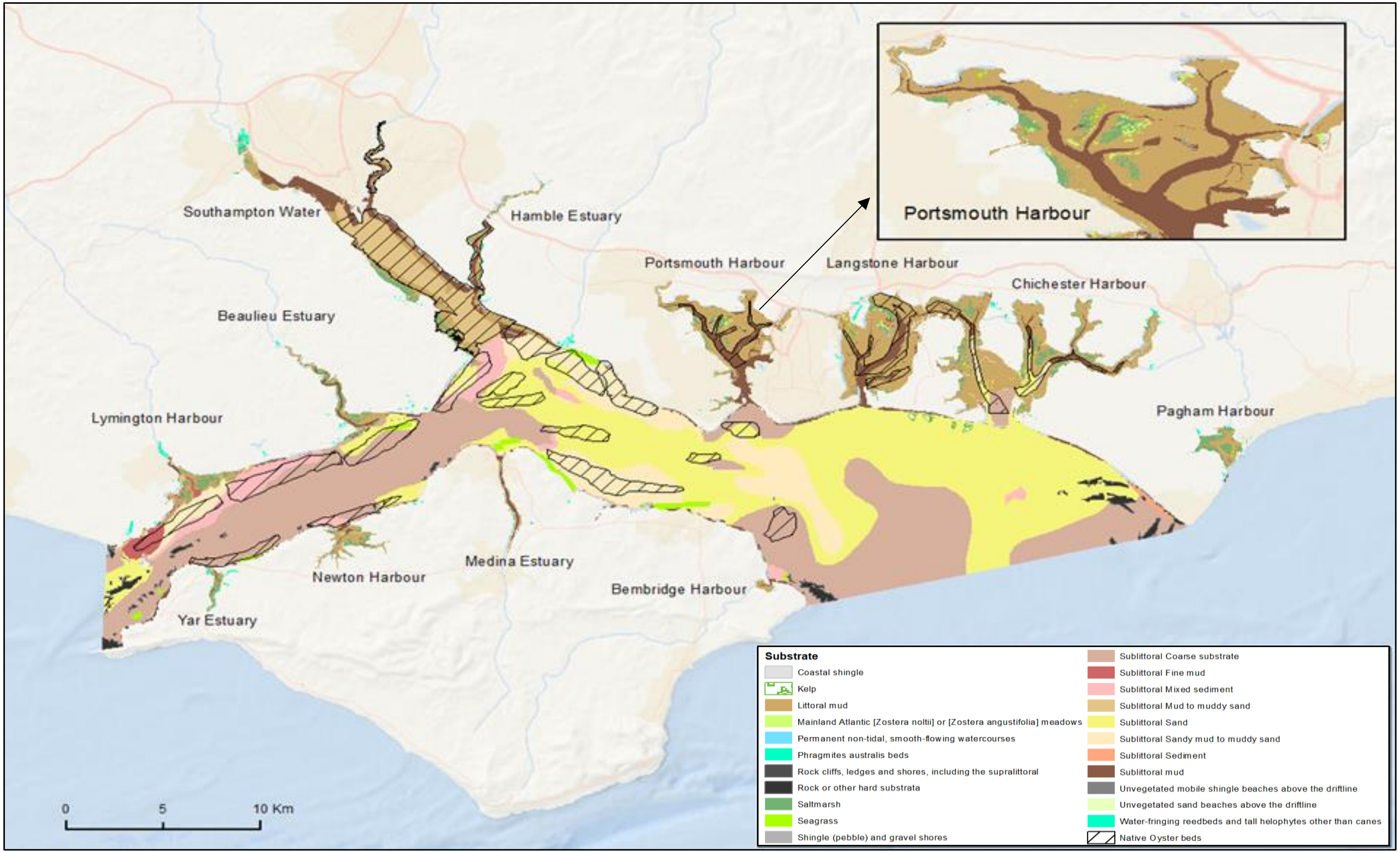


Figure 5 Mapped extent of biotopes (EUNIS Level 3 or greater) within the SEMS, showing Portsmouth Harbour as an example in more detail.

Table 1 Biotope data available for the SEMS from the following data sources: Coastal Channel Observatory (CCO), Environment Agency (EA), Centre for Environment, Fisheries and Aquaculture Science (CEFAS), Hampshire & Isle of Wight Wildlife Trust (HIWWT), The Southern Inshore Fisheries and Conservation Authority (SIFCA) and The European Marine Observation and Data Network (EMODnet).

Waterbody-Survey year	Dataset Name	Source	Type	JNCC Confidence	References/Notes
2006-2014	Inventory of eelgrass beds in Hampshire and the Isle of Wight	HIWWT	Survey	1	Seagrass biotopes; Marsden & Chesworth, (2014).
2018	West Isle of Wight & North Solent Subtidal Seagrass Surveys 2018	EA	Survey	4	Seagrass biotopes
2013	Classification of Bivalve Mollusc Production Areas in England and Wales- Portsmouth Langstone and Chichester Harbour- Sanitary Survey Report(s)	CEFAS	Survey	2	Main native oyster biotopes in Portsmouth, Langstone and Chichester Harbours.
2013	Southeast Regional Coastal Monitoring Programme- Terrestrial Ecological Mapping	CCO	Survey	2.5	Littoral sediment, saltmarsh and reedbed biotopes.
2016	Environment Agency Saltmarsh Zonation - December 2016 update	EA	Survey	4	Saltmarsh biotopes.
2018	Solent Oyster Fishery 2018 Stock Survey Report	SIFCA	Survey	2	Native oyster biotopes Solent and Southampton Water.
2018	EUSeaMap 2018	EMODnet	Modelled	4	Subtidal sediment biotopes.

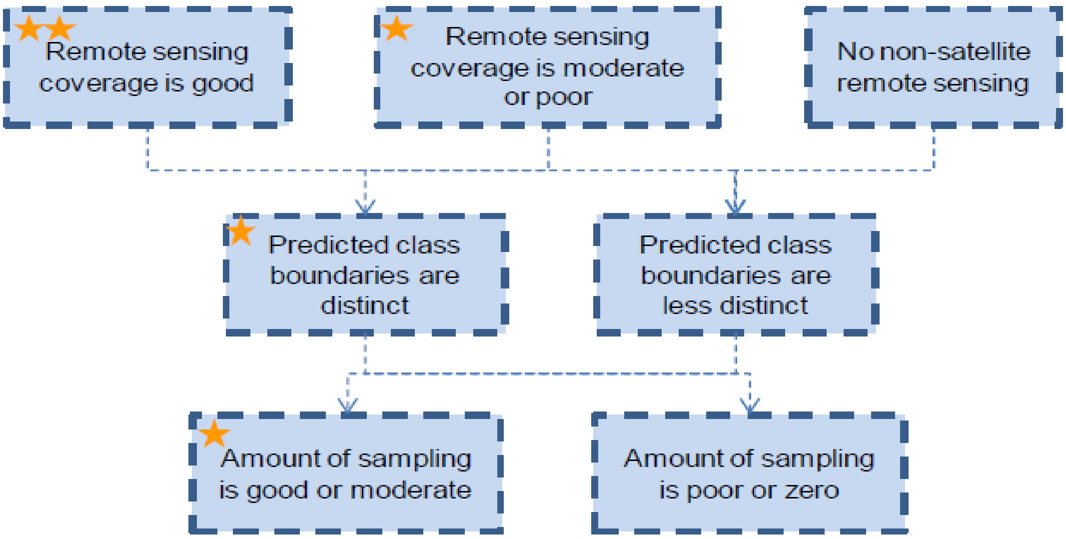


Figure 6 A three-step confidence assessment framework for EUNIS habitat classified seedbed maps (JNCC). The assessor starts at the top and follows the arrows. Routes through the decision tree are displayed as dashed lines to indicate that these are potential routes. Stars/points are awarded according to the answers given and the final score is the sum of the stars/points. Further description of the generic criteria for each step are outlined in Lillis, (2016).

3 Review and Assessment of the Provision of Ecosystem Services from Natural Capital in the SEMS

This section identifies the content of the non-monetary marine ecosystem account, given the SEMS boundary identified in Section 2. It considers, which ecosystem services to include (Section 3.1) the stocks that provide these services and how to measure their flows (Section 3.2). It then quantifies the capacity of biotopes to provide N and P removal ecosystem services (Sections 3.3 & 3.4) alongside regional estimates for anthropogenic N and P loadings that could potentially be removed by the biotopes (Section 3.5). The final section refines the baseline estimates by considering how regional condition assessments could be integrated into natural capital assessments (Section 3.6). A full discussion of the limitations and uncertainties regarding the structure and results to actually report these parts of the account are developed in Section 5.

3.1 Selection of Ecosystem Services

Having defined the boundaries of the SEMS and its natural capital stocks, the next step was to determine which ecosystem services from the marine environment are key to its value as natural capital, and therefore should be quantified in the account. Marine ecosystem services are classified in many different ways by different sources, with different treatment of services as intermediate and final services (e.g. Austen *et al.,* 2010; TEEB 2010; Haines-Young and Potschin, 2011; Luisetti *et al.,* 2014). This report adopts the list of services used in the UK NEAFO project Work Package 4 (Turner *et al.,* 2014; see Figure 7). This ecosystem service framework explicitly links ecosystem structure, processes and functioning to outcomes in the form of services which contribute to goods (benefits) that are consumed by humans (Figure 8). The intermediate service category has been developed to avoid double counting. For example, the nutrient cycling intermediate supporting service can be valued both in the context the amount of nutrients that can be recycled or immobilised (e.g. by a replacement cost method), but also in addition to the benefits of clean water (e.g. by a health and/or re- creational use metric) potentially overestimating the value of the resource.

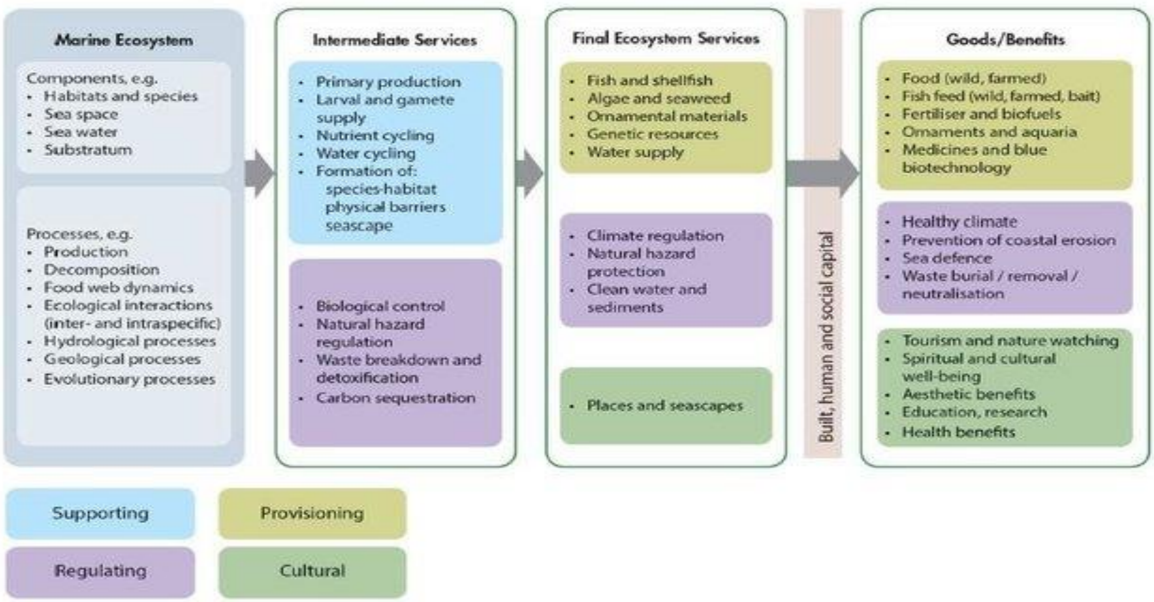


Figure 7 NEAFO framework (applied to coastal and marine ecosystem services from Turner *et al.,*2014)

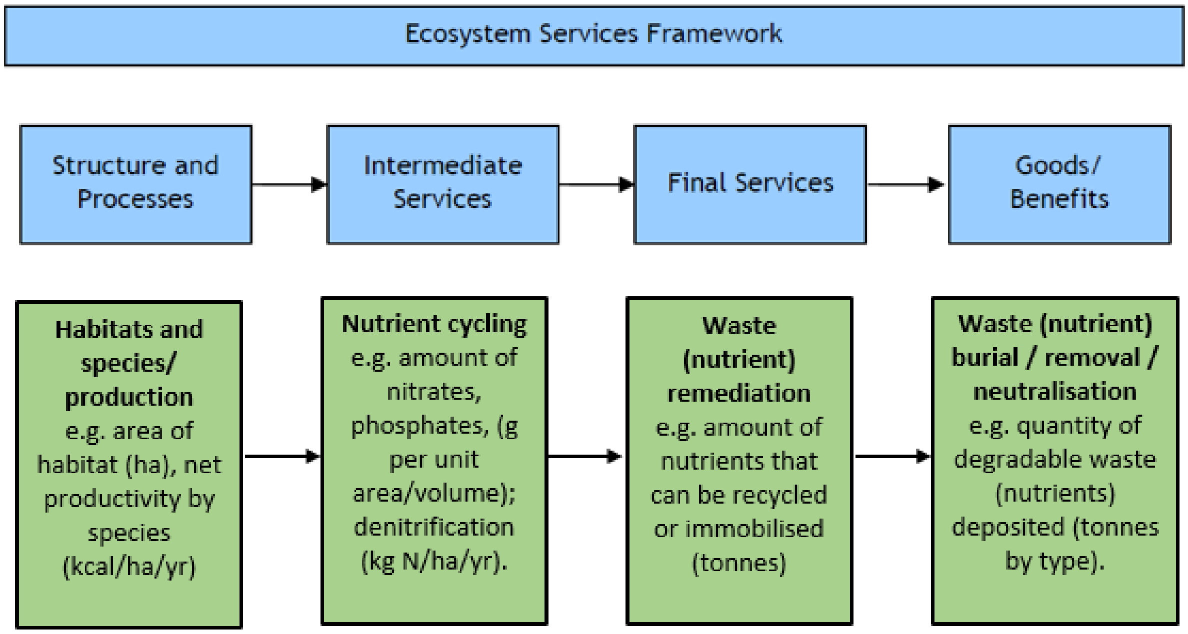


Figure 8 Ecosystem Service Framework (adapted from Turner *et al.,* 2014)

Although the ecosystem services of relevance for the water quality management in the SEMS cover many aspects from fisheries to tourism, the first part of the report aimed only to quantify the natural capital value of habitats and species in the context of N and P assimilation and flows. In Part B of this report the list of ecosystem services was further extended to include ancillary ecosystem services of habitats/species linked to water quality. This took into account clear definitions and practical frameworks to assess the water quality related ecosystem services (e.g. Keeler *et al.,* 2012; Grizzetti *et al.,*2016). This has resulted in a total of six ecosystem services for consideration within this project (for definitions and examples, see Table 3 in Appendix I):

1. Waste (nitrogen) remediation, detoxification and storage
2. Waste (phosphorus) remediation, detoxification and storage
3. Climate regulation (carbon sequestration and storage)
4. Commercial fish and shellfish harvesting
5. Recreation, tourism and leisure
6. Nursery function and supporting the existence of biodiversity

3.2 Accounting for Nutrient Flows

This section builds on the preceding sections' work to identify natural capital stocks, and looks at the approach to reporting the flow of marine ecosystem services relating to N and P remediation. The marine ecosystem account will be informed by annual flows (to estimate natural capital stock value).

Firstly, the extent (ha) of each biotope occurring within SEMS, including those within designated MPAs were calculated from the spatial habitat layer, in ARC GIS version 10.7 (Table 2). Of the entire spatial extent of biotopes within SEMS, 55.75% were initially contained within MPAs but this was updated to account for recent addition of the Solent and Dorset Coast SPA in January 2020. 100% of biotopes are now covered in the region, with the new legislation now covering much of the previously unprotected sub-tidal areas that were not previously encompassed in SPA's. Review of evidence on provision of waste (nutrient) remediation ecosystem services (intermediate services and ES) from biotope features in the SEMS is based on the scoring method of Potts *et al.,* (2014), which identified significant contribution from multiple biotopes within the SEMS (Table 2). In particular, there is high potential provision of nutrient cycling from coastal saltmarsh, reedbeds and seagrass with the confidence in the association greatest for saltmarsh and reedbeds.

Seven biotopes identified as having distinct contributions to nutrient fluxes reviewed using the matrix data provided in (Potts *et al.,* 2014) were then taken forward for further analysis. Littoral and sublittoral coarse sediment classifications (A2.11 and A5.1) were not analysed due to their limited potential to sequester nutrients (Table 2: 1 = Low contribution). Other EUNIS biotopes important to water quality and nutrient regulation (e.g. kelp beds, polychaete reefs, maerl beds, epiphyte and sponge communities) were recognised but excluded from further analysis owing to the fact they only comprised small areas in the SEMS (e.g. Kelp beds: 0.27% Table 2), were not present within the immediate SEMS boundary (e.g. maerl beds) or there was a lack of baseline data (e.g. polychaete reefs and sponge communities - many of these habitats require high resolution EUNIS level 5 data). Future efforts to include the full breadth of natural capital stocks available in a region would be important to allow the value of all biotopes to be considered in any future management decisions. EUNIS classifications were also aggregated for systems with similar or highly comparable mechanisms of nutrient exchange (e.g. A5.2/3 sublittoral mud and sand). The final list of biotopes analysed were:

- Littoral mud (A2.3)
- Littoral mud with macroalgal mats (A2.3)
- *Ostrea edulis* beds on shallow sublittoral muddy mixed sediment (A5.435)
- Reedbeds (C3.2, C32.1)
- Saltmarsh (A2.5)
- Seagrass (A2.61, A5.53, A5.545)
- Subtidal sediments (A5.2, A5.3, A5.34, A5.4)

Biophysical rates for N and P as a function of long-term burial in sediments and coupled nitrification-denitrification (hereafter denitrification) were then quantified from previous studies of temperate estuarine and coastal biotopes (Table 3). Nutrients can refer to dissolved inorganic nitrogen (DIN), dissolved, organic nitrogen (DON), particulate nitrogen (PN), dissolved inorganic, phosphorus (DIP), dissolved organic phosphorus (DOP) and particulate phosphorus (PP) in the SEMS context. For this analysis, we concentrate on DIN and DIP, as reduction of these nutrients is a primary focus of these mechanistic pathways. Literature between 1990 and 2020 on provision of ecosystem services from marine and coastal biotopes was reviewed to identify any updated evidence for supply of DIN and DIP (hereafter N and P) removal from the selected marine biotopes. Wider relevant studies from both peer-reviewed and grey literature sources were reviewed to support the matrix results. Net rates of nutrient removal ($g\ m^{-2}\ yr^{-1}$) are influenced by a number of factors including season (e.g. Westbrook *et al.,* 2019), local hydrology regimes (e.g. Ní Longphuirt *et al.,* 2016), nutrient loading rates (e.g. Smyth *et al.,* 2015) and the balance of population/biotope level processes (e.g. photosynthesis, respiration and dissolution) (e.g. Gilbertson *et al.,* 2012). Our study has used the most relevant and appropriate values reported in the scientific literature for calculating mean biophysical rates linked to N and P removal, while the median and range values calculated here also provide important information for decision makers on nutrient uptake variability. In addition, surprisingly no studies — to our knowledge — have examined the long-term storage capacity of native oyster beds (*O. edulis*) including properties such as denitrification and P burial. We, therefore, generated estimates of nutrient loss (denitrification) and sequestration (e.g. in shells and *via* burial) for this biotope using studies with *Crassostrea virginica* (Eastern American oyster). By combining the assembled remediation rates (Table 3) with data on the current area of the regions biotopes (Table S1), estimates of the relative quantities of N and P (tonnes yr^{-1}) removed by these biotopes were derived.

Table 2 Matrix assessment of provision of waste (nutrient) remediation services (UK NEA FO) from habitats in SEMS, including biotopes features of MPAs (building on Potts *et al.*,2014)

			Waste Remediation
			Regulating services
Natural Capital Stocks: Biotopes in the Solent Marine Site	Area (ha)	Area in MPAs (ha) pre-Jan 2020	Nutrient cycling
A2.11: Shingle (pebble) and gravel shores	708	708	1
A2.3: Littoral mud	6204	6204	3
A2.3: Littoral mud (with mat forming macroalgae)	1616	1616	3
A2.5: Coastal saltmarshes and saline reedbeds	1261	1111	3
A3: Infralittoral rock and other hard substrata	756	345	
A5: Sublittoral sediment	1438	215	1
A5.1: Sublittoral coarse sediment	9496	3914	1
A5.2: Sublittoral sand	10088	2817	3
A5.3: Sublittoral mud	7502	5516	3
A5.34: Infralittoral fine mud	400	344	3
A5.4: Sublittoral mixed sediments	1497	288	3
A5.435: *Ostrea edulis* beds on shallow sublittoral muddy mixed sediment	2839	1155	1
A5.52: Kelp and seaweed communities on sublittoral sediment	121	121	1
A5.53: Sublittoral seagrass beds and A2.61: Seagrass beds on littoral sediments	698	691	2
B1.21: Unvegetated sand beaches above the driftline	88	11	
B2: Coastal shingle	136	15	
B2.2: Unvegetated mobile shingle beaches above the driftline	251	28	
B3: Rock cliffs, ledges and shores, including the supralittoral	40	2	
C3.21: *Phragmites australis* beds	273	226	3
C2.3: Permanent non-tidal, smooth-flowing watercourses	25	5	

Scale of ecosystem service supplied relative to other features

- # Significant contribution
- # Moderate contribution
- # Low contribution
- # No or negligible ESP
- Not assessed

Confidence in evidence

- 3 UK-related, peer-reviewed literature
- 2 Grey or overseas literature
- 1 Expert opinion
- Not assessed

Table 3 Nitrogen and phosphorus removal rates used for biotope types occurring in the Solent showing mean, median ± range (min and max) reported values. Burial estimates represent a net sequestration of nutrients in the underlying sediment of the biotope. Negative values indicate net loss of the nutrient from the biotope. [1] References related to N denitrification, sequestration and storage, [2] References related to P sequestration and storage. * Native oyster estimates were made using the Eastern American oyster (*Crassostrea virginica*). Table from Watson *et al.,* (2020).

Biotope	Ecosystem Process/ function	Nitrogen[1] (g N m^{-2} yr^{-1})				Phosphorous[2] (g P m^{-2} yr^{-1})				References
		Mean	Median	Min	Max	Mean	Median	Min	Max	
Coastal saltmarshes	Burial	10.8	10.2	6.1	16.2	4.7	4.7	2.3	7	(Adams *et al.,* 2012[1,2])
	Denitrification	25.2	27.5	14.5	38.1	-	-	-	-	(Blackwell *et al.,* 2010[1])
Seagrass beds	Burial	4.9	3.9	2.7	8.0	-2.2	-4.3	-12.8	12.5	(Eyre *et al.*, 2016[1]; Holmer *et al.,* 2006[2])
	Denitrification	15.1	14.3	14.1	16.1	-	-	-	-	(Eyre *et al.*, 2016[1])
Reedbeds	Burial	5.8	3.2	1.8	12.4	7.5	7.6	1.9	12.8	(Kuusemets and Lõhmus, 2005[2]; Windham and Meyerson, 2003[1])
	Denitrification	5.8	4.4	2.6	10.5	-	-	-	-	Venterink *et al.,* (2003[1])
Littoral mud	Burial	9.2	9	7	11.4	-4.2	0.6	-19.3	1.3	(Adams *et al.*, 2012[1]; Burrows *et al.,* 2017[3]; Thornton *et al.,* 2007[2])
	Denitrification	6.9	4.3	3.4	12.9	-	-	-	-	(Eyre *et al.,* 2016[1])
Littoral mud (macroalgal mats)	Burial	33.3	24.6	4.7	78.2	31.4	29.7	4.3	64.4	(Palomo *et al.,* 2004[2]; Trimmer *et al.,* 2000[1,])
	Denitrification	0.6	0.4	0	1.7	-	-	-	-	(Trimmer et al., 2000[1])
Sublittoral sediment	Burial	3.1	3.6	1.6	4.2	-6.1	0.2	-28	2.9	(Eyre *et al.,* 2016[1]; Thornton *et al.,* 2007[2])
	Denitrification	2.8	3.0	2.5	3.1	-	-	-	-	(Eyre *et al.,* 2016[1])
Native oyster (*Ostrea edulis*) beds*	Assimilation in tissues and shell (g/individual)	0.18	0.14	0.02	0.4	0.1	0.09	0.003	0.4	(Higgins *et al.,* 2011[1,2])
	Burial	2.1	0.6	0	7.8	2.3	0.7	0	8.4	(Newell *et al.,* 2005[1,2])
	Denitrification	16.4	3.7	2.7	55.6	-	-	-	-	(Kellogg *et al.,* 2014[1])

3.3 Total Waste (Nutrient) Remediation Flows

A summary of the total amount of nutrient removed (tonnes yr^{-1}) by each biotope in the SEMS are given in Figure 9. Across the biotopes the net effect on N & P removal varied substantially and in the case of P sometimes included both positive and negative values even within the same biotope. Of the 14 biophysical rates measured here, 11 mean values were positive and 13 median values were positive, generally indicating enhancement in nutrient removal rates.

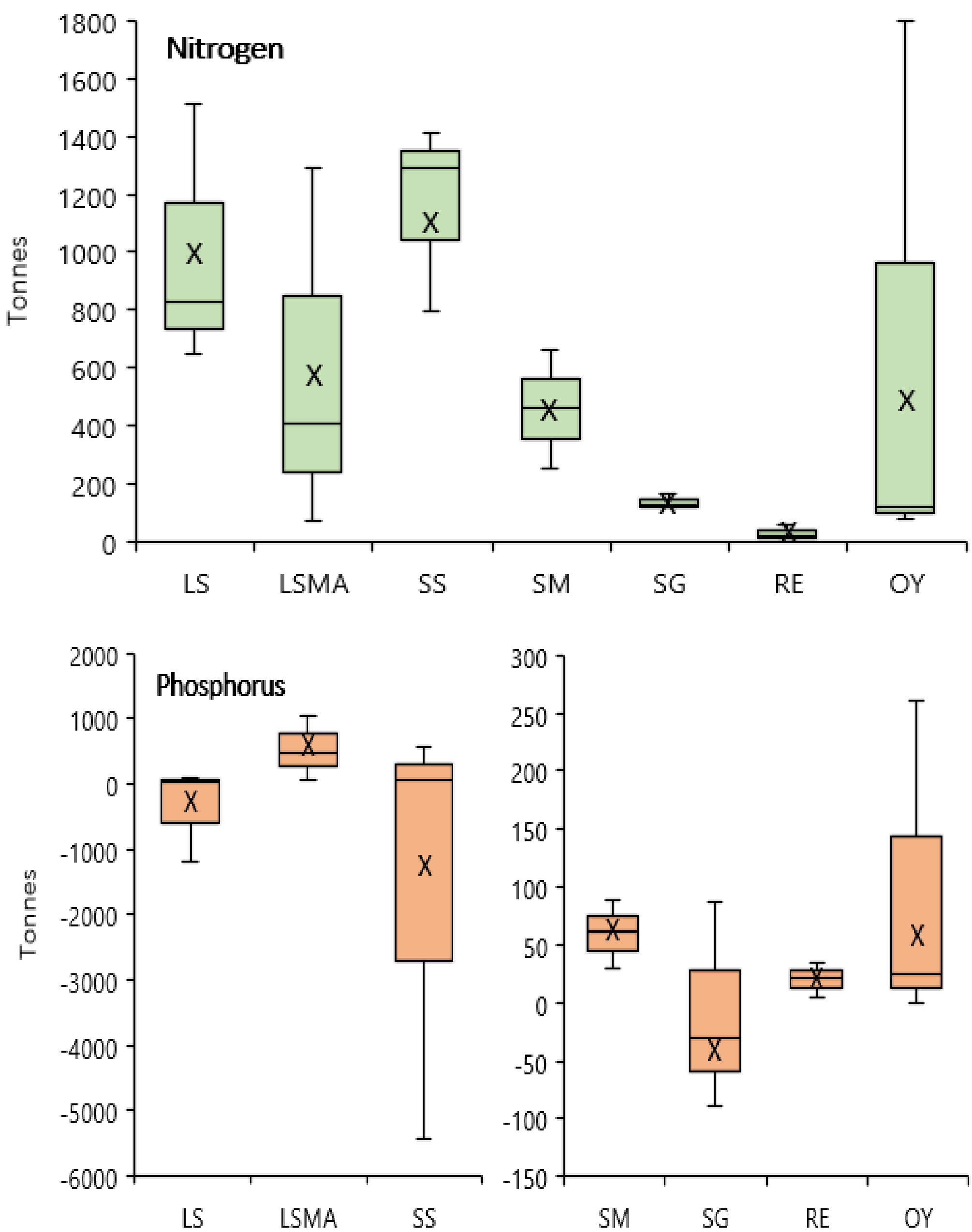


Figure 9 Nitrogen and phosphorous removal by the SEMS coastal and subtidal benthic biotopes. Figures showing the lower (Q1) and upper (Q3) quartiles, median (-), mean (X) and range (error bars). LS (Littoral sediment), LSMA (Littoral sediment with macroalgae), SS (Subtidal sediment), SM (Saltmarsh), SG (Seagrass), RE (Reedbeds), OY (Native oysters). Figure from Watson *et al.,* (2020)

Negative mean and median biophysical values were recorded for P burial by seagrass and sublittoral sediments with the latter also exhibiting the greatest range of biophysical rates of all the biotopes with mean negative P burial values recorded in excess of -1000 tonnes yr^{-1}. P removal by littoral sediments were also negative when considering mean burial rates but slightly positive when considering the median rates. A combined negative efflux of -1455 P tonnes yr^{-1} was calculated for littoral and subtidal sediments using the mean biophysical values. These results are in line with other studies that suggest estuaries are generally net heterotrophic and therefore their sediments often represent a net source of P to the ocean (Deborde *et al.,* 2007) lowering the total potential value of this ecosystem service.

Littoral sediments were the highest contributing benthic biotopes for N removal when considered together with macroalgae-dominated sediments (1546 tonnes yr^{-1} based on the mean), due to a combination of their area (13.66 % Table 2) and the high burial rates of the macroalgae (33.3 g N m^{-2} yr^{-1}; Table 3). However, subtidal sediments also made a substantial contribution to N removal (1150 tonnes yr^{-1}, based on the mean) primarily due to their large area (43% Table 2). Saltmarsh, seagrass and reedbed biotopes were also highly productive in removing N, but their contribution to the total nutrient budget was often small because of the limited combined extent of these biotopes (4.9% Table 2). In general, native oysters enhanced N removal rates, but the biophysical values varied by up to two orders of magnitude. This is because the reported denitrification rates (2.7-55.6 g N m^{-2} yr^{-1}; Table 3) will change with season, geographic location and oyster densities (Kellogg *et al.,* 2014).

The highest P burial totals occurred in littoral sediments overlain with macroalgae (480 tonnes yr^{-1} base on mean rates). Saltmarsh and reedbed biotopes were recognised to have high P burial rates per m^2 (especially reedbeds: 7.5-7.6 g m^{-2} yr^{-1} mean/median; Table 3) but, lagged behind macroalgal sediment biotopes in terms of total P burial. Saltmarsh biotopes in the region removed approximately three times more P (~60 tonnes yr^{-1}, based on the median) than reedbed biotopes (~20 tonnes yr^{-1}, based on the median). Oysters were also a net sink for P, removing comparable amounts to saltmarsh (~58 tonnes yr^{-1}) when estimated using mean biophysical rates.

By combining the mean values for all the biotopes, we estimate that 3831 tonnes N yr^{-1}, are currently removed by existing biotopes in the SEMS region. In contrast, 813 P tonnes yr^{-1} P are released by biotopes, representing a net P source to the water column. The total median rates for the SEMS were lower for N, removing 567 fewer tonnes of each nutrient per yr^{-1} (total 3248 tonnes N yr^{-1}), while P burial was 1452 tonnes per yr^{-1} greater representing a region-wide positive sink of P (639 tonnes P yr^{-1}). These considerable differences in relative biotope contributions to nutrient removal illustrate how ratios of functionality (e.g. mean vs median rates) for different biotopes dictate potential gains (or losses) in ecosystem service production depending on how they are analysed. We, therefore, strongly recommend that users of this data explicitly state the calculation method used; but also acknowledge the variability for different biotopes because of experimental methods, species choice and the influence of local environmental factors. Given that the median values assessed here are more likely to be conservative estimates and less influenced by outliers and skewed data we therefore, use these estimates for subsequent analysis.

3.4 Regional Waste (Nutrient) Remediation Flows

When considered at the level of individual catchments, littoral sediments were the most important biotope for N removal *via* net burial and denitrification in Southampton Water and Chichester Harbour, accounting for 33% and 45.6% of each region's total N remediation potential (Table 4). Littoral sediments were also the second largest removers of N in eight other catchments including: Lymington Estuary, Beaulieu Estuary, Portsmouth Harbour, Langstone Harbour, Pagham Harbour, Yar Estuary, Bembridge Harbour and the Solent open water region. Littoral sediments overlain with macroalgae were likewise highly important in removing N in the Hamble Estuary, Portsmouth Harbour, Langstone Harbour, Medina Estuary and Bembridge Harbour representing between 31-41% of each regions N removal potential (Table 4). In contrast, in the open water portion of the Solent the largest rates of N removal *via* burial and denitrification (74%) occurred in the subtidal sediments due to the large extent of this biotope (84%) reflecting the geomorphology of the system.

Of the angiosperm biotopes, saltmarsh communities were the largest contributing biotope for N remediation in several of the smaller estuaries in the SEMS including Lymington Estuary, Beaulieu Estuary, Pagham Harbour, Yar Estuary and Newton Harbour (Table 4) removing between 16 - 47 tonnes of N yr^{-1} (representing 43-67% of each regions N removal potential). They were also the second largest removers of N in Lymington Estuary, Southampton Water and the Hamble Estuary. Seagrass and reedbed biotopes generally removed less N than saltmarsh biotopes with the exception of the Medina estuary where seagrass was the second largest biotope in terms N remediation, removing 20% of this regions N (Table 4). After sediment biotopes, seagrass biotopes were an important biotope in the open water region of the Solent, representing 50% of the seagrass area in the SEMS region, and removing the equivalent of all the other regions seagrass combined (~63 tonnes N yr^{-1}). Similarly, native oyster biotopes were largely congregated within the open water region of the Solent (59% of the SEMS population area) and were responsible for removing ~46 tonnes N yr^{-1} (Table 4) which is comparable with the amounts removed by saltmarsh in the same area. In Portsmouth, Langstone and Chichester Harbours native oyster biotopes were also comparable with seagrass biotopes' ability to remove N, potentially removing 12-18 tonnes N yr^{-1}.

P sequestration *via* burial in macroalgal mat and saltmarsh sediments was the largest and second largest output term in the P budget of ten of the thirteen systems (Table 5) including: Lymington Estuary, Beaulieu Estuary, Southampton Water, Portsmouth Harbour, Chichester Harbour, Pagham Harbour, Yar Estuary, Newton Harbour, Newton Harbour, Medina Estuary and Bembridge Harbour. Exceptions to this trend were found in the Hamble Estuary, Langstone Harbour and the Solent open water region where reedbed, littoral sediments and subtidal sediments respectively sequestered more P than saltmarsh communities (Table 5). The high efficiency of P removal in macroalgal mats most likely represents these algae forming thick canopies, which act as natural physical barriers, reducing P (and N) fluxes from the sediment to the overlying water (Palomo *et al.,* 2004) while at the same time enhancing the formation of non-soluble P compounds, mainly as calcium-bound phosphorus, since great amounts of calcium are available in estuarine sediment pore-waters. In contrast, few studies have investigated temperate seagrass meadow sedimentary P pools, with those that have generally suggesting very low biological uptake rates of inorganic P (e.g. Holmer *et al.,* 2006; Nayar, 2015). Based on the median seagrass burial rates used here, we estimate a negative efflux of P from this biotope (~ 30 tonnes yr^{-1}) probably due to a higher mineralisation of organic-matter inputs. It is important to note that the values calculated here are *average annual* values and if we had used the higher range biophysical values to represent N or P burial or considered seasonal patterns then the values such as of seagrass burial may be higher or positive. For example, some studies (e.g.

Touchette and Burkholder, 2000) have shown seagrass to be a net sink of P in "pristine" sites and are more likely to permanently remove P in winter than in summer.

Table 4: Nitrogen removal potential (tonnes yr^{-1}) of biotopes in the SEMS. Green cells represent the highest biotope removal potential and yellow cells represent the second largest removal potential.

Nitrogen (Median)	Littoral sediments	Littoral sediments (with macro)	Sublittoral sediments	Saltmarsh	Seagrass	Reedbeds	Native oyster	Region total
Lymington Estuary	23.52	10.51	5.44	31.63	0	1.09	0	**72**
Beaulieu Estuary	20.20	5.25	15.91	36.91	0	1.60	0	**80**
Southampton Water	95.32	8.71	85.09	86.86	0	0.86	15.28	**292**
Hamble Estuary	5.69	17.56	6.40	12.56	0	4.14	9.66	**56**
Portsmouth Harbour	78.54	84.32	40.58	27.12	15.63	0.10	12.08	**258**
Langstone Harbour	163.87	91.07	23.07	23.35	18.90	1.41	21.26	**343**
Chichester Harbour	218.09	117.59	42.10	114.49	21.08	1.60	18.84	**534**
Pagham Harbour	16.61	3.50	2.52	47.45	0.91	0.70	0	**72**
Yar Estuary	3.59	3.75	0	16.19	1.82	1.28	0	**27**
Newton Harbour	12.49	19.27	2.39	25.99	0.00	0.01	0	**60**
Medina Estuary	2.79	8.51	3.71	4.14	4.91	0.04	0	**24**
Bembridge Harbour	3.19	2.25	0.80	1.51	0.36	0.03	0	**8**
Solent (open water)	182.60	30.77	1063.92	46.32	63.23	4.61	45.81	**1437**
Biotope total	**827**	**403**	**1292**	**475**	**127**	**17**	**123**	**3263**

Table 5: Phosphorous removal potential (tonnes yr^{-1}) of biotopes in the SEMS. (-) value indicates a net release of phosphorous. Green cells represent the highest biotope removal potential and yellow cells represent the second largest removal potential.

Phosphorous (Median)	Littoral sediments	Littoral sediments (with macro)	Sublittoral sediments	Saltmarsh	Seagrass	Reedbeds	Native oyster	Region total
Lymington Estuary	0.97	12.49	0.19	4.11	0	1.29	0	**19**
Beaulieu Estuary	0.84	6.24	0.57	4.80	0	1.9	0	**14**
Southampton Water	3.94	10.35	3.08	11.30	0	1.01	3.1	**33**
Hamble Estuary	0.24	20.87	0.23	1.63	0	4.91	2.01	**30**
Portsmouth Harbour	3.25	100.21	1.46	3.52	-3.73	0.11	2.64	**107**
Langstone Harbour	6.78	108.24	0.83	3.03	-4.51	1.67	4.27	**120**
Chichester Harbour	9.03	139.75	1.52	14.89	-5.03	1.9	3.43	**166**
Pagham Harbour	0.69	4.16	0.09	6.17	-0.21	0.83	0	**12**
Yar Estuary	0.15	4.46	0	2.10	-0.43	1.52	0	**8**
Newton Harbour	0.52	22.90	0.08	3.38	0	0.006	0	**27**
Medina Estuary	0.12	10.11	0.13	0.53	-1.17	0.05	0	**10**
Bembridge Harbour	0.13	2.68	0.02	0.19	-0.08	0.04	0	**3**
Solent (open water)	7.56	36.57	38.51	6.02	-15.1	5.47	8.55	**88**
Biotope total	**34**	**479**	**47**	**62**	**-30**	**21**	**24**	**636**

3.5 Nutrient Loads in the SEMS

While the ecosystem scale budgets calculated above are a useful tool for expanding our knowledge about potential sinks of N and P in coastal systems, it is also important to understand sources (loads) of N and P if we are to manage eutrophication in coastal waters. Assessing nutrient loads however can be challenging with figures varying whether they represent data monitored through field surveys or computed using catchment modelled predictions. Here, we refer to EA modelling output undertaken for the Partnership for Urban South Hampshire (PUSH) for a basin-level analysis. We used modelled load data because monitored load data rarely covered the entire study area. Table 6 below details the modelled average nutrient loads in each WFD region and their proportion relative to the whole SEMS. In the Solent, pollution from nutrients, mainly comes from agriculture (50%) and coastal background sources (40%) with only 10% estimated to originate from urban discharges. Each year, it is estimated that about 5016 tonnes of N (in the form of DIN) and 602 tonnes of P (in the form of DIP) enters the SEMS; with the largest loadings entering the highly urbanised Southampton Water catchment (approximately 30% and 40% of total N and P loads respectively).

Table 6 Modelled annual average nutrient loads for each SEMS region (adapted from PUSH 2019 & EA 2013-2016 macroalgal modelling data).

Region	Total N loading (kg yr^{-1})	Relative proportion N (%)	Total P Loading (Kg yr^{-1})	Relative proportion P (%)
Lymington Estuary	145,030	2.89	2,370	0.39
Beaulieu Estuary	121,280	2.42	1,140	0.19
Southampton Water	1,520,106	30.30	244,870	40.65
Hamble Estuary	198,128	3.95	19,270	3.20
Portsmouth Harbour	800,249	15.95	76,028	12.62
Langstone Harbour	370,749	7.39	31,276	5.19
Chichester Harbour	1,275,378	25.42	185,089	30.73
Pagham Harbour	225,702	4.50	23,171	3.85
Yar Estuary	37,950	0.76	2,279	0.38
Newton Harbour	133,247	2.66	9,597	1.59
Medina Estuary	117,635	2.34	3,656	0.61
Bembridge Harbour	71,017	1.42	3,588	0.60
Total	**5,016,471**		**602,334**	

To investigate the potential for different biotopes to impact regional-scale water quality, we then combined the previously calculated annual N and P removal rates with the above estimates on nutrient loading (Table 6) to calculate the percentage of annual external N and P inputs into the SEMS that are currently remediated by natural processes. Budget calculations show that on an annual basis, estuarine processes retain and remove 35% of total N and 91% of total P that would otherwise pass into the Solent channel that separates the Isle of Wight from the mainland of England. These results suggest most of the estuarine P is sequestered and stored in the biota of the estuary, especially macroalgal mat and saltmarsh biotopes (Table 5). Net removal estimates after considering the biotope removal potential of the Solent channel suggest the remaining P (53.7 tonnes yr^{-1} or 9% of total loadings) is sequestered here, while the fate of N is less clear, with an annual excess of approximately 1753 tonnes N yr^{-1} (40% of total loadings) unaccounted for. Unmeasured sources are most likely exported to the deep ocean, re-introduced to estuaries as a result of influx from hydrological process from the ocean or are recycled *via* pelagic N-fixation. The nutrient loading above mainly calculates levels of N and P in terms of inorganic N, but dissolved organic N has been found to comprise 7-13 % of the potential biologically available N in rivers entering the SEMS. Thus, we therefore advise that this uncertainty is recognised and a recommended 10% precautionary buffer approach is adopted when considering the nutrient loading data.

Regional estimates for the proportion of the N load that could potentially be removed by biotopes varied widely between the catchments (Figure 10), ranging from 11% in Bembridge Harbour to 92% in the case of Langstone Harbour. The comparatively high remediation capacity of habitats in Langstone harbour is a result of diversions of nutrient inputs away from the harbour in recent years. For example, improvements to the Southern Water sewage treatment plant at Budds Farm at the northern end of the harbour, including the construction of a 5.7km sea outfall under the harbour which takes waste water from the pumping station at Eastney out to sea, have resulted in large reductions in N loads entering the Harbour. However, our analysis does highlight that a relatively large proportion of N removed in Langstone Harbour (~25%) is still achieved *via* the growth of green macroalgal mats which are often associated with detrimental effects on infaunal species and wading bird populations (Thornton 2016).

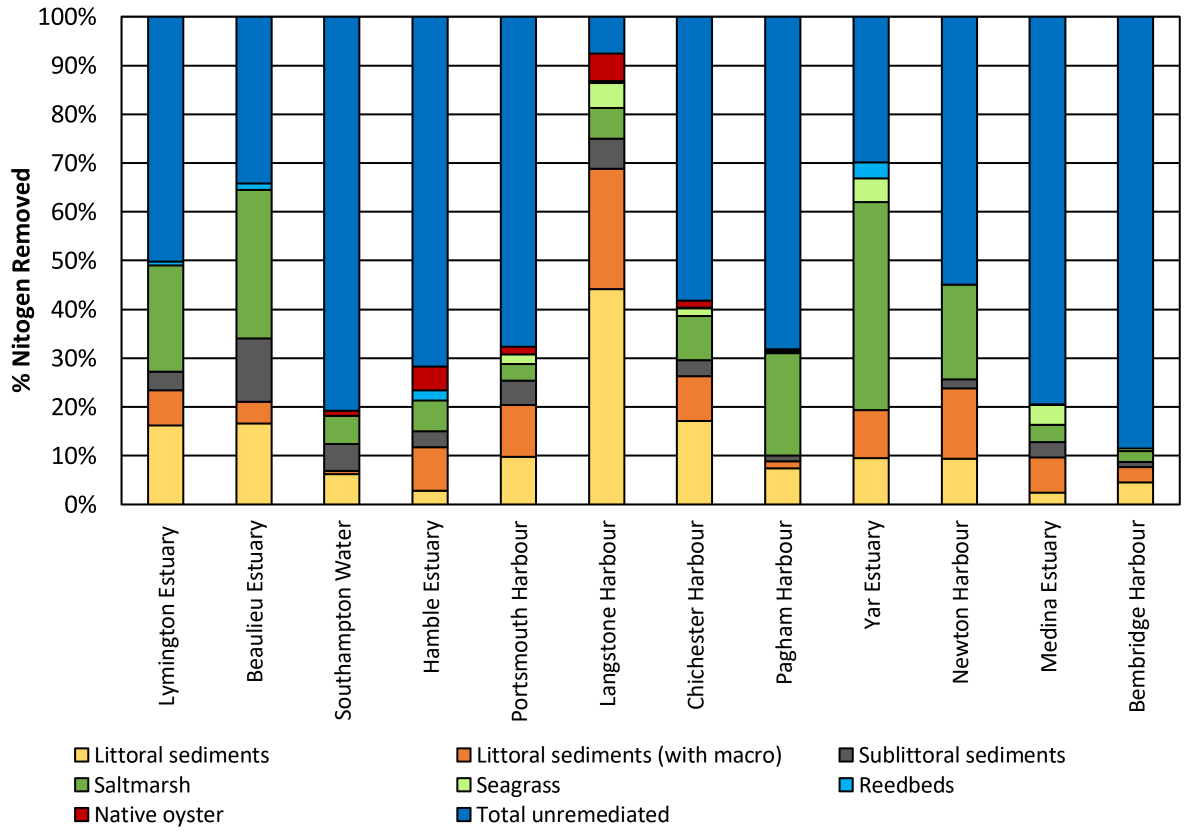


Figure 10 Percentage of N remediated by biotopes in the SEMS vs catchment N loading

Estimates of P removal from the land-margin showed that net burial in sediments by all considered biotopes appears to remove more P from every region of the SEMS (i.e. 100%: see appendix Table S4 for full values) except; Southampton Water (13% removed), Chichester Harbour (89% removed) and Pagham Harbour (51% removed). Several studies have also found that the percentage of P entering estuaries and that is ‘‘permanently’’ buried in sediments is quite variable (Grelowski *et al.,* 2000; Van Beusekom and De Jonge, 1998), ranging from 10% or less up to more than 100% (in the case of estuaries which are trapping large amounts of P from offshore), with the percentage accumulation of

P in sediments often decreasing as an estuary becomes more eutrophic. The greater uptake of P, than of N, could also have resulted from a faster turnover time of P than N, with studies carried out in Chesapeake Bay (Malone *et al.,* 1996) indicating that dissolved DIP turnover times tend to be short (2 h, independently of the season), while dissolved DIN turnover times were longer (1– 500 h during spring, and 1–10 h during summer). In summary, despite the narrow focus on only a handful of biotopes in the Solent's coastal waters, these results serve as a useful baseline for the current removal capacity of biotopes in the SEMS region as well as a framework for considering the potential use of any habitat or species to ameliorate eutrophic conditions.

3.6 Factoring in Condition to Natural Capital Assessments

Estimates of natural capital should clearly aim to be as comprehensive as possible, though recognizing that a staged approach may sometimes be necessary. Until now we have not tried to quantify the local ecological ***condition*** of the natural capital's stocks, but instead have assumed that all biotopes in the SEMS are of a similar condition (e.g. moderate; based on mean/median estimates) and contribute to the provisioning of waste (nutrient) remediation purely on the basis of their ***extent*** in the region. However, like other assets, natural capital stocks are susceptible to changes in quality over time. The procedures set out above should in principle deal with this to the degree that our generalised ecosystem assessment (i.e. the mean, median and range of biophysical values) can be adjusted to record a worsened condition, this results in an estimated reduced flow of ecosystem services. In turn, as the net present value of current and future services, the estimated stock value will be lowered accordingly (If the condition accounts showed an improvement, the reverse would be the case). This procedure is in line with UK (ONS & Defra, 2017) and international (SEEA, 2018) guidance (Figure 10) and implicitly involves local qualitative adjustments of the initial natural capital extent stock.

Figure 10 The main components of natural capital accounts (ONS & Defra, 2017)

To represent the local condition of biotopes in the SEMS, we used the 2016 WFD (cycle 2) waterbody and Food Standards Agency (FSA) summary condition assessments, together with their qualifying sub-feature assessments, for each of the regions in the SEMS (Table 7). WFD classification data for benthic invertebrates in transitional and coastal waters were used as a proxy to represent littoral sediment and subtidal sediment condition. Similarly, angiosperm biological elements were used to represent saltmarsh and seagrass communities, while the ecological status of macroalgal mats was defined using the WFD macroalgae classification. These data were then compared with more recent condition assessments of the Solent made by Natural England (NE; Table 7) to gauge differences between different assessment classifications. In the case of reedbeds this approach was particularly important, as terrestrial wetland habitats, reedbeds fall under the WFD primarily as a feature of water-dependent Protected Areas, but are often not formally monitored as part of the surface water monitoring programmes (WFD CIS Guidance Document No 12). As this information was not available for the SEMS, we used the NE condition assessment as a proxy for reedbed biotope condition.

Table 7 Relative assessment of ecological condition across the SEMS. Based on WFD status data, Natural England condition assessment data and Food Standards Agency (FSA) protocol for classification of shellfish production areas, England and Wales. WFD is assigned on a scale of High, Good, Moderate, Poor or Fail. *The Hamble Estuary is not a WFD waterbody, therefore the values used here reflect its status within the larger Southampton Water complex. **Some areas of these sites are in favourable condition but this is borderline and these areas are at high risk. (N/A) no data available, (-) biotope not present.

Assessment Unit	Littoral sediments with macroalgal mats		Invertebrates (littoral and sublittoral sediments)		Angiosperms (saltmarsh and seagrass)		Reedbeds		Native oysters
Assessment Classification	**WFD (2016)**	**NE 2018**	**WFD (2016)**	**NE 2018**	**WFD (2016)**	**NE 2018**	**WFD (2016)**	**NE 2018**	**WFD (2016) /FSA**
Lymington Estuary	Good	(N/A)	High	Unfavourable No Change	(N/A)	Unfavourable: (Unknown Condition)	(N/A)	Unfavourable - Recovering	-
Beaulieu Estuary	Good	(N/A)	Good	Unfavourable No Change	(N/A)	Unfavourable: (Unknown Condition)	(N/A)	Favourable	(N/A)
Southampton Water	Good	**Unfavourable no change	Good	Unfavourable No Change	Good	Unfavourable: (Unknown Condition)	(N/A)	Unfavourable - Recovering	(Fail) Prohibited
Hamble Estuary*	Good	**Unfavourable no change	Good	Unfavourable No Change	Good	Unfavourable: (Unknown Condition)	(N/A)	Unfavourable - Recovering	(Fail) Prohibited
Portsmouth Harbour	Moderate	**Unfavourable no change	High	Unfavourable No Change	Moderate	Unfavourable: (Unknown Condition)	(N/A)	(N/A)	(Fail) Prohibited
Langstone Harbour	Good	Unfavourable recovering	Good	Unfavourable No Change	(N/A)	Unfavourable: (Unknown Condition)	(N/A)	(N/A)	(Fail) Prohibited
Chichester Harbour	Moderate	**(Unfavourable no change	Moderate	Unfavourable No Change	(N/A)	Unfavourable: (Unknown Condition)	(N/A)	Unfavourable Recovering	(Fail) Prohibited
Pagham Harbour	Good	(N/A)	Good	Unfavourable No Change	Moderate	Unfavourable: (Unknown Condition)	(N/A)	(N/A)	-
Yar Estuary	Moderate	(N/A)	Moderate	Unfavourable No Change	(N/A)	Unfavourable: (Unknown Condition)	(N/A)	(N/A)	-
Newton Harbour	Moderate	(N/A)	Good	Unfavourable No Change	(N/A)	Unfavourable: (Unknown Condition)	(N/A)	(N/A)	-
Medina Estuary	Moderate	(N/A)	Moderate	Unfavourable No Change	(N/A)	Unfavourable: (Unknown Condition)	(N/A)	(N/A)	(N/A)
Bembridge Harbour	Moderate	(N/A)	High	Unfavourable No Change	(N/A)	Unfavourable: (Unknown Condition)	(N/A)	(N/A)	-
Solent (open water)	Good	(N/A)	Good	Unfavourable No Change	Moderate	Unfavourable: (Unknown Condition)	(N/A)	(N/A)	(Fail) Prohibited

Likewise, *in lieu* of site-specific condition assessment data for native oyster populations we used regulations relating to the commercial production and sale of live bivalve molluscs from classified production areas. These regulations are implemented by the EA to measure WFD compliance by means of the Food Safety and Hygiene (England) Regulations 2013. Data on shellfish waters in England and Wales are compiled by CEFAS using the results of monthly bacteriological sampling. Production areas are then classified by the Food Standards Agency (FSA) according to the *Escherichia coli* levels present in the samples. This classification is then broken-down into one of five categories that then determines the areas where bivalves can be harvested from and how they have to be treated post harvesting to ensure they are safe for human consumption. Although only a relative measure of ecological condition, this classification structure has recently been used by local fisheries and water quality experts at the EA and SIFCA to assess water quality impacts on ecosystem service provision by bivalves in the SEMS (Williams & Davies, 2018a & Williams *et al.,* 2018).

A summary of the different classifications used in this analysis to represent condition are given in (Table 7). Here we have grouped each of the three classifications mentioned above, in a qualitative manner whereby condition determines the level of ecosystem service that the biotope is capable of providing. This assumption based on recent scientific understanding that the relationship between ecological condition and regulating ecosystem services is generally linear in nature and are expected to have a positive relation with the ecological status (see Figure 11; Grizzetti *et al.,* 2019). For example, in the case of native oysters there is evidence that better water quality conditions, leading to improved reproduction and survival of bivalve shellfish, will in turn increase the nutrient removal capacity of these biotopes (e.g. Kellogg *et al.,* 2014).

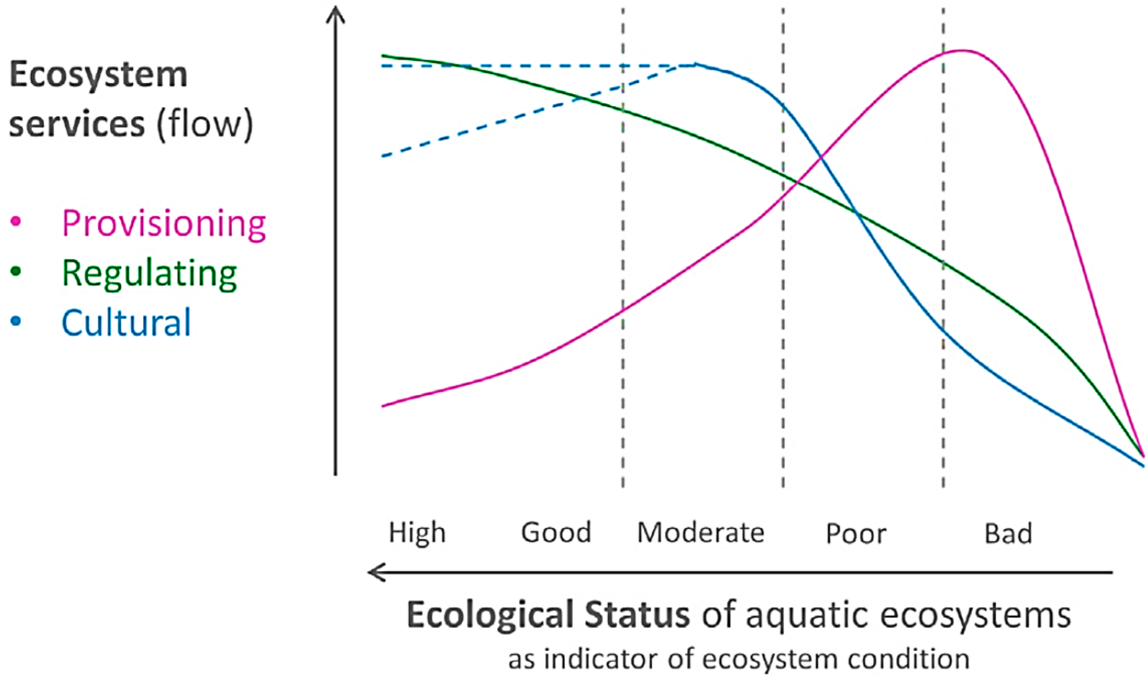


Figure 11 Expected relationship between the level of ecosystem services (flow) and ecological status in aquatic ecosystems. From Grizzetti *et al.,* (2019).

To facilitate a 5 point scoring approach, we extended our biophysical estimates to include the interquartile range (IQR), specifically adding biotope removal values for the upper (Q3) and lower (Q1) quartiles (i.e. the 75th and 25th percentiles; see Appendix Table S5 &6). Using this framework, the level of provision made by each respective biotope was adjusted in a hieratical process based on WFD condition data, but then if data were absent for that biotope, we reverted to NE condition data. In the case of some angiosperm biotopes where there was no WFD regional data and the NE condition data was: Unfavourable: (Unknown Condition) it is assumed that they are providing an arithmetic median (Q2) level of ecosystem service (this is an attempt not to overvalue the contributions of these biotopes; although this itself may also be an overestimate as there is some evidence of declines of these biotopes in several areas of the Solent). Additionally, the level of ecosystem service provided by biotopes will not all be of equal importance based on the criteria for condition assessments. In the case of macroalgal mats, movement towards achieving good or high status for this biotope indicates a reduction in the algal mat biomass (Table 8) and therefore a reduction in its nutrient removal potential (and vice versa as the condition assessment moves to poor or bad the algal mat biomass will increase, thereby increasing its nutrient removal potential). The separation and reversal of macroalgal mats ecosystem service provisioning vs condition in Table 8, is therefore an artefact of how its Ecological Quality Ratio is determined using WDF assessment measures.

Table 8 Level of nutrient removal ecosystem service from biotopes in various condition states

WFD ecological status	Bad	Poor	Moderate	Good	High
NE condition assessment	Partially Destroyed	Unfavourable Declining	Unfavourable No Change	Unfavourable Recovering	Favourable
FSA native oyster	Prohibited	Class C	Class B	Long Term Class B	Class A
Level of nutrient remediation ecosystem service (all other biotopes)	Minimum	Q1	Median (Q2)	Q3	Max
Level of nutrient remediation ecosystem service (macroalgal mats)	Maximum	Q3	Median (Q2)	Q1	Minimum

The level of baseline (median) ecosystem service that would be expected if all biotopes were adjusted based on regional condition classification data is presented in Table 9. These levels are placed in the context of the overall baseline levels (extent only) and two hypothetical scenarios whereby if it was assumed that all biotopes transitioned to "Good" or "High" status reflecting potential future improvements in biotope condition. Overall, we estimate there would be a 10% (328 tonnes yr^{-1}) increase in N removal and a 26% increase in P removal (167 tonnes yr^{-1}) if condition is factored into the baseline calculations. Similarly, we would expect a 25% to 59% increase in N removal and a 14% to 80% increase in P removal, if biotopes were to transition to "Good" or "High" status respectively (Table 9).

Table 9 Summary of the nutrient removal capacity of the SEMS based on various estimates of condition

	Baseline (Extent Only) (Tonnes yr^{-1})	Baseline (with Extent and Condition) (Tonnes yr^{-1})	All Good Condition (Tonnes yr^{-1})	All High Condition (Tonnes yr^{-1})
Nitrogen	3263	3590	4487	5712
Phosphorus	636	811	915	1451
Total	**3899**	**4474**	**5402**	**7162**

Regional adjustments based on condition are shown in Tables 10 and 11. Relative to the baseline there were improvements in N removal across all regions with the exception of the Hamble estuary, Medina estuary and Newton Harbour which remained similar to the baseline estimates (Table 10). With the decline in the contribution of macroalgal mats to N removal (based on improved "Good" condition in several regions e.g. Lymington, Beaulieu, Langstone etc. see Table 7), littoral sediment becomes the largest contributing biotope for N removal in many regions including: Lymington Estuary, Beaulieu Estuary, Portsmouth Harbour, Langstone Harbour and Bembridge Harbour. However, macroalgal mats remain a large contributor to the potential N removal budget, particularly in the Portsmouth and Chichester Harbours. Saltmarsh communities remain the largest contributor for N removal in many of the smaller estuaries including; Pagham Harbour, the Yar Estuary and Newton Harbour, but this now includes the Hamble Estuary (Table 10). There are also large declines in the contribution of native oysters to both N and P budgets reflecting the adjustment from the baseline "moderate" state of this biotope to "bad".

Overall, our results suggest that factoring in the relative local condition of the Solent's biotopes would generally improve the total baseline N and P removal estimates, but at a regional level there would predominantly be reductions in P removal, owing to reduced macroalgal mat biomass and downgraded native oyster biotope condition. It is important to note that the measurement of ecosystem condition following the concepts in an ecosystem accounting model is a complex and subjective task due to the need to consider multiple ecosystems and multiple characteristics. For instance, if we had purely used the Natural England condition assessments (rather than in combination with the WDF assessments), we would get results much closer to our initial baseline assessment values. Therefore, pending further testing of different condition indicators, we suggest that over time the non-monetary accounts developed here can be broadened in scope and filled with a larger range of condition indicators. For completeness, an updated version of the regional N and P loading estimates vs that which could potentially be removed by biotopes adjusted by condition is given in Appendix Table S7 and Figure S2.

Table 10: Nitrogen removal potential (tonnes yr^{-1}) of biotopes after factoring in condition. Green cells represent the highest biotope removal potential and yellow cells represent the second largest removal potential. (+) represents an improvement in the region's nutrient removal potential relative to the baseline while a (-) represents a reduction in a region's nutrient removal potential. (=) represents no change in the region's nutrient removal potential. (ES) Ecosystem service.

Nitrogen (Median)	Littoral sediments	Littoral sediments (with macro)	Sublittoral sediments	Saltmarsh	Seagrass	Reedbeds	Native oyster	Region total	Improvement in ES relative to baseline
Lymington Estuary	42.96	6.23	5.95	31.63	0	3.89	0	**91**	**+19**
Beaulieu Estuary	28.55	3.12	16.66	36.91	0	5.73	0	**91**	**+11**
Southampton Water	134.70	5.16	89.07	86.86	0	1.96	9.53	**327**	**+35**
Hamble Estuary	8.04	10.42	6.70	12.56	0	9.47	6.02	**53**	**-3**
Portsmouth Harbour	143.46	84.32	44.37	27.12	15.63	0.10	7.59	**323**	**+65**
Langstone Harbour	231.58	54.02	24.15	23.35	18.90	1.41	13.36	**367**	**+24**
Chichester Harbour	218.09	117.59	63.15	114.49	21.08	3.66	11.83	**550**	**+16**
Pagham Harbour	23.48	7.34	2.64	47.45	0.91	0.70	0	**83**	**+11**
Yar Estuary	3.59	3.75	0	16.19	1.82	1.28	0	**27**	**=0**
Newton Harbour	17.66	19.27	2.50	25.99	0	0.01	0	**65**	**+5**
Medina Estuary	2.79	8.51	3.71	4.14	4.91	0.04	0	**24**	**=0**
Bembridge Harbour	5.83	2.25	1.19	1.51	0.36	0.03	0	**11**	**+3**
Solent (open water)	258.06	64.51	1113.66	46.32	63.23	4.61	28.78	**1579**	**+142**
Biotope total	**1119**	**386**	**1374**	**475**	**127**	**33**	**77**	**3590**	**↓**
Improvement in ES relative to baseline	**+292**	**-17**	**+82**	**=0**	**=0**	**+16**	**-46**	**→**	**Extra removal +328 tonnes**

Table 11: Phosphorous removal potential (tonnes yr^{-1}) of biotopes after factoring in condition. Green cells represent the highest biotope removal potential and yellow cells represent the second largest removal potential. (+) represents an improvement in the region's nutrient removal potential relative to the baseline while a (-) represents a reduction in a region's nutrient removal potential. (=) represents no change in the region's nutrient removal potential. (ES) Ecosystem service.

Phosphorous (Median)	Littoral sediments	Littoral sediments (with macro)	Sublittoral sediments	Saltmarsh	Seagrass	Reedbeds	Native oyster	Region total	Improvement in ES relative to baseline
Lymington Estuary	2.37	7.16	2.41	4.12	0	2.18	0	**18**	**-1**
Beaulieu Estuary	1.44	3.58	3.82	4.80	0	3.20	0	**17**	**+3**
Southampton Water	6.78	5.93	20.41	11.30	0	1.36	0.02	**46**	**+13**
Hamble Estuary	0.40	11.96	1.54	1.63	0	6.60	0.01	**22**	**-8**
Portsmouth Harbour	7.92	100.21	17.99	3.53	-3.73	0.12	0.02	**126**	**+19**
Langstone Harbour	11.65	62.02	5.53	3.04	-4.51	1.67	0.03	**79**	**-41**
Chichester Harbour	9.03	139.75	1.52	14.90	-5.03	2.55	0.01	**163**	**-3**
Pagham Harbour	1.18	2.39	0.60	7.51	-0.22	0.84	0	**11**	**-1**
Yar Estuary	0.15	4.46	0	2.10	-0.43	1.52	0	**8**	**=0**
Newton Harbour	0.89	13.12	0.57	3.38	0	0.01	0	**18**	**-9**
Medina Estuary	0.12	10.11	0.13	0.53	-1.17	0.05	0	**10**	**=0**
Bembridge Harbour	0.32	1.53	0.35	0.20	-0.09	0.07	0	**2**	**-1**
Solent (open water)	12.98	20.96	255.15	6.03	-15.10	9.22	0.04	**289**	**+201**
Biotope total	**55**	**383**	**310**	**62**	**-30**	**29**	**0.13**	**811**	**↓**
Improvement in ES relative to baseline	**+21**	**-96**	**+263**	**=0**	**=0**	**+8**	**-24**	**→**	**Extra removal + 175 tonnes**

4 Economic Valuation of Benthic Water Quality Ecosystem Services

In this part of the report, economic valuation methods are applied to identify values for the N and P removal ecosystem services identified in Part 3. The valuation of water quality related ecosystem services is known to be a challenging area of analysis – hence the need for this project. This section starts by exploring different valuation options have been considered that transfer different types of existing evidence ('value transfer') or that would require further primary research to be completed. We then discuss the potential for using a replacement cost method for replacing ecosystem services with a human-engineered system as an estimate of the value of providing ecosystem services.

4.1 Valuing Waste (Nutrient) Remediation

The ideal valuation approach for nutrient reduction services would be based on an individual's Willingness to Pay (WTP) for explicit and marginal improvements in an economic good such as increased recreational opportunities, enhanced aesthetics, and greater biodiversity protection. The recreational value stated by Southeast UK survey respondents in e.g., the National Water Environment Benefits Survey (2007, 2013; see DEFRA Enabling a Natural Capital Approach (ENCA) resources database) could be linked to improvement provided by a biotope (e.g. saltmarsh) to provide a value-based monetary estimate (this is considered further in Part B section 6.4 Recreational valuation). However, if respondents also *internally* considered benefits to fish populations or carbon sequestration when stating their WTP for the hypothetical water clarity improvement, then we may be double counting benefits and thus over-estimating the contribution of each biotope's water claritygenerating functions (Farber *et al.,* 2006). Therefore, at present we lack valuation studies of the appropriate geographic scale and detail to be able to isolate and defensibly estimate values for individual contributions of each function to the final economic good. For example, we need data on, among other things, how to apportion Solent-specific WTP values for water clarity on a per hectare/biotope basis.

An alternative approach for valuing nutrient reduction, used in this study, is to value the biophysical change directly (reduction in N or P) rather than relying on an ecological endpoint (e.g., improvement in water clarity), which is then used to value a subsequent economic good (e.g. tonnes of N removed). While values for N reduction can be found from market prices for N offset credits (e.g., ENtrade), we believe these prices to be too variable for application at the SEMS catchment scale of this analysis (e.g. the Reverse Auction approach involves local farmer or land manager bids for the volume of activity they are prepared to offer in a particular catchment area e.g. length of hedgerow planted). We therefore rely instead on the replacement cost approach which has been used frequently in the scientific literature (Cole and Moksnes, 2016; Costanza *et al.,* 2014; La Notte *et al.,* 2017) and uses the cost difference associated with reaching a nutrient reduction target based on the capacity of natural systems as opposed to utilising human-generated alternatives.

To generate relevant estimates of economic value associated with natural N and P removal, we rely on actual mitigation costs of nutrient reduction measures undertaken on the UK's southeast coast. Replacement costs for removing a kilogram of N vary substantially so were extracted from a combination of nutrient management and planning documents for Poole Harbour (Bryan *et al.,* 2013; RSPB 2013; BPPDC 2017), which together provide some of the most comprehensive regionally-focussed valuation estimates for N in the UK. Mitigating costs for additional P loads to achieve neutral development were taken from an interim (2019-2025) plan for the River Avon (RAWG, 2019) another neighbouring catchment of the Solent. A summary of these costs is given in Table 12 and included measures such as catchment sensitive farming approaches (CSF), payments for ecosystem services schemes (PES) and costs involved with upgrades to existing wastewater treatment plants and associated drainage infrastructure. Our dataset assumes managers select feasible measures for a given watershed and then select the least-cost option, which is based on the average cost for that measure. The average annual cost effectiveness for removing N varies respectively from £23-1100 N kg^{-1} or between £5-895 kg^{-1} in the case of P (Table 12; see also Appendix Table S7 for full breakdown). Average replacement costs of reducing N and P from these sources are estimated as £295 kg^{-1} for N and £282 kg^{-1} for P which we used in our calculations as our mid-range conservative ecosystem replacement value estimates. To incorporate cost variability, we also consider the lowest and highest cost values mostly based on diffuse CSF initiatives and traditional water treatment to remove N and P from upstream point sources. We note, however, that the existence of relatively expensive waste water infrastructure alternatives for reducing N and P may cause our maximum estimates to overestimate the true economic benefit of this service. Thus, we believe that the median price for N and P used in this study provides a reasonable approximation of a typical cost and provides evidence of a willingness to invest in these types of services.

Table 12 Summary of estimated replacement and abatement costs - nitrogen and phosphorous removal Table from Watson *et al.,* (2020)

Value	Valuation references	Notes
£5-23 N or P (£ kg^{-1})	Bryan *et al.,* (2013); RSPB (2013); BPPDC (2017); RAWG (2019)	Application of CSF measures (e.g. use of clover in place of N fertiliser, establishment of cover crops following winter wheat, regulatory controls on agricultural P).
£295-895 N or P (£ kg^{-1})		Change of agricultural land use to less intensive grass production through direct land purchase or Payments for Ecosystem Services (PES) schemes (e.g. conversion to woodland or wetlands).
£282 -1100 N (£ kg^{-1})		Upgrades to existing wastewater treatment plants and associated drainage infrastructure including reducing flow to Sewage Treatment Works (STWs) through water efficiency measures and/or improvements to sewage discharge quality (e.g. N or P stripping).

Based on the average replacement costs cited above, total N and P removal by the SEMS biotopes are estimated to be worth in the region of: £962 million (N) and £179 million (P) when applying the median biophysical rates (Table 13). More refined estimates factoring in the condition of the biotopes in the SEMS would estimate these values to be considerably higher at just over £1 billion (N) and £228 million (P) respectively. A full breakdown of (minimum, average and maximum) costs for N and P are given in Appendix Table S10.

Table 13 Summary of the estimated nutrient reduction value provided by biotopes on the UK's Southeast coast. Biophysical estimates presented in the table have been rounded to the nearest whole (tonne yr^{-1}). To ensure greater accuracy economic calculations were based on biophysical estimates to four decimal places. Negative values indicate net loss of the nutrient from the biotope. (M = million).

Unit	Biotope	Economic value captured	Biophysical change valued in analysis (Median tonnes yr^{-1})	Total average value per hectare (Median £ Annualized)	Total average value (£ Median Annualized)	Biophysical change valued in analysis (SEMS condition tonnes yr^{-1})	Total average value per hectare (SEMS condition £ Annualized)	Total average value (£ SEMS condition Annualized)
Nitrogen	Littoral sediments	Based on the cost of replacing artificial substitutes with the ecological service of waste remediation where cost is a proxy for nitrogen removal benefits of this regulation.	827	£39,300	£243.97 M	1119	£53,198	£330.04 M
	Littoral sediments (with macroalgae)		403	£73,578	£118.89 M	386	£70,553	£114.01 M
	Subtidal sediments		1292	£19,559	£381.12 M	1374	£20,797	£405.26 M
	Saltmarsh		475	£111,009	£139.98 M	475	£111,009	£139.98 M
	Seagrass		127	£53,607	£37.41 M	127	£53,602	£37.41 M
	Reedbeds		17	£18,869	£5.15 M	33	£35,550	£9.71 M
	Native oyster (*Ostrea edulis*)		123	£12,774	£37.44 M	77	£8,002	£22.72 M
	Total N		3264		£962.65 M	3591		£1,059.16 M
Phosphorus	Littoral sediments	Based on the cost of replacing artificial substitutes with the ecological service of waste remediation where cost is a proxy for phosphorus removal benefits of this regulation.	34	£1,555	£9.65 M	55	£2,510	£15.57 M
	Littoral sediments (with macroalgae)		479	£83,853	£135.09 M	383	£68,867	£108.06 M
	Subtidal sediments		47	£677	£13.17 M	310	£4,487	£87.43 M
	Saltmarsh		62	£13,807	£17.39 M	62	£13,807	£17.39 M
	Seagrass		-30	-£12,239	-£8.53 M	-30	-£12,239	-£8.53 M
	Reedbeds		21	£21,448	£5.84 M	29	£30,349	£8.29 M
	Native oyster (*Ostrea edulis*)		25	£2,483	£6.77 M	0.14	£14	£0.04 M
	Total P		639		£179.39 M	811		£228.63 M
	SEMS Total		**3903**		**£1,142.03 M**	**4402**		**£1,288.25 M**

Generally, the total values (£) per hectare of vegetated biotopes were higher than comparative estimates for "bare" (littoral or sublittoral) sediment or oyster beds (Table 13), with saltmarsh being the most valuable biotope for N uptake per ha (£111,009 yr^{1}). Littoral sediments overlain with macroalgae were the most valuable for P on a per ha basis (£68,867 yr^{-1}) followed by reedbeds (£30,349 yr^{1}). When considering the total biotope values, littoral and sublittoral sediments were collectively more valuable for N removal, estimated to be worth 330 million yr^{-1} and 405 million yr^{-1} respectively (Table 13) while littoral sediments with macroalgae were of greater worth for P, with remediation estimates of £108 million yr^{-1}.

These valuations can also be disaggregated to provide nutrient reduction values (£) specific to different regions in the SEMS. Table 14 provides a summary of the value of N and P removal in each region depending on if the condition of the biotopes is factored into the valuation estimates. Omitting the open water region of the Solent (which has the highest economic value for N and P removal; when considering biotope condition), the four largest estuaries in the SEMS (Portsmouth, Langstone and Chichester Harbours and Southampton Water) had the highest economic value associated with removing N and P. A more detailed breakdown in Table 15 for the value of N removal by individual biotopes in each region suggests that in all four of these estuaries bare littoral sediment (i.e. those without macroalgae) currently provide the largest equivalent economic value, with replacement costs ranging from around £40-70 million. Replacement costs for N by saltmarsh biotopes were also particularly high (£25-33 million) in Chichester Harbour and Southampton Water (Table 15).

Table 14 Summary of the estimated nutrient reduction value of biotopes in each region based on extent and condition. (M = million).

	Nitrogen baseline	Nitrogen baseline with condition	Phosphorus baseline	Phosphorus baseline with condition
Lymington Estuary	£21.30 M	£26.75 M	£5.37 M	£5.14 M
Beaulieu Estuary	£23.56 M	£26.83 M	£4.05 M	£4.75 M
Southampton Water	£86.18 M	£96.55 M	£9.24 M	£12.92 M
Hamble Estuary	£16.52 M	£15.70 M	£8.43 M	£6.25 M
Portsmouth Harbour	£76.22 M	£95.16 M	£30.30 M	£35.55 M
Langstone Harbour	£101.16 M	£108.19 M	£33.93 M	£22.40 M
Chichester Harbour	£157.47 M	£162.22 M	£46.67 M	£45.89 M
Pagham Harbour	£21.15 M	£24.34 M	£3.31 M	£3.47 M
Yar Estuary	£7.86 M	£7.86 M	£2.20 M	£2.20 M
Newton Harbour	£17.74 M	£19.30 M	£7.58 M	£5.07 M
Medina Estuary	£7.11 M	£7.11 M	£2.76 M	£2.76 M
Bembridge Harbour	£2.40 M	£3.30 M	£0.84 M	£0.67 M
Solent (open water)	£423.99 M	£465.86 M	£24.70 M	£81.57 M
Total	**£962.65 M**	**£1,059.16 M**	**£179.39 M**	**£228.63 M**

Table 15 Regional breakdown of the nitrogen removal value (£) by biotope (based on extent and condition) in order of total value. (M = million).

	Subtidal sediments	Littoral sediments	Saltmarsh	Littoral sediments (with macroalgae)	Seagrass	Native oyster (*Ostrea edulis*)	Reedbeds
Lymington Estuary	£1.75 M	£12.67 M	£9.33 M	£1.84 M	£0 M	£0 M	£1.15 M
Beaulieu Estuary	£4.91 M	£8.42 M	£10.89 M	£0.92 M	£0 M	£0 M	£1.69 M
Southampton Water	£26.28 M	£39.74 M	£25.62 M	£1.52 M	£0 M	£2.81 M	£0.58 M
Hamble Estuary	£1.98 M	£2.37 M	£3.71 M	£3.07 M	£0 M	£1.78 M	£2.79 M
Portsmouth Harbour	£13.09 M	£42.32 M	£8.00 M	£24.87 M	£4.61 M	£2.24 M	£0.03 M
Langstone Harbour	£7.12 M	£68.32 M	£6.89 M	£15.94 M	£5.57 M	£3.94 M	£0.42 M
Chichester Harbour	£18.63 M	£64.34 M	£33.77 M	£34.69 M	£6.22 M	£3.49 M	£1.08 M
Pagham Harbour	£0.78 M	£6.93 M	£14.00 M	£2.17 M	£0.27 M	£0 M	£0.21 M
Yar Estuary	£0 M	£1.06 M	£4.78 M	£1.11 M	£0.54 M	£0 M	£0.38 M
Newton Harbour	£0.74 M	£5.21 M	£7.67 M	£5.68 M	£0 M	£0 M	£0 M
Medina Estuary	£1.10 M	£0.82 M	£1.22 M	£2.51 M	£1.45 M	£0 M	£0.01 M
Bembridge Harbour	£0.35 M	£1.72 M	£0.44 M	£0.66 M	£0.11 M	£0 M	£0.01 M
Solent (open water)	£328.53 M	£76.13 M	£13.66 M	£19.03 M	£18.65 M	£8.49 M	£1.36 M
Total	**£405.26 M**	**£330.04 M**	**£139.98 M**	**£114.01 M**	**£37.41 M**	**£22.72 M**	**£9.71 M**

Table 16 Regional breakdown of the phosphorous removal value (£) by biotope (based on extent and condition) in order of total value. (M = million).

	Littoral sediments (with macroalgae)	Subtidal sediments	Saltmarsh	Littoral sediments	Reedbeds	Native oyster (*Ostrea edulis*)	Seagrass
Lymington	£2.02 M	£0.68 M	£1.16 M	£0.67 M	£0.61 M	£0 M	£0 M
Beaulieu	£1.01 M	£1.08 M	£1.35 M	£0.41 M	£0.90 M	£0 M	£0 M
Southampton	£1.67 M	£5.75 M	£3.19 M	£1.91 M	£0.38 M	£0.01 M	£0 M
Hamble	£3.37 M	£0.43 M	£0.46 M	£0.11 M	£1.86 M	£0 M	£0 M
Portsmouth	£28.26 M	£5.07 M	£0.99 M	£2.23 M	£0.03 M	£0.01 M	-£1.05 M
Langstone	£17.49 M	£1.56 M	£0.86 M	£3.29 M	£0.47 M	£0.01 M	-£1.27 M
Chichester	£39.41 M	£0.43 M	£4.20 M	£2.55 M	£0.72 M	£0 M	-£1.42 M
Pagham	£0.67 M	£0.17 M	£2.12 M	£0.33 M	£0.24 M	£0 M	-£0.06 M
Yarmouth	£1.26 M	£0 M	£0.59 M	£0.04 M	£0.43 M	£0 M	-£0.12 M
Newton	£3.70 M	£0.16 M	£0.95 M	£0.25 M	£0 M	£0 M	£0 M
Medina	£2.85 M	£0.04 M	£0.15 M	£0.03 M	£0.01 M	£0 M	-£0.33 M
Bembridge	£0.43 M	£0.10 M	£0.06 M	£0.09 M	£0.02 M	£0 M	-£0.02 M
Solent (open water)	£5.91 M	£71.95 M	£1.70 M	£3.66 M	£2.60 M	£0.01 M	-£4.26 M
Total	**£108.05M**	**£87.43 M**	**£17.39 M**	**£15.58 M**	**£8.29 M**	**£0.04 M**	**-£8.54 M**

Regional biotope valuations for P were also highest in Solent open water region (when factoring in condition; Table 15) followed by Portsmouth, Langstone and Chichester Harbours. However, this time the greatest replacement costs in these harbours were related to littoral sediments overlain with macroalgae (~£17-40 million; Table 16). Due to the biophysical estimates for seagrass being negative the overall and regional economics values for P removal by seagrass are also negative. This continued P release could therefore be considered as a costly ecosystem disservice as increasing P into interstitial porewaters may under some circumstances speed up eutrophication. The negative economic values for seagrass P sequestration also suggest that other biotopes or man-made alternatives would be better replacement cost options to negate P loading if this was considered a problem in a particular region (e.g. from our loading comparisons in section 3.5 Chichester and Pagham Harbour would be the only areas with a combination of seagrass beds and P limiting impacts). However, such trade-offs of course must consider the other potential benefits provided by seagrass (e.g. high N sequestration) in areas that are not P limited (e.g. Portsmouth or Langstone Harbour) where the benefits of other ecosystem services many outweigh its (potential) disservices.

5 Summary of Existing Evidence for Nitrogen and Phosphorus Removal by Biotopes in the Solent

5.1 Key Points on the Extent and Condition of Natural Capital

Data on extent and condition of NC stocks are central to assessing the flow of ecosystem services and associated benefits from NC, and datasets for biotopes, species and water column NC assets are relevant for the assessment of a wide range of ecosystem services beyond water quality. Within the SEMS — where extent of features has been assessed by survey and condition assessments undertaken — detailed data sources are available on the extent and condition of designated biotope features. Yet, even in a system as well studied as the SEMS many coastal biotopes are still difficult to map and quantify accurately due to significant knowledge gaps. The creation of an up to date biotope map in this project is based on 'best available evidence' and the translation of JNCC confidence scores demonstrates that there remains a lack of confidence for some elements of the baseline data that can inform on the 'extent' of the biotope natural capital assets. Overall, the confidence in extent of intertidal biotopes within the main harbours and estuaries of the SEMS (e.g. littoral sediments, macroalgal mats, saltmarsh and reedbeds) was generally higher than subtidal biotopes (e.g. seagrass and oyster beds) which often (but not always) occur outside of the main designated sites (i.e. in the Solent channel). This reflects technical and logistical challenges combined with financial constraints for collecting consistent temporal data of biotopes in a sublittoral location. Therefore, the total extent of some biotopes (e.g. seagrass) in the SEMS could be assumed to be an underestimate of total area as little or no survey work has been conducted in some locations to confirm presence and extent. For example, a recent re-survey by the EA in 2018 (Green 2018) of seagrass bed extents originally surveyed by Marsden & Scott (2014) in the West Isle of Wight and North Solent area (West of Yarmouth and the Needles MCZ) have highlighted extent to be 23% bigger (Yarmouth beds only) than in the previously recorded survey. The report also outlines that it is likely that the North Solent bed is bigger than the extent reported here, as the bed boundaries were not encountered to the east, west and north of the surveyed area.

There is also potential for the EUNIS classification process to focus at smaller spatial scales. Here, we generally considered EUNIS biotopes at level 3 & 4, but the resolution limits fine-scale spatial variation in biotopes or their characterising species (Parry 2019). More detailed mapping data at level 5 would be particularly pertinent for NC assessments as species composition, especially within sediment systems is known to influence N and P cycling across very small biogeographical areas (Adams *et al.,* 2012; Cook *et al.,* 2004). As an example, only two locations in the SEMS have sufficient EUNIS level 5 mapping data available (Thomas *et al.,* 2016) to disaggregate the effects of benthic community composition on sediment ES flows. As we have done here, there is also a strong case to separate green macroalgal mat sediment systems dominated by *Ulva* and *Enteromorpha* spp*.* from littoral and sublittoral sediment EUNIS classifications related to the different functioning of these biologically mediated biotopes. Other EUNIS biotopes not included in this assessment may also have an important role in maintaining water quality. While many of these biotopes (e.g. kelp beds, polychaete reefs, maerl beds, epiphyte and sponge communities etc.) were absent or only comprised small areas in the SEMS, future efforts to include the full breadth of NC biotopes available in a region would be important to allow the value of all biotopes to be considered in any future management decisions.

An advantage of extent datasets is that they are regularly updated, and thus are useful for accounting purposes, provided that uncertainty is well addressed when comparing maps over different points in time. A drawback, however, is that extent data do not entirely capture the dynamics or condition of ecosystems and their relative ability (or capacity) to provide ecosystem services over time. For example, the condition of biotope features is not currently provided by modelled seabed data (e.g. UKSeamap). Therefore, indicators for monitoring Good Environmental Status in relation to WFD are potentially highly relevant to the assessment of extent and condition of NC assets because such assessments are routinely updated. For the purpose of this work, ecosystem condition is used as a synonym for 'ecosystem state' and embraces legal concepts (e.g. ecological status under the Water Framework Directive) and refers to the physical, chemical, and biological condition or quality of a biotope at a particular point in time. This definition corresponds well with the definition published in the SEEA EEA technical recommendations: "ecosystem condition reflects the overall quality of an ecosystem asset in terms of its characteristics". In turn, these indicators should be relevant for policy and decision-making, because they reflect a key water quality condition indicator for estuaries and coastal ecosystems and their ability to deliver ecosystem services.

Our examination of the condition of the Solent's marine and coastal biotopes with regard to their capacity to deliver key water quality related ecosystem services, indicates that many biotopes in the SEMS were delivering at a higher (e.g. littoral sediments and reedbeds) or lower capacity (macroalgal mats and native oysters) than if we had only examined the baseline extent data. This is an important consideration, because condition assessments are often omitted when creating NC or ecosystem service accounts (e.g. ONS 2019), potentially leading to an undervaluation of the UK's ecosystems services and in the case of the SEMS the (theoretically) millions of additional pounds the natural environment conveys to the economy. It should be noted that these valuations are dependent on the measure of condition used to develop these accounts (in this case WFD measures) and that adjustments to the analysis to reflect future changing ecosystem conditions, whether due to, ecological recovery, restoration or an altered management regime, may also need further consideration.

5.2 Linking the Natural Capital Stocks to Flows of Ecosystem Services

Alongside mapping the extent and condition of NC, the sensitivity surrounding the metrics used for assessing the flows of ES from stocks of NC also warrants discussion. In this part of the report we primarily focused on the ecosystem service of waste (nutrient) remediation which has been defined as "*The removal of waste products from a given environment by ecosystem processes that act to reduce concentrations of wastes by the mechanisms of cycling/detoxification, sequestration/storage and export*"(Watson *et al.,* 2016b). As with many other regulating ecosystem services (e.g. carbon sequestration), the capacity of biotopes to reduce nutrient loads is complex and is rarely measured using a single composite indicator. Fluxes of N, N_2, NO_2, P, PO_4^{3-} to and from the water column are often the most direct methods available for estimating the combined processes of denitrification, burial in sediments and assimilation in biogenic tissues, but in many cases these measurements were not available from field studies in the SEMS. Even within the constraints of the scientific literature selected for this report, there was considerable variation in methods used across studies. As such, we used the most pertinent studies for temperate coastal biotopes, but all used a variety of methods: field sampling, laboratory experiments, differing flux measurement techniques, or species (e.g. *Crassostrea virginica)*. This lack of standardization led to our conservative approach for estimating N and P removal estimates by presenting upper and lower reported bounds alongside the mean and median values. The median and range values together provide important information regarding N and P removal variability, with some median values being considerably higher or lower than the mean values. For example, the mean P burial by littoral and sublittoral sediments is -1455 P tonnes yr^{-1}, whilst the median value is 81 P tonnes yr^{-1} due to the median estimates of P burial not being skewed by the extremely large P release at only a small proportion of reference study sites (e.g. Thornton *et al.,* 2007). We therefore strongly recommend that users of this report explicitly state the calculation method used; but also acknowledge the variability for different biotopes because of experimental methods, species choice and the influence of local environmental factors. Overall, we recommend the use of the median biophysical rates calculated here for N and P as these are likely the most conservative estimates of total nutrient removal capacity.

The median biophysical rates selected in this study generally indicate higher N removal rates (per m^2) in coastal saltmarsh, seagrass meadows and native oyster beds, while P removal is highest in macroalgal dominated sediments and reedbed biotopes (per m^2). Yet, when considered at the level of the whole SEMS intertidal sediment systems (including those with macroalgae) were the largest contributing biotopes for N removal, while macroalgal mat sediments, subtidal sediments and saltmarsh were the largest removers of P (when factoring in extent and condition). Nevertheless, when considered at the level of individual catchments, other biotopes (e.g. seagrass in the Solent channel or reedbeds in the Hamble estuary) often had considerable N or P removal potential. These considerable differences in relative biotope contributions to nutrient removal illustrate how potential gains (or losses) in ecosystem service production can be interoperated depending on how they are analysed and delineated over large spatial scales. By breaking our estimates down to a regional level allows managers and policymakers to know which ecosystem service provided by a marine or coastal biotope in a specific location is most valuable and most vital to human livelihoods, or if restored could provide the greatest gains.

Developing quantitative targets that more explicitly state how much of each outcome (i.e. how much tonnes yr^{-1} of N or P removal) that is needed to meet "healthy" conditions or meet prescribed policy goals (e.g. development offsets of N) is the next most important and difficult step in putting these biophysical estimates into context. N and P loading estimates from terrestrial and offshore sources have been established for all regions within the SEMS (with the exception of the Solent Channel). Thus, by comparing our biophysical estimates against these input figures, we can begin to explore the effect of human activities on system targets to reduce these nutrients (i.e. achieving nutrient neutrality for new development in the Solent region). This work identified that there is an N surplus (a positive figure) after bioremediation takes place in all of the catchments in the SEMS suggesting that mitigation is required to achieve N neutrality. There were of course large differences between the catchments, suggesting some may require greater interventions than others. For example, the low surplus of N in Langstone Harbour could reflect recent efforts to reduce N into the harbour, suggesting it is already on the path to recovery. In the case of P, our calculations revealed a deficit (a negative figure) in many catchments suggesting no mitigation is required. However, there was an overall surplus of P in Southampton Water, Chichester Harbour and Pagham Harbour. We advise that this issue is examined further in relation to the impact of housing development on these designated sites. Where N or P budget calculations indicate a surplus of nutrients, the use of holistic ecosystem-based management strategies for restoring or improving the condition NC stocks (habitats/biotopes) could be one mitigation option to retain some of the additional nutrient input (e.g. saltmarsh restoration or creation). Biotope restoration however, is not a substitute for reduction in land-based inputs to the system, but rather a potential "safety net" to reduce additional downstream impacts.

The proportion of N & P that can be considered permanently removed for the purposes of water quality will also depend on the material's fate and the time scale of interest (Beeaumont *et al.,* 2014; Kellogg *et al.,* 2014). Our study has considered two of the primary mechanisms for N and P removal in marine systems (long-term burial and denitrification), but also for the first time has included biogenic assimilation in oysters alongside these other processes. In the case of N and P assimilated into oysters, separation of non-harvested and harvested populations is essential when evaluating NC and valuing ES, as only harvested biogenic material will result in permanent nutrient removal. SEMS native oyster shells were, therefore, considered net nutrient sinks, as they are part of commercially exploited stocks, although the impact of fishing controls and restoration could change this assumption in the future.

Long term burial and denitrification estimates in this study have also been estimated from extrapolation of sedimentation rates and N and P content of established biotope sediments (e.g. Adams *et al.,* 2012) to give an indication of the potential level of nutrient "stock" over a yearly cycle. While N removed *via* denitrification is likely permanent, annual burial estimates of N & P could change on an annual basis depending on the influence of local environmental factors and biogeographically relevant taxa e.g. tidal regime, substrate, life history and climate factors could all affect annual burial storage estimates. For vegetated and angiosperm-based biotopes, there is increasing evidence that a large proportion of the N, and P burial assimilatory benefits provided by these biotopes occurs through export and storage of detritus to pelagic sediments and the deep sea (Duarte and Krause-Jensen, 2017; Krause-jensen and Duarte, 2016; Queirós *et al.,* 2019). In the case of intertidal sediments overlain with macroalgae, many studies have shown that on a seasonal time scale macroalgae in eutrophic waters switch from being a net sink of N and P early in the growing season, to a net source of nutrients in late summer when productivity declines (Tyler *et al.,* 2001;Gao *et al.,* 2013). Intertidal macroalgal mat sediment systems may, therefore, be considered as temporary or seasonal stocks of NC that will inevitably act to alter the local exchange of mass and energy at the sediment–water interface on an annual basis thereafter acting as nutrient donors to long term reservoirs located elsewhere (e.g. subtidal depositional areas or the deep sea; see (Krause-jensen and Duarte, 2016). Perhaps the most relevant evidence for these processes in the Solent comes from Trimmer *et al.,* (1998;2000) who calculated the rate of N mineralised within the sediments of Langstone and Chichester Harbour, and suggested that the vast majority of sequestered N (>98%) remained within the sediment systems of the harbours over a yearly cycle. However, few other studies have verified the potential proportion of N or P burial macroalgal mats may export to offshore food webs, limiting the inclusion of lateral nutrient export as a removal mechanism in our calculations. Further measurements of dissolved organic nutrient production generated by macroalgal mat and other angiosperm biotopes, could further increase the significance of the sequestration fluxes we estimate.

5.3 Conclusions on the Nutrient Valuations

The range of economic values calculated in this study are designed for use around the UK's coast to capture the current monetary value of coastal biotopes for maintaining water quality with respect to removing N and P. Our analysis indicates that N removal by biotopes is around 5x more valuable (£962 million yr^{-1}) than P (179 million yr^{-1}), based on median biophysical values and average replacement costs. This is largely due to the fact that on a per unit basis, estuaries are generally heterotrophic with regards to P and therefore their sediments often represent a net source of P to the ocean (e.g. segrass sediments and if we had considered the mean biophysical rates for littoral or sublittoral sediments these would be negative for P) lowering the total potential net sequestration value of this ES. Based on our median biophysical rates, we estimate the total present value of benefits from the resulting removal of nutrients to be approximately £1.1 billion (equivalent to ~ 35,965 UK £ ha^{-1}). This value is at the mid-to upper end of other monetary estimates in the literature for valuing N and P in coastal ecosystems (e.g. Costanza *et al.,* 2014; Cole and Moksnes, 2016) but may nonetheless be considered conservative and useful for raising awareness of society's dependence on regulating ES to improve water quality.

Importantly, the average replacement costs per hectare of individual biotopes in this study also showed substantial variation (~£677-£111,000) with regional total values varying strongly between watersheds based on an individual catchment's aggregated collection of biotopes. For the SEMS and other marine protected sites, this conclusion is important because improved decision-making requires information on the economic value associated with relatively small marginal changes in ecosystems. The inclusion of per ha values here indicate that saltmarsh biotopes were the most valuable (£ ha) at removing N, while littoral sediment overlain with macroalgal mat biotopes were the most valuable (£ ha) at removing P. This latter result is somewhat controversial as much of the literature on macroalgal mats focuses on their negative effects, including induction of hypoxia, release of toxic hydrogen sulphide into the sediments, and the loss of ecologically and economically important species (Raffaelli, 2000; Wezel *et al.,* 2002; Thornton, 2016). However, elucidating the net impact of macroalgal mats' (both positive and negative effects) remains an important challenge (Lyons *et al.,* 2012). In this quantitative assessment, we chose to focus on community and ecosystem-level responses of macroalgal blooms. Results at these spatial scales suggest that in addition to their relatively high economic replacement costs, littoral sediments overlain with macroalgal mats have ecological effects that may increase transfer of nutrients from the water column to the sediments, thereby reducing N and P levels in eutrophic waters. However, we acknowledge that macroalgal mat effects on other socio-ecological outcomes remain to be synthesized in our valuation estimates (e.g. impacts on protected habitats and bird species, leisure and tourism activities, property prices).

Truly robust replacement cost values would require more nuanced regional quantitative adjustments that consider the future extent and condition of the NC stocks and also the willingness to pay of beneficiaries to implement the actual replacement cost of the NC asset (e.g. saltmarsh creation) with another mitigation measure, either natural (e.g. catchment sensitive farming) or manmade (e.g. sewage water upgrades). For example, in the case of water quality it is seldom that natural remediation is adequate and hence built water treatment works are often also required. A good example of this combined approach has recently been proposed for the Solent, when in September 2019, councillors in Fareham approved a series of measures including the creation of wetlands to remove N, reduce fertiliser use on farmland and improve the wastewater treatment works at Peel Common. Catchment based nutrient natality schemes therefore will require specialist design input based on sound environmental information that will need consultation with relevant statutory bodies. These processes are likely to be easier where wetlands and other aspects of NC are an integral part of a larger development. To aid such processes, we suggest that our replacement cost values could be used in more detailed cost benefit calculations to achieve nutrient neutrality. They also provide a sound baseline to inform on issues such as N and P mitigation in compliance with the Habitats Regulations and the UK governments 25-year plan to restore and enhance NC *via* environmental net gain solutions.

Other ES provided by coastal biotopes, such as leisure and tourism activities, the production of harvestable fish and invertebrates, and sequestering of CO_2 could also add significant ancillary biotope-related value to locally connected human beneficiaries beyond those estimated here. Thus, the next part of this report will expand our indicator assessments to better understand the full range of ES provide by EUNIS biotopes (Vermaat *et al.,* 2016; Rees *et al.,* 2019), and the potential risks to those biotopes and an evaluation of any loss or gain of monetary value that could result through future impacts or restoration activities (eg. Russell and Greening, 2015). Nevertheless, even without these additional value analyses, it is clear that there have already been large cost savings in terms of water quality for the SEMS human population.

Part B: Analysis of Additional Ecosystem Services Relating to Water Quality

This section builds on the analysis in Part A by quantifying and valuing a number of other water quality-related ecosystem goods and services. Available evidence for biophysical rates and economic value transfer were identified through literature review. Section 6.5 constructs an initial marine ecosystem account for nitrogen, phosphorus, carbon, fisheries, nursey function and recreation services in the Solent.

6 Potential for Valuing Additional Ecosystem Services Linked to Water Quality

The analysis of additional individual ecosystem services with potential economically significant links to water quality are considered in the following section:

- Climate regulation (carbon sequestration and storage)
- Commercial, recreational and subsistence fisheries.
- Nursery function and supporting the existence of biodiversity.
- Recreation, tourism and leisure.

Table 17 Prioritisation of Water Quality Related Ecosystem Services for Economic Valuation

Ecosystem service		Considered in this study	Justification and example of how this ecosystem service links to improving water quality
Regulating	Climate Regulation (carbon sequestration and storage)	Yes	Numerous environmental benefits may result from activities that sequester CO_2 and contribute to environmental security. Habitats that sequester CO_2 help reduce soil erosion and improve water quality and are consistent with more sustainable and less chemically dependent water treatment processes.
	Natural Hazard Protection (e.g. floods, storms)	No	**Not investigated in this study.** The quality of water supply in coastal and island regions is at risk from rising sea level and changes in precipitation. Flood events have contrasting effects on water quality including disruption of normal drainage systems, spillage of raw sewage and animal waste, accelerated discharge of industrial urban toxic materials and nutrients into waterways.
	Sediment Stabilisation	No	**Not investigated in this study.** Saltmarsh and Seagrass habitats will provide ancillary water quality value (£) by reducing sediment erosion.
	Waste Remediation, Detoxification and Storage	Yes	The ecosystem service of Waste Remediation (WR) enables humans to utilise the natural functioning of ecosystems to process and detoxify a large number of waste products (including N and P, heavy metals, persistent organic pollutants, plastics and pathogens) and therefore avoid harmful effects on human wellbeing and the environment. In this report we only consider the N and P removal aspect of WR.
Provisioning	Commercial, recreational and subsistence fisheries. (including inter-tidal Harvesting and Polychaete Bait Fisheries)	Yes	Water quality is very important to fish and shellfish fisheries as poor-quality water can affect the health and growth of the fish or shellfish stocks. In this report we only consider commercial finfish and bivalve stocks. Further investigation is also needed of non-commercial finfish, bivalve and polychaete bait stocks. Some bait fishery valuation estimates are available for Portsmouth, Chichester and Pagham Harbours however (see. Watson *et al.,* 2016a).
	Fresh Water Provisioning	No	**Not investigated in this study** as fresh water provisioning is mostly related to upper catchments such as rivers and lakes. Poor water quality is likely to have a significant impact on water provisioning costs.
	Raw Materials (e.g. biofuels) and Medicinal Resources	No	**Not investigated in this study.** While harvesting of macroalgae is possible, this service is not currently significantly exploited in the UK.
Cultural	Recreation, Tourism and Leisure	Yes	Good water quality, when it can be perceived by recreational users, contributes positively to human wellbeing and is likely to increase demand for recreation, tourism and leisure activities at public waterways.
	Amenity, non-use values	Yes (indirectly)	Amenity and non-use values are captured in the willingness to pay economic valuation methods used to value other bundles of services e.g. Recreation, Tourism and Leisure. Hedonic pricing methods could alternatively be used in future, linking property prices to blue space aesthetic value to derive a standalone valuation for this service of interest to a specific policy question. Care should be taken however not to double count the value of cultural ecosystem services.
Supporting	Nursery Function- and supporting the existence of biodiversity	Yes	Recruitment levels and population sizes of the concerned marine species may be dramatically affected by changes in water quality.
	Biodiversity	No	**Not investigated in this study.** Assessment undertaken as part of bundled (i.e. other) ecosystem services

The following ecosystem services are ***not*** investigated further due to being beyond the funding and scope of this project, but are recognised to provide important ancillary water quality value (Table 17):

- Fresh water provisioning
- Natural Hazard Protection (e.g. floods, storms)
- Raw materials (e.g. biofuels) and medicinal resources
- Sediment stabilisation
- Waste remediation (other than nutrients, e.g., heavy metals, persistent organic pollutants, plastics, pathogens, radioactive wastes).

Also, amenity, non-use values and biodiversity are not considered further as single ecosystem service(s), but are instead considered as part of the discussion of bundled ecosystem services.

6.1 Climate Regulation (Carbon Sequestration and Storage)

This ecosystem service, is defined by the Common International Classification of Ecosystem Services (CICES2016) classification as "climate regulation". This is achieved by numerous ecosystem processes, including both stocks and flows, which can be increased or decreased by different human activities. The important welfare benefit provided by this service is maintaining an equitable climate, which facilitates the existence of life. The SEEA (2012) describes accounting for both the storage and sequestering of carbon as one of the main challenges of quantifying this ecosystem service both in physical and monetary terms, which is still an on-going discussion (e.g. Beaumont *et al.,* 2014). In order to account for both stocks and flows of carbon, the SEEA suggests considering the service of carbon sequestration as *'the net accumulation of carbon in an ecosystem due to both growth of the vegetation and accumulation in below-ground carbon reservoirs'*, and the carbon storage service as *'the avoided flow of carbon resulting from maintaining the stock of above- and below-ground sequestered in the ecosystem'*. The main biotopes providing this service in the SEMS, following the Potts *et al.,* (2014) classification shown in Table 18, are saltmarshes, reedbeds, littoral sediments, seagrass and although not assessed in this report kelp beds. Although kelp beds are likely to have high standing stocks of C as reviewed by Potts *et al.,* 2014 kelp C production is not thought to be stored within kelp beds (Burrows *et al.,* 2017), and therefore these biotopes alone offer no potential for long term C sequestration. Instead The lateral export of kelp detritus into deep sea sediments may ultimately be the fate of a significant fraction of the C produced by kelp, and represent the major contribution of the biotope to the blue carbon inventory. Coastal sublittoral sediments, while not assessed by Potts *et al.,* (2014) were included here due their large extent in the case study area.
In the case of vegetated systems (e.g., saltmarshes, reedbeds, seagrasses and macroalgae), the plants capture CO_2 from the atmosphere and then provide long term storage of that carbon through burial in near-shore sediments. This is sometimes known as 'blue carbon'. It is important to specify that the process valued in monetary terms is sequestration as defined by the SEEA (2012). Carbon capture/fixation in biogenetic material/tissues does not raise the welfare benefit, the benefit comes with carbon burial when the CO_2 is locked away by burial/accumulation in the sediments.

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

- Extent of the marine and coastal biotopes providing the service;
- Carbon sequestration rate (e.g. tonnes of C /m^2/ buried in sediments year)
- The monetary value of carbon (C).

The extent (area) of marine biotopes that provide this service is shown in Table 19. While carbon can occur in many forms for economic valuation and natural accounting purposes, rates of production and sequestration of organic and inorganic carbon (units = g C m^{-2} yr^{-1}), were sourced from the scientific literature (Table 19). In the case of Native oyster (*Ostrea edulis*), we also generated estimates of sequestration (assimilation in shell) for this biotope using studies with *Crassostrea virginica* (Eastern American oyster).

Table 18 Matrix assessment of the provision of carbon sequestration and storage services (UK NEA FO) from habitats in SEMS, including biotope features of MPAs (building on Potts *et al.,* 2014)

			Climate Regulation
			Regulating services
Natural Capital Stocks: Biotopes in the Solent Marine Site	Area (ha)	Area in MPAs (ha) pre-Jan 2020	Carbon Sequestration and Storage
A2.11: Shingle (pebble) and gravel shores	708	708	
A2.3: Littoral mud	6204	6204	3
A2.3: Littoral mud (with mat forming macroalgae)	1616	1616	3
A2.5: Coastal saltmarshes and saline reedbeds	1261	1111	3
A3: Infralittoral rock and other hard substrata	756	345	
A5: Sublittoral sediment	1438	215	
A5.1: Sublittoral coarse sediment	9496	3914	
A5.2: Sublittoral sand	10088	2817	
A5.3: Sublittoral mud	7502	5516	
A5.34: Infralittoral fine mud	400	344	
A5.4: Sublittoral mixed sediments	1497	288	
A5.435: Ostrea edulis beds on shallow sublittoral muddy mixed sediment	2839	1155	1
A5.52: Kelp and seaweed communities on sublittoral sediment	121	121	1
A5.53: Sublittoral seagrass beds and A2.61: Seagrass beds on littoral sediments	698	691	2
B1.21: Unvegetated sand beaches above the driftline	88	11	
B2: Coastal shingle	136	15	
B2.2: Unvegetated mobile shingle beaches above the driftline	251	28	
B3: Rock cliffs, ledges and shores, including the supralittoral	40	2	
C3.21: *Phragmites australis* beds	273	226	3
C2.3: Permanent non-tidal, smooth-flowing watercourses	25	5	

Scale of ecosystem service supplied relative to other features	
#	Significant contribution
#	Moderate contribution
#	Low contribution
#	No or negligible ESP
	Not assessed

Confidence in evidence	
3	UK-related, peer-reviewed literature
2	Grey or overseas literature
1	Expert opinion
	Not assessed

Table 19 Carbon annual removal rates used for biotope types occurring in the SEMS showing mean, median ± range (min and max) reported values. Negative values indicate net loss of the nutrient from the biotope. * Native oyster estimates were made using the Eastern American oyster (*Crassostrea virginica*). Table from Watson *et al.,* (2020)

EUNIS biotope	Area (ha)	Ecosystem process/function	Carbon ($g\ C\ m^{-2}\ yr^{-1}$)				References
			Mean	Median	Min	Max	
Coastal saltmarshes	1261	Burial	210	139.5	18	1713	Burrows *et al.,* 2017
Seagrass beds	698	Burial	83	110	19	191	Burrows *et al.,* 2017; Duarte *et al.,* 2005; Romero *et al.,* 1994
Reedbeds	273	Burial	382	484.5	5.17	554	Brix *et al.,* 2001
Littoral sediment	6204	Burial	155.2	130	18.7	291.6	Burrows *et al.,* 2017
Littoral sediment (macroalgal mats)	1616	Burial	264.2	312.3	96.1	336.3	Trimmer *et al.,* 2000;1998
Sublittoral sediment	19486	Burial	50.6	35	4.6	150	Burrows *et al.,* 2017
*Native oyster (*Ostrea edulis*) reefs	2839	Assimilation in shell (g/individual)	4.9	4.4	0.6	10	Higgins *et al.,* 2011
		Burial	-10.5	4	-71	21	Fodrie *et al.,* 2017

In a similar manner to calculating the N and P biophysical rates in part A, we chose to take a conservative approach for the aggregation of C buried in SEMS marine biotopes by selecting the median burial rates available with the reviewed ranges of sequestration rates in Table 19. Based on data concerning the extent of coastal and marine biotopes the estimate of total quantity of C stored in sediments and native oyster shells is 24,075 tonnes yr^{-1}. This estimate is disaggregated by biotope type in Table 20. Generally, littoral sediments including those with macroalgae were the greatest potential reservoirs of C. Saltmarsh and reedbed biotopes also had relatively high C burial potentials in the Hamble estuary, the Yar estuary and Pagham harbour, while seagrass had the second greatest C removal potential in the Medina estuary.

To represent how the local condition of biotopes in the SEMS may influence C sequestration, we then adjusted the level of baseline (median) ecosystem service based on regional condition classification data (see Part A Section 3.6). Overall, we estimate there would be a 59% increase (14, 224 tonnes yr^{1}) in C sequestration if condition is factored into the baseline calculations. Regional adjustments based on condition are shown in Table 21. Relative to the baseline there were improvements in C removal across all regions with the exception of the Hamble estuary and Chichester Harbour while in the Medina estuary and Newton harbour C sequestration remained similar to the baseline estimates (Table 21). Total C sequestration estimates including biotope extent and condition re-estimated here to be in the region of 38,299 tonnes of C yr^{-1}.

Table 20: Carbon removal potential (tonnes yr^{-1}) of biotopes in the SEMS. Green cells represent the highest biotope removal potential and yellow cells represent the second largest removal potential.

Carbon (Median)	Littoral sediments	Littoral sediments (with macro)	Sublittoral sediments	Saltmarsh	Seagrass	Reedbeds	Native oyster	Region total
Lymington Estuary	230.10	131.16	28.70	117.18	0	82.37	0	**590**
Beaulieu Estuary	197.60	65.58	84	136.71	0	121.13	0	**605**
Southampton Water	932.36	108.67	449.19	321.75	0	64.74	29.65	**1906**
Hamble Estuary	108.95	219.22	33.81	46.53	0	313.17	18.73	**740**
Portsmouth Harbour	768.30	1052.38	214.20	100.44	94.6	7.41	23.44	**2261**
Langstone Harbour	1602.90	1136.70	121.80	86.49	114.4	106.59	41.24	**3210**
Chichester Harbour	2133.30	1467.72	222.25	424.08	127.6	121.13	36.54	**4533**
Pagham Harbour	162.50	43.72	13.30	175.77	5.5	53.29	0	**454**
Yar Estuary	35.10	46.84	0	59.99	11	96.9	0	**250**
Newton Harbour	122.20	240.46	12.60	96.26	0	0.44	0	**472**
Medina Estuary	27.30	106.18	19.60	15.35	29.7	3.29	0	**201**
Bembridge Harbour	31.20	28.11	4.20	5.58	2.2	2.62	0	**74**
Solent (open water)	1786.20	384.10	5616.45	171.59	382.8	348.84	88.87	**8779**
Biotope total	**8138**	**5031**	**6820**	**1758**	**768**	**1322**	**238**	**24075**

Table 21: Carbon removal potential (tonnes yr^{-1}) of biotopes after factoring in condition. Green cells represent the highest biotope removal potential and yellow cells represent the second largest removal potential. (+) represents an improvement in the region's nutrient removal potential relative to the baseline while a (-) represents a reduction in a region's carbon removal potential. (=) represents no change in the region's carbon removal potential. (ES) Ecosystem service.

Carbon (Median)	Littoral sediments	Littoral sediments (with macro)	Sublittoral sediments	Saltmarsh	Seagrass	Reedbeds	Native oyster	Region total	Improvement in ES relative to baseline
Lymington Estuary	516.13	85.76	123	117.18	0	94.18	0	**936**	**+346**
Beaulieu Estuary	320.42	42.88	235.20	136.71	0	138.50	0	**874**	**+269**
Southampton Water	1511.86	71.05	1257.73	321.75	0	69.38	-248.51	**2983**	**+1077**
Hamble Estuary	129.44	143.33	94.67	46.53	0	335.63	-196.42	**553**	**-187**
Portsmouth Harbour	1723.36	1052.38	918	100.44	94.6	7.41	-181.35	**3715**	**+1454**
Langstone Harbour	2599.16	743.22	341.04	86.49	114.4	106.59	-319.15	**3672**	**+462**
Chichester Harbour	2133.30	1467.72	222.25	424.08	127.6	129.81	-282.75	**4222**	**-311**
Pagham Harbour	263.50	28.59	622.30	175.77	5.5	53.30	0	**1149**	**+695**
Yar Estuary	35.10	46.84	0	59.99	11	96.9	0	**250**	**=0**
Newton Harbour	198.15	157.22	35.28	96.26	0	0.44	0	**487**	**+15**
Medina Estuary	27.30	106.18	19.60	15.35	29.7	3.29	0	**201**	**=0**
Bembridge Harbour	69.98	18.38	18.00	5.58	2.2	2.99	0	**117**	**+43**
Solent (open water)	2896.39	251.14	15726.06	171.59	382.8	398.88	-687.7	**19139**	**+10360**
Biotope total	12424	4215	19613	1758	768	1437	-1916	**38299**	↓
Improvement in ES relative to baseline	**+4286**	**-816**	**+12793**	**=0**	**=0**	**+115**	**-2154**	→	**Extra removal +14224 tonnes**

To estimate the economic value of C sequestration by each biotope, we used the British Department of Energy and Climate Change (DECC) low, medium and high range of non-traded carbon prices (Table 22) per tonne of CO_2 equivalent prices (DECC, 2011) based on the marginal abatement cost method. As suggested by others (e.g. Luisetti *et al.,* 2013; Beaumont *et al.,* 2014) we used the non-traded values which represent the maximum marginal abatement cost needed to meet a specific emission reduction target in the future.

Table 22 Summary of short-term carbon abatement costs

Value	Valuation references	Notes
£30-90 Average value £60 (£ Tonnes^{-1})	DECC, 2011	CO_2 abatement potential identified by the UK government based on a short-term non-traded price of carbon of £60 per tonne CO_2 in 2020, with a range of +/- 50% (i.e. central value of £60, with a range of £30 - £90).

Based on data concerning the extent of SEMS coastal and marine biotopes, C sequestration is estimated to be worth £1.44 million yr^{-1} (Table 23) when applying the medium (average) non-traded carbon prices. After factoring in the condition of biotopes we revised this estimate to be approximately £2.3 million yr^{-1} (Table 24), although this could be nearly £3.45 million if the high (maximum) carbon trading prices were utilised. These valuations can also be disaggregated to provide C sequestration values (£) specific to different regions or biotopes in the SEMS (Tables 23, 24).

Table 23 Regional breakdown of the carbon removal value (£) by biotope (based on extent only).

Baseline Extent Only	**Littoral sediments**	**Littoral sediments**	**Sublittoral sediments**	**Saltmarsh**	**Seagrass**	**Reedbeds**	**Native oyster**	**Region total**
Lymington Estuary	£13,806	£7,870	£1,722	£7,031	£0	£4,942	£0	**£0.04 M**
Beaulieu Estuary	£11,856	£3,935	£5,040	£8,203	£0	£7,268	£0	**£0.04 M**
Southampton Water	£55,942	£6,520	£26,951	£19,305	£0	£3,884	£1,779	**£0.11 M**
Hamble Estuary	£6,537	£13,153	£2,029	£2,792	£0	£18,790	£1,124	**£0.04 M**
Portsmouth Harbour	£46,098	£63,143	£12,852	£6,026	£5,676	£445	£1,406	**£0.14 M**
Langstone Harbour	£96,174	£68,202	£7,308	£5,189	£6,864	£6,395	£2,474	**£0.19 M**
Chichester Harbour	£127,998	£88,063	£13,335	£25,445	£7,656	£7,268	£2,192	**£0.27 M**
Pagham Harbour	£9,750	£2,623	£798	£10,546	£330	£3,197	£0	**£0.03 M**
Yar Estuary	£2,106	£2,810	£0	£3,599	£660	£5,814	£0	**£0.02 M**
Newton Harbour	£7,332	£14,428	£756	£5,776	£0	£26	£0	**£0.03 M**
Medina Estuary	£1,638	£6,371	£1,176	£921	£1,782	£197	£0	**£0.01 M**
Bembridge Harbour	£1,872	£1,687	£252	£335	£132	£157	£0	**£0.004 M**
Solent (open water)	£107,172	£23,046	£336,987	£10,295	£22,968	£20,930	£5,332	**£0.53 M**
Biotope total	**£0.49 M**	**£0.30 M**	**£0.41 M**	**£0.11 M**	**£0.05 M**	**£0.08 M**	**£0.01 M**	**£1.44 M**

Table 24 Regional breakdown of the carbon removal value (£) by biotope (based on extent and condition).

Baseline Extent and condition	Littoral sediments	Littoral sediments	Sublittoral sediments	Saltmarsh	Seagrass	Reedbeds	Native oyster	**Region total**
Lymington Estuary	£30,968	£5,146	£7,380	£7,031	£0	£5,651	£0	**£0.06 M**
Beaulieu Estuary	£19,225	£2,573	£14,112	£8,203	£0	£8,310	£0	**£0.05 M**
Southampton Water	£90,712	£4,263	£75,464	£19,305	£0	£4,163	-£14,911	**£0.18 M**
Hamble Estuary	£7,766	£8,600	£5,680	£2,792	£0	£20,138	-£11,785	**£0.03 M**
Portsmouth Harbour	£103,402	£63,143	£55,080	£6,026	£5,676	£445	-£10,881	**£0.22 M**
Langstone Harbour	£155,950	£44,593	£20,462	£5,189	£6,864	£6,395	-£19,149	**£0.22 M**
Chichester Harbour	£127,998	£88,063	£13,335	£25,445	£7,656	£7,789	-£16,965	**£0.25 M**
Pagham Harbour	£15,810	£1,715	£37,338	£10,546	£330	£3,198	£0	**£0.07 M**
Yar Estuary	£2,106	£2,810	£0	£3,599	£660	£5,814	£0	**£0.02 M**
Newton Harbour	£11,889	£9,433	£2,117	£5,776	£0	£26	£0	**£0.03 M**
Medina Estuary	£1,638	£6,371	£1,176	£921	£1,782	£197	£0	**£0.01 M**
Bembridge Harbour	£4,199	£1,103	£1,080	£335	£132	£179	£0	**£0.01 M**
Solent (open water)	£173,783	£15,068	£943,564	£10,295	£22,968	£23,933	-£41,262	**£1.15 M**
Biotope total	**£0.75 M**	**£0.25 M**	**£1.18 M**	**£0.11 M**	**£0.05 M**	**£0.09 M**	**-£0.11 M**	**£2.30 M**

6.2 Commercial, Recreational and Subsistence Fisheries.

Fish and shellfish landed into various ports within the Solent provide food and employment as well downstream economic benefits to the local community. Shellfish aquaculture is one of the main types of fishery in the SEMS. Traditionally the most significant shellfishery of the area was the native oyster (*Ostrea edulis*) fishery, historically hosting the largest self-sustaining native oyster stock in Europe, but this has severely declined historically and recently due to overexploitation (see Gravestock *et al.,* (2014). Other priority shellfish species harvested in the region include the Manila clam (*Ruditapes philippinarum*), Hard-shell clam (*Mercenaria mercenaria*) and Cockle (*Cerastoderma edule*). The wider Solent shellfishery includes Portsmouth Harbour, Langstone Harbour, Chichester Harbour, Southampton Water and portions of the open water area of the Solent which are shown in Figure 12. At present, whist a lease is held, there is no known legal harvesting of shellfish taking place in the Beaulieu or Medina estuaries.

MMO landings data (2017) which have been further refined by the Southern Inshore Fisheries and Conservation Authority (SIFCA), indicate that landings for clam and cockle species in the SEMS area totalled at ~84 tonnes valued at £331,790. The Solent clam fishery has experienced a 70% decline in catches since 2015 (~280 tonnes) and experienced a 53% decline in catches between 2010 (~602 tonnes) and 2015. Based on a three-year average of the MMO/SIFCA data, we estimate the direct Gross Value Added (GVA) value of the Solent's shellfisheries to be in the region of £677,471 per annum. Although only native oysters (*Ostrea edulis*) are commercially exploited in Chichester Harbour at present, there is interest in harvesting all the other bivalves found in the harbour (including Manila clams, Hard-shelled clams and cockles) in the future.

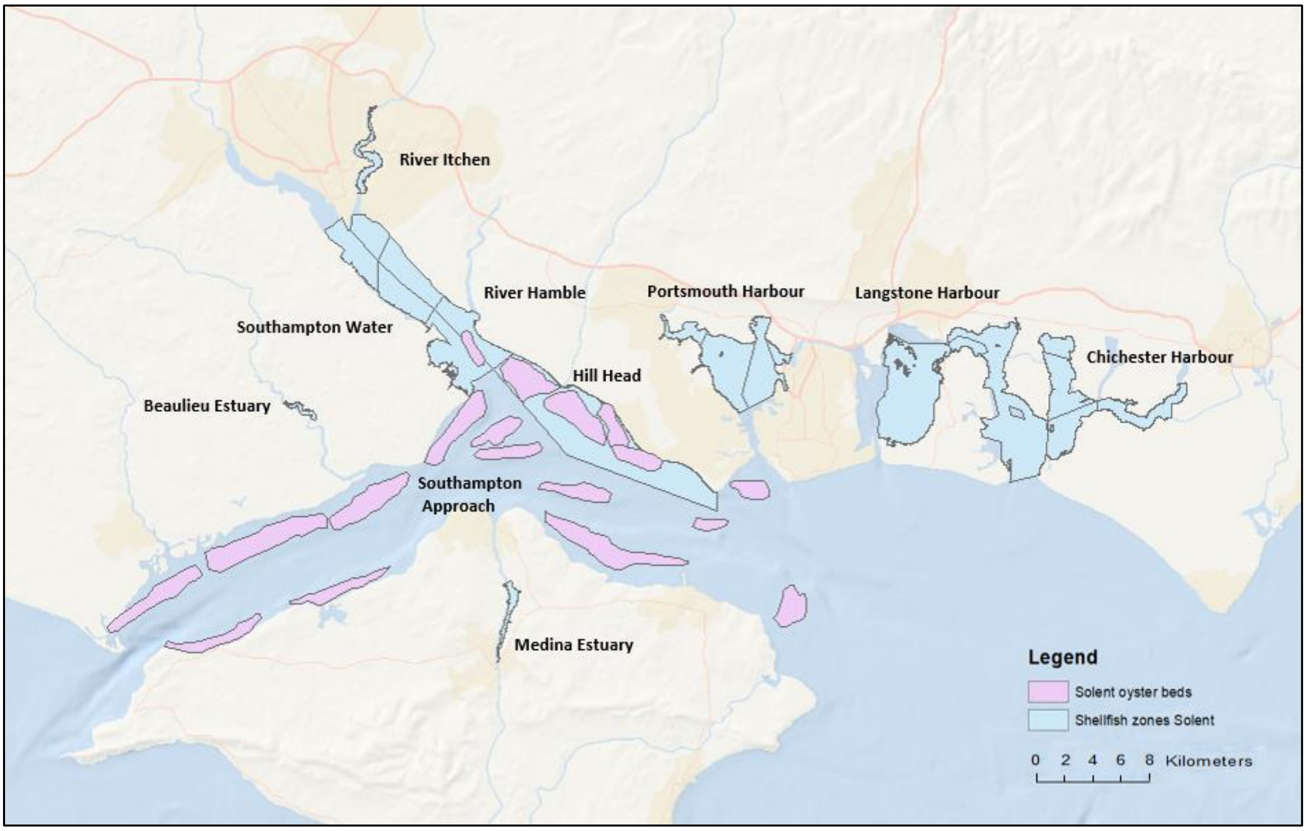


Figure 12 Location of commercial shellfish beds and designated shellfish zones in the Solent

However, the scope of the MMO data does not factor in the indirect economic benefits from shellfish harvesting on supply-chain expenditure (e.g. economic benefits from shellfish depuration processes, boat and machinery maintenance, shellfish transportation, shellfish wholesalers and local shellfish retailers). These additional factors have recently been modelled for the Solent by Williams & Davies, (2018a) & Williams *et al.,* (2018) with the aim to assess the potential indirect GVA from shellfish harvesting, under different water quality scenarios in the Solent. These results are shown in Table 26. The model calculates the Direct and Indirect GVA generated for each given scenario based on if there is a change in shellfish bed quality (and hence water quality), which in turn has impacts on the size of harvest of four key shellfish species (Manila Clam, Hard-shell clam, Cockle and Native Oyster) and potentially the level of required depuration processes required. The conclusion from the research is that better water quality leads to a higher Direct and Indirect GVA, as a result of the increases in shellfish harvest. Reported uplifts of £86,251 per annum are proposed when improving the water quality of beds to class A (Table 26). Based on the baseline estimates (i.e. scenario 2 the current water quality condition of the shellfisheries) it is estimated that the total Gross Value Added (GVA) value of the Solent's shellfisheries is **£1,712,686** per annum. A breakdown of these baseline estimates by species is given in Table 27. In this scenario, all shellfish beds are classified as Class C water level (or above, if their classification is already above C). As all shellfish beds are classified as either C or B, this has no impact on the harvest size. Approximately 49% of total GVA derives from the Direct GVA generated from harvesting Manila clams (£507,939).

Table 25: Shellfish landings by species, volume and value from the Solent (2015-2017). Based on MMO landings data and Southern IFCA expert judgement (from Williams & Davies, 2018a; Williams *et al.,* 2018).

Bivalve species and year	Portsmouth Harbour	Langstone Harbour	Chichester Harbour	Southampton water	Southampton approach	Hill Head	Total (tonnes)	Sum of Value £
2015								
Clams (*M. mercenaria*)	4.5	4.23	0	0	19.67	2.31	30.71	£92,130
Cockles	10.96	0	0	0	0.263	0	11.223	£22,446
Manilla Clam	36.89	6.88	0	0	158.85	6.22	208.84	£939,780
Native Oysters	1.6	1.6	26.3	0	0	0	29.5	£88,500
Pacific Oysters	NA	NA	NA	NA	NA	NA	0.07	£54
2016							**280.34**	**£1,142,910**
Clams (*M. mercenaria*)	0.58	4.13	0	0.767	2.87	2.87	11.217	£33,651
Clams (*R. decussata*)	NA	NA	NA	NA	NA	NA	0.2	£700
Cockles	3.77	0	0	0	1.216	0	4.986	£9,972
Manilla Clam	6.84	4.13	0	8.78	47.62	9.66	77.03	£346,635
Native Oysters	2.5	24.5	28.5	0	0	0	55.5	£166,500
Pacific Oysters	NA	NA	NA	NA	NA	NA	NA	£254
2017							**148.93**	**£557,712**
Clams (*M.Mercenaria*)	2.37	5.07	0	3.36	3.36	2.41	16.57	£49,710
Clams (*R. decussata*)	NA	NA	NA	NA	NA	NA	0.44	£1,500
Cockles	0.98	0	0	0	0.284	0	1.264	£2,528
Manilla Clam	19.126	8.56	0	7.57	8.93	8.57	52.756	£237,402
Native Oysters	2.1	5.72	5.73	0	0	0	13.55	£40,650
Pacific Oysters	NA	NA	NA	NA	NA	NA	NA	N/A
						Grand Totals	**84.58**	**£331,790**

Table 26 Total GVA of shellfisheries based on improving the classification of all shellfish production areas of the Solent (from Williams & Davies, 2018a; Williams *et al.,* 2018).

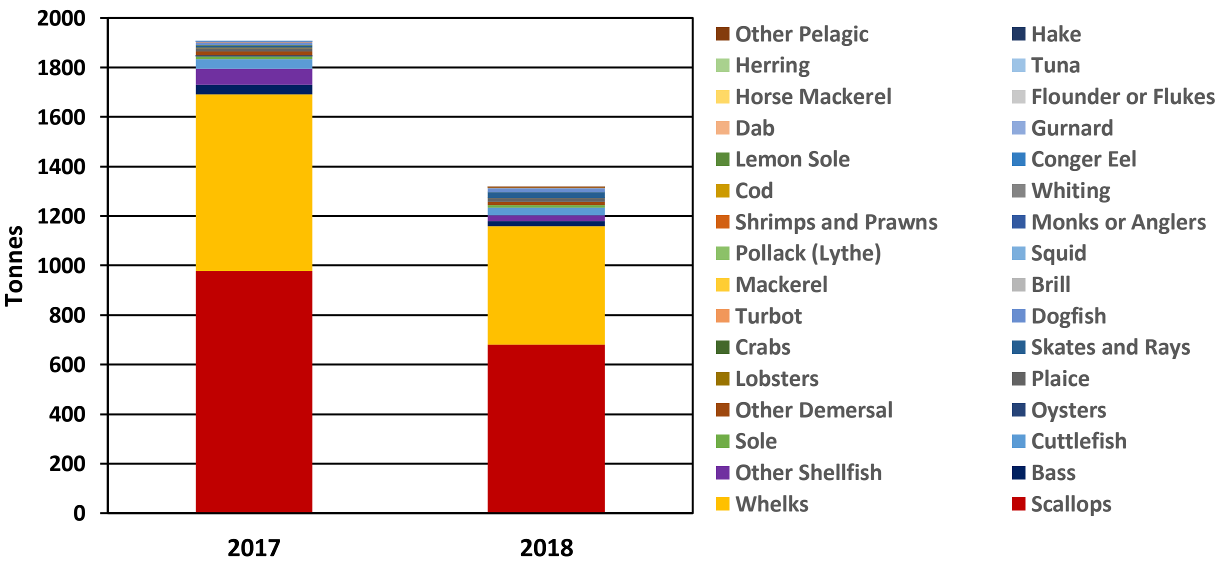
Solent total	**Scenario 1 - Do nothing**	**Scenario 2 - Improvement of all beds to class C (or above)**	**Scenario 3 - Improvement of all beds to class B**	**Scenario 4: Improvement of all beds to class A**	**Scenario 5 - Graded improvements**
Direct GVA	£0	£684,880	£705,099	£773,586	£757,468
Indirect GVA	£0	£1,027,806	£1,142,162	£1,403,609	£1,363,826
Total GVA	**£0**	**£1,712,686**	**£1,847,260**	**£2,177,195**	**£2,121,294**

Table 27 Total GVA of the four priority shellfish species considered in scenario 2 - Improvement of all beds to class C (or above) (from Williams & Davies, 2018a; Williams *et al.,* 2018).

	Manilla clam (*Tapes. spp*)	Hardshell clam (*M. mercenaria*)	Cockle (*C. edule*)	Native oyster (*O. edulis*)	Total
Direct GVA	£507,939	£58,497	£11,649	£106,795	£684,880
Indirect GVA	£332,126	£175,276	£71,393	£449,011	£1,027,806
Total GVA	**£840,065**	**£233,773**	**£83,042**	**£555,806**	**£1,712,686**

In addition, there are a number of harbours where finfish and other demersal species are landed in the Solent, although not all the fish and bivalves landed are from the Solent fishery. Portsmouth Harbour is the largest port for landings in the SEMS area (landing 1317 tonnes in 2018, worth £2.8 million; Figure 13). This was down from 1,900 tonnes landed in 2017 (worth £3.9 million; Figure 13).

Figure 13 Landings of finfish and demersal species into Portsmouth Harbour (MMO statistics)



For many of these species only a very small fraction will be taken from the Solent and at this stage this cannot be differentiated from local catches. Nonetheless, the landings data give an idea of the economic importance of the wider fishing industry to the Solent. As with the shellfish example above, the landing value of finfish and demersal species represents the direct GVA added to the local economy by the industry. Indirect GVA output was calculated through multiplying direct GVA by the '% sold domestically' and then applying an output multiplier, created by Seafish (a Non-Departmental Public Body (NDPB) which calculates the amount of economic change that occurs as a result of changes in an industrial sector. This information was taken from a recent study of finfish in Poole Harbour (Williams & Davies 2018b) and MMO annual sea fisheries statistics We combined this data then used data from 2017 and 2018 to focus on the top 10 species landed in Portsmouth Harbour, by taking an average landing value across the two years.

The information gathered informed the modelling of local economic impacts. Table 28 presents the local economic impact calculations for the top 10 species landed in Portsmouth Harbour. Total economic activity is estimated at **£7,835,330** and is calculated as the sum of direct GVA and indirect GVA.

Table 28: Local economic impact calculations for the top 10 species in Portsmouth Harbour

Species	Landing Value (£)	% sold domestically	Output multiplier (Seafish)	Direct GVA (£)	Indirect GVA (£)	Total economic activity (£)
Scallops	£1,941,404	50.00%	3.56	£1,941,404	£3,455,698	£5,397,102
Whelks	£708,549	5.00%	3.56	£708,549	£126,122	£834,671
Bass	£291,959	72.50%	3.91	£291,959	£827,632	£1,119,591
Cuttlefish	£121,058	5.00%	3.91	£121,058	£23,667	£144,725
Sole	£80,739	40.00%	3.91	£80,739	£126,276	£207,015
Plaice	£19,950	67.50%	3.91	£19,950	£52,653	£72,603
Skates and Rays	£16,317	25.00%	3.91	£16,317	£15,950	£32,266
Turbot	£6,144	25.00%	3.91	£6,144	£6,006	£12,150
Brill	£4,892	25.00%	3.91	£4,892	£4,782	£9,673
Crabs	£2,464	35.00%	3.56	£2,464	£3,070	£5,534
				£3,193,476	**£4,641,855**	**£7,835,330**

Table 29 brings together the economic activity values calculated for commercial shellfish, finfish and demersal species to produce a conservative value for the Solent of **£9,548,016.**

Table 29 Total economic activity for Solent fisheries. (N/A) Not assessed in this report

	Direct GVA	Indirect GVA	Total Economic Activity
Shellfish aquaculture in the Solent	£684,880	£1,027,806	£1,712,686
Commercial marine finfish and demersal species (Top 10 species Portsmouth only)	£3,193,476	£4,641,855	£7,835,330
Charter boat fleet	N/A	N/A	N/A
Freshwater and migratory fisheries	N/A	N/A	N/A
Inter-tidal and polychaete bait fisheries (*N. virens*) collected from the Solent and Poole Harbour region (Watson *et al.,* (2016a)	£4,300,000	N/A	£4,300,000
		Total	**£13,848,016**

It is important to note this is almost certainly an underestimate of the total economic activity provided by this provisioning ecosystem service. For example, recreational fishing (including charter boats and freshwater fishing) and the harvesting of marine invertebrates for fishing bait have often been excluded from NC assessments as these fisheries are often data-limited. While it was out of the scope of this report to investigate the direct and indirect GVA values of recreational and subsistence fishing, estimates from Watson *et al,* (2016a) suggest the harvesting value of the polychaete *Nereis (Alitta) virens*, could add as much as £4.3 million in additional value to the Solent region e.g. *via* direct retail value, with additional value through tackle shops and boat chartering revenues.

Further works that would be needed to derive a truly robust valuation of this ecosystem service are listed below:

- Total GVA estimates of commercial finfish and demersal species into smaller ports in the Solent (e.g. Langstone Harbour, Southampton Water and Harbours on the Isle of Wight).
- Estimates of recreational sea angling by individuals (fishing with a hook and a line for non-commercial purposes). Recent estimates from Poole Harbour suggest the benefits from chartered fishing could be substantial (estimated by Williams & Davies (2018b) at £5.7 Million per year).
- Total GVA estimates of migratory and freshwater species e.g. salmon and sea trout which make use of the Solent on their way to and from spawning grounds in the Rivers Test, Itchen, Meon and Hamble.
- Complete valuation estimates of inter-tidal and polychaete bait fisheries species (e.g. in addition to *Nereis (Alitta) virens*.

6.3 Nursery Function and Supporting the Existence of Biodiversity.

Many biotopes in the SEMS are highly productive and provide nursery, feeding, and spawning grounds for commercially and ecologically important fish, shellfish, and birds. In an ecosystem service context, several conceptual discrepancies and empirical challenges have arisen when trying to quantify this service. The main reasons behind this are that, on the one hand, this ecosystem service could be interlinked or correlated with other services that directly rely on it (e.g. fisheries) and, on the other hand, it refers to biodiversity components and ecosystem functions (i.e. nursery function). In the context of this report, we follow a recently published report by Liquete *et al.,* (2016) which concludes that *"nursery function should be considered as an ecosystem service in its own right when linked to concrete human benefits, not when it represents general biodiversity or ecosystem condition".* With reference to the EUNIS classification scheme, the biotopes mainly providing this service are coastal saltmarshes, reedbeds, seagrass and infralittoral rock, although all sediment biotopes are recognised as important (Table 30).

Therefore, to aid in the development of habitat conservation strategies by environmental resource managers, other decisionmakers, and stakeholders, we present an approach for evaluating the aggregate importance of different biotopes for species of interest and apply the approach to the coastal fisheries of the Solent. A representative set of 44 fish taxa, and another seven commercially or ecologically important species that use the coastal zones of the SEMS were selected for each region based on available presence/absence records from various data sources (see Appendix Figure S3). The most robust estimates of young and juvenile fish were based on IFCA small fish survey assessments (2015-2018) representing six harbours of the SEMS (Yarmouth, Newton, Medina, Langstone, Chichester and Pagham). For the remaining estuaries a combination of data held in the NBA marine atlas and CEFAS Young Fish Survey data (2001-2010) were used as a proxy for presence/absence of particular species in that region. It should be noted that the latter approach is likely biased towards larger and more commercially important species and does not cover many of the smaller fish species (e.g. gobies, blennies etc.) that would be sampled using the finer resolution sampling practices of the IFCA small fish surveys.

Table 30 Matrix assessment of nursery habitat function in SEMS, including biotopes features of MPAs. N/A (not assessed) building on Potts *et al.*, 2014)

Natural Capital Stocks: Biotopes in the Solent Marine Site	Area (ha)	Area in MPAs (ha) pre-Jan 2020	Nursery function and supporting biodiversity Supporting services Formation of species habitat
A2.11: Shingle (pebble) and gravel shores	708	708	1
A2.3: Littoral mud	6204	6204	1
A2.3: Littoral mud (with mat forming macroalgae)	1616	1616	1
A2.5: Coastal saltmarshes and saline reedbeds	1261	1111	3
A3: Infralittoral rock and other hard substrata	756	345	2
A5: Sublittoral sediment	1438	215	2
A5.1: Sublittoral coarse sediment	9496	3914	1
A5.2: Sublittoral sand	10088	2817	3
A5.3: Sublittoral mud	7502	5516	3
A5.34: Infralittoral fine mud	400	344	3
A5.4: Sublittoral mixed sediments	1497	288	3
A5.435: Ostrea edulis beds on shallow sublittoral muddy mixed sediment	2839	1155	1
A5.52: Kelp and seaweed communities on sublittoral sediment	121	121	1
A5.53: Sublittoral seagrass beds and A2.61: Seagrass beds on littoral sediments	698	691	3
B1.21: Unvegetated sand beaches above the driftline	88	11	
B2: Coastal shingle	136	15	
B2.2: Unvegetated mobile shingle beaches above the driftline	251	28	
B3: Rock cliffs, ledges and shores, including the supralittoral	40	2	
C3.21: *Phragmites australis* beds	273	226	3
C2.3: Permanent non-tidal, smooth-flowing watercourses	25	5	

Scale of ecosystem service supplied relative to other features

#	Significant contribution
#	Moderate contribution
#	Low contribution
#	No or negligible ESP
	Not assessed

Confidence in evidence

3	UK-related, peer-reviewed literature
2	Grey or overseas literature
1	Expert opinion
	Not assessed

We evaluated biotope use of commercially important fish species and invertebrates by examining four different ecological biotope functions:

(i) Spawning: records of ripe adults, observation of spawning, or the presence of newly spawned eggs;

(ii) Nursery: reference to the concentration of juvenile stages or at least the presence of juveniles;

(iii) Feeding: the use of habitats by adults as feeding grounds or at least the presence of adults not related to spawning; and

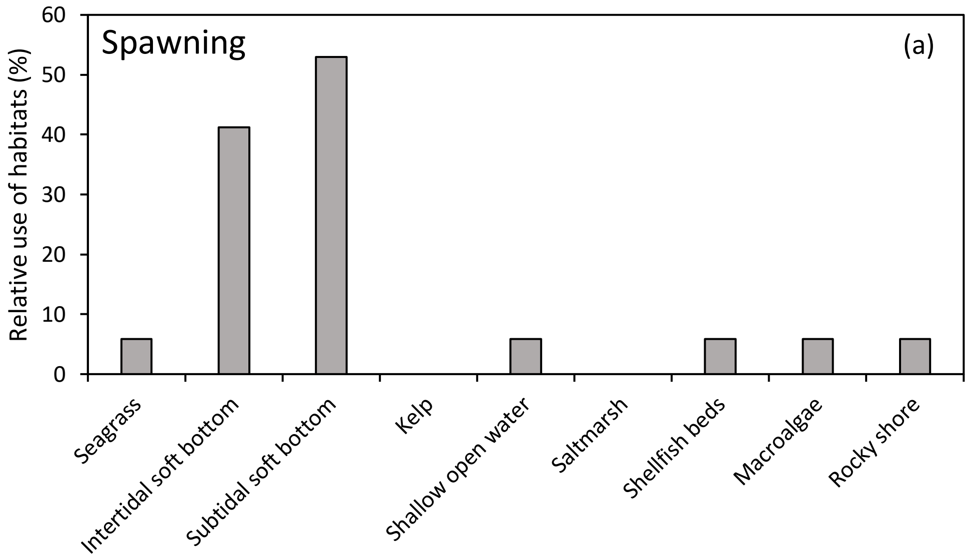
(iv) Migration: mainly refers to the directional movement of diadromous species.

The categorization was based on the methodology used by Seitz *et al.,* (2014) with our conclusions referring to the biotope definitions of the four nursery functions identified in table 30. There was insufficient published information available to resolve this information for non-commercial species and/or quantify the likely production benefit afforded by individual biotopes for individual species (e.g. how much juvenile bass biomass might be produced or enhanced by an additional ha of seagrass). Therefore, a non- monetary approach to valuation was considered for this ecosystem service. To achieve this, we summed all scores for each biotope category and biotope function to gauge the overall relative importance as living space for the species of interest.

Our results suggest that representatives of the Solent species (n=17) utilized most habitat that we investigated (Figure 14), with the exception of kelp and saltmarsh as a spawning area (Figure 14a). Subtidal soft bottom was the biotope used as spawning, feeding and nursery areas by the largest proportion of species, closely followed by intertidal soft bottom was also used heavily as nursery ground (71% of species). Subtidal soft bottom biotope was particularly important for young bass which is an important commercial and recreational fishing target in Europe, and is protected in several regions of the SEMS by jurisdictional Bass nursery areas. Rocky shore, saltmarsh, seagrass and shellfish beds were also used by many species for feeding and as a nursery ground (Figure 14 b,c).This included sandeels (Appendix Figure S3) which have repeatedly been noted in the IFCA small fish surveys to be one of the most commonly occurring small fish in several regions (sometimes over 85% of small fish caught in some regions e.g. Langstone). Shallow open water biotopes were the only habitat used as a migratory function by the European eel (Figure 14d).

Unfortunately, for most species, there was inadequate information to judge the degree to which these coastal biotopes limit population growth and fishery production under different water quality scenarios. There is an obvious lack of information on how fish utilize some biotope types in the SEMS, particularly complex hard-bottom habitats such as kelp forests, rocky shores, and macroalgae, where many census techniques are inadequate. One recommendation is to focus future studies on these biotope types to attain quantitative data on fish (both population- and individual-level data) and their dependence on these biotopes. It is also clear from our analysis that many non-commercially important species in the Solent utilize coastal biotopes. For most species, however, there is insufficient information to judge whether these coastal biotopes (or non-coastal habitats used during other parts of the life cycle) are actually essential and limiting to population growth and fishery production. Further studies are needed to attain quantitative data on coastal biotope use by fish and invertebrates to aid the definition of key biotopes for protection and restoration efforts and to integrate biotope quality in stock assessment and ecosystem-based fishery management. This should include but not be limited to the following:

- Non-commercial marine fish
- All freshwater and anadromous fish (e.g. salmon, trout spp., river lamprey)
- Lobster and brown crab with spider crab
- Cuttlefish and seahorses
- Conger Eel (*Conger oceanicus*)
- Seals and other marine mammals
- Skates and rays
- Thresher shark and basking sharks
- Wading birds and wildfowl



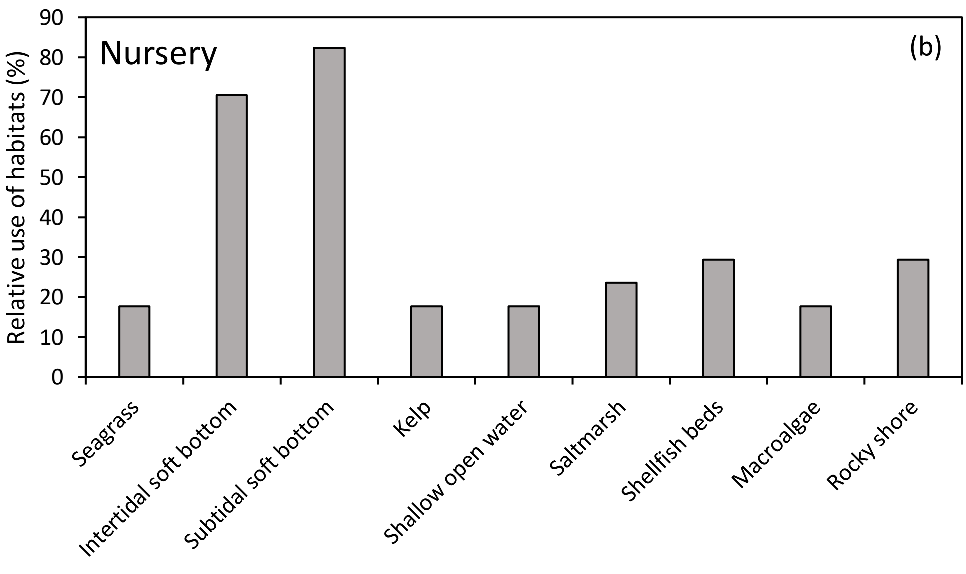


Figure 14 Percentage (%) of Solent fish or other ecologically important species using coastal habitats for (a) spawning, (b) nursery grounds, (c) feeding and (d) migration.

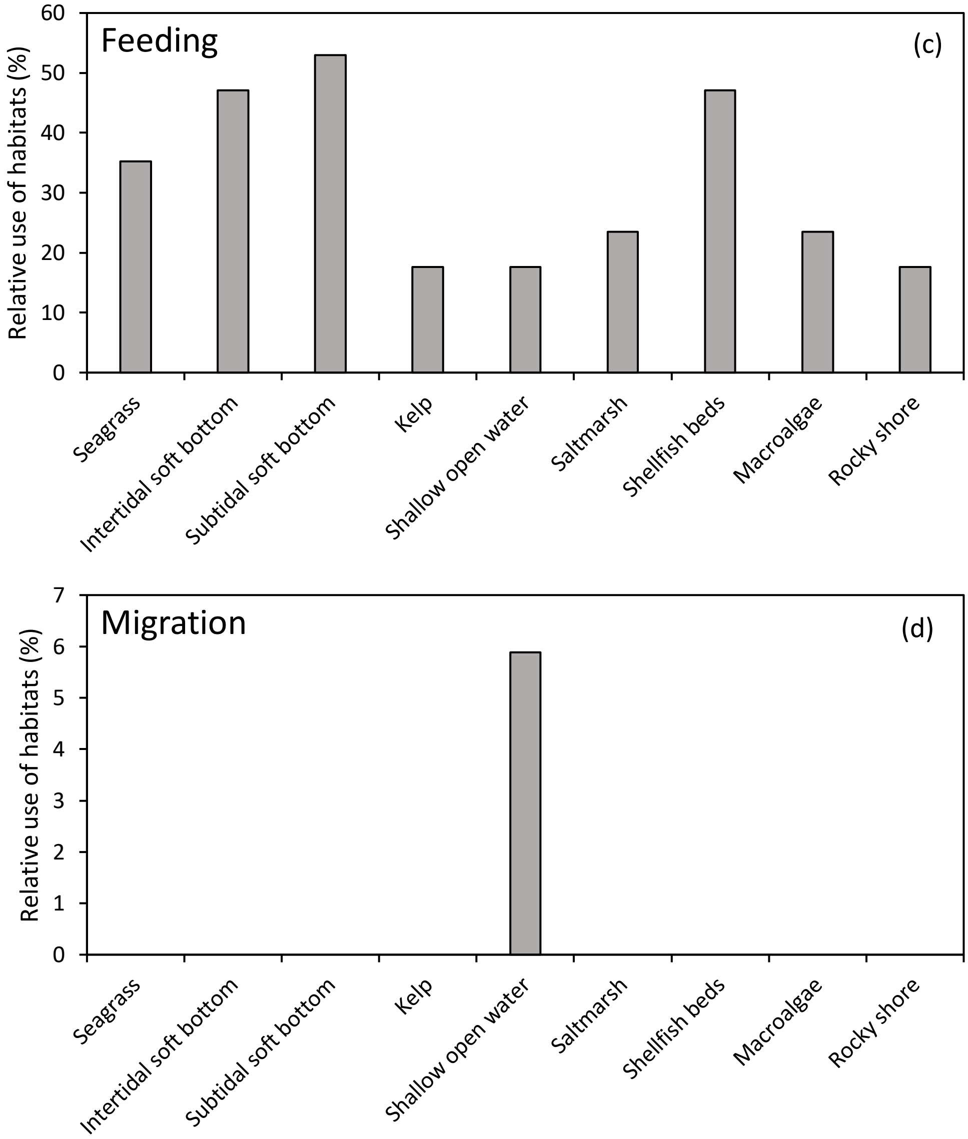
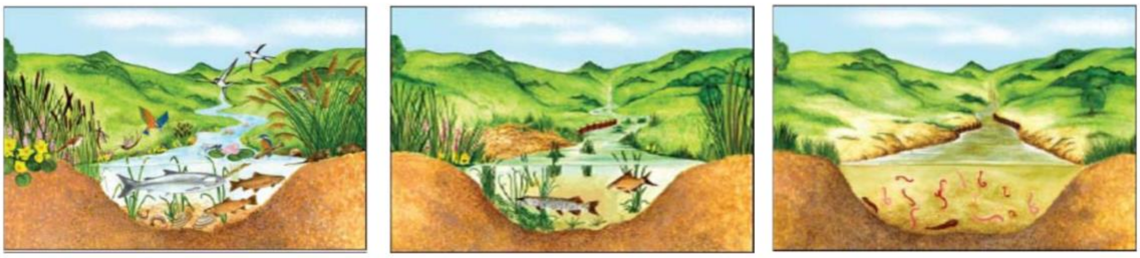


Figure 14 (continued) Percentage (%) of Solent fish or other ecologically important species using coastal habitats for (a) spawning, (b) nursery grounds, (c) feeding and (d) migration.

6.3 Recreation, Tourism and Leisure

Coastal and marine environments provide numerous recreational opportunities, e.g. walking, sport activities, nature and wildlife watching, sea angling, boating activities. Economic valuation methods suitable for assessing the benefits associated with these activities can be broadly divided into revealed preferences and stated preferences approaches (Pascual, *et al.,* 2010). The application of both these approaches is often limited by data availability, as they require statistical modelling and primary valuation studies. Updated National Water Environment Benefits Survey (NWEBS) values have been acknowledged by Defra and Environment Agency as providing the best and most practical way to use the currently available evidence on monetary values for non-market cultural benefits for implementation under the WFD and *de facto* are useful when considering water quality goals. The NWEBS values capture society's 'willingness to pay' for recreational services (such as water sports), aesthetic services and existence values but, do not include estimates of improvements to provisioning and regulating services. In order to illicit the NWEBS values, a survey of over 1500 people in the UK was undertaken. This work, is described in more detail in 'NWEBS briefing note' (Metcalfe, 2013) and 'Enabling a Natural Capital Approach: Guidance" (DEFRA 2020).

Respondents to the 2007 questionnaire considered changes to portions of rivers at 'high', 'medium' and 'low' quality nationally and locally using illustrations (Figure 15) of coastal areas which included variations on the following six components: fish, other animals such as invertebrates, plant communities, the clarity of water, the condition of the waterbody and flow of water, and the safety of the water for recreational contact. To derive monetary values for the ecosystem service benefits the WTP values are equally divided across the six components listed above and applied per km of water body improved within a catchment. Values per km are provided for South West WFD catchments of changes in quality from bad to poor, poor to moderate, and moderate to good (Table 31). Low, central and high annual estimates (£) are provided. Using the South West "Moderate to Good" WTP values as a proxy, we combine this data with the area of each SEMS water body (km) in need of improvement (Table 32).



High Medium Low

Figure 15 National Water Environment Benefits Survey (NWEBS): 2007 questionnaire illustrations

Table 31 National Water Environment Benefits Survey (NWEBS): Annual per km^2 values, £000's, 2012 prices, for South West coastal and transitional waters

Bad to Poor			Poor to Moderate			Moderate to Good		
Low	Central	High	Low	Central	High	Low	Central	High
£5.7	£6.9	£8.1	£6.5	£8	£9.4	£7.6	£9.3	£10.9

The annual benefits for each option are estimated as the benefits transfer value multiplied by the length of water body that will benefit. The results suggest that willingness to pay estimates for improving water quality across the whole Solent region to be approximately **£3,176,649 yr^{-1}**(Table 32). This is based on the assumption that an improvement maintains the water body status at good; otherwise no value would be available and these benefits would not be monetised. The central values have been used in the main appraisal; the low and high values can be used in sensitivity analysis. Household population information provided by the EA for each catchment could also be used to convert the NWEBS catchment values to £/km/hh/year terms but this is not explored here. Table 32 is somewhat limited, since it does not control for various factors that influence water environment valuations (e.g. scope of improvement, availability of substitutes, characteristics of beneficiary populations), nor the type of benefits that would be seen in the water body. Nonetheless, the NWEBS values are among the best available values on which to base estimates of benefits to cultural ecosystem services *via* improvements in water quality and have been used here to reflect the magnitude of change due to restore, improve or maintain specific catchments. The calculations take account of the current overall status of each water body so this should reduce the potential for over-estimation of benefits.

Table 32 Solent willingness to pay' estimates for improving or maintaining water quality in all WFD water bodies, from moderate to good condition based on recreational services, aesthetic services and existence values. Southampton Water estimates include the Hamble Estuary.

WFD Catchment	Current overall WFD 2016 status	WFD Area (km²)	Low WTP (£) To "Good" Condition	Central WTP (£) To "Good" Condition	High WTP (£) To "Good" Condition
Lymington Estuary	Moderate	2.45	£18,637	£22,806	£26,730
Beaulieu Estuary	Moderate	3.07	£23,370	£28,597	£33,517
Southampton Water*	Moderate	30.91	£234,940	£287,493	£336,954
Portsmouth Harbour	Moderate	16.42	£124,827	£152,748	£179,028
Langstone Harbour	Moderate	18.91	£143,697	£175,840	£206,092
Chichester Harbour	Moderate	2.78	£21,156	£25,888	£30,342
Pagham Harbour	Moderate	2.57	£19,550	£23,923	£28,039
Yar Estuary	Moderate	0.51	£3,878	£4,746	£5,562
Newton Harbour	Moderate	1.92	£14,575	£17,835	£20,903
Medina Estuary	Moderate	1.63	£12,366	£15,131	£17,735
Bembridge Harbour	Moderate	0.81	£6,161	£7,539	£8,836
Solent (open water)	Moderate	259.58	£1,972,815	£2,414,102	£2,829,432
		Total	**£2,595,971**	**£3,176,649**	**£3,723,169**

6.5 Summary of Water Quality Improvements: Physical and Monetary Ecosystem Services Accounts in the Solent Marine Site

In the following section the results obtained in Part A and B of this report regarding the calculation of ecosystem services using physical and monetary accounts will be summarised, to provide estimates of the financial and societal value of natural resources to people in the Solent. Tables 33 and 34 show the basic structure of possible physical and monetary use tables for the Solent marine and coastal environments following the guidance of building the ONS UK natural capital accounts (ONS 2017). We also extend the present extent and condition physical and monetary estimates, with forecasts of the potential uplifts in ecosystem service supply provided if the overall WFD waterbody classifications transitioned to "Good" or "High" status reflecting potential future improvements in water quality and biotope condition. The results obtained demonstrate that better water quality leads to a higher gross and indirect output as a result of the increases in the condition of natural capital stocks (i.e. biotopes). The total monetary benefits accrued under "good status" water quality conditions are estimated as **£1,601.82 Million**, approximately a **£297.45 Million yr^{-1}** increase on current water quality conditions. The increase in total economic activity is notably larger between moderate ('the status quo') and "high" status conditions (**£2,116.99 Million**), with an increase of approximately **£812.62 Million yr^{-1}.** The conclusion from the research is clear that restoring and improving the existing condition of biotopes should be seen as a major consideration for management in the Solent. Although the findings are not presented in terms of a cost-benefit ratio, the estimates provided here have the capacity to do so should those cost estimates become available in the future.

Table 33 SEMS marine and coastal ecosystem services physical account. (N/A not assessed). *Only shellfish uplift due to improvements in water quality included

Ecosystem service	EUNIS Habitat	Current WFD Status- 2016	Future scenario if all biotopes transition to "Good" WFD condition	Future scenario if all biotopes transition to "High" WFD condition
Waste remediation (tonnes nitrogen stored yr^{-1})	(A.23, A2.4) Littoral sediments, Littoral sediments (with macroalgae), (A5.2, A5.3, A5.4) Subtidal sediments, (A2.5) Saltmarsh, (A5.53, A5.545, A2.61) Seagrass, (C3.2, C32.1) Reedbeds, (A5.435) Native oyster (*Ostrea edulis*) reefs.	3590	4487	5712
Waste remediation (t phosphorus stored yr^{-1})	(A.23, A2.4) Littoral sediments, Littoral sediments (with macroalgae), (A5.2, A5.3, A5.4) Subtidal sediments, (A2.5) Saltmarsh, (A5.53, A5.545, A2.61) Seagrass, (C3.2, C32.1) Reedbeds, (A5.435) Native oyster (*Ostrea edulis*) reefs.	811	915	1451
Climate regulation (t carbon stored yr^{-1})	(A.23, A2.4) Littoral sediments, Littoral sediments (with macroalgae), (A5.2, A5.3, A5.4) Subtidal sediments, (A2.5) Saltmarsh, (A5.53, A5.545, A2.61) Seagrass, (C3.2, C32.1) Reedbeds, (A5.435) Native oyster (Ostrea edulis) reefs.	38299	49449	81216
Commercial, recreational and subsistence fisheries. (t catch yr^{-1})	(X01): Estuaries, (A5.435) Native oyster (*Ostrea edulis*) reefs.	1693 (Finfish and shellfish) 130 (polychaete biomass)	1862* (Finfish and shellfish) 130 (polychaete biomass)	2032* (Finfish and shellfish) 130 (polychaete biomass)
Nursery function and supporting the existence of biodiversity. (Commercially important fish taxa or species)	(A.23, A2.4) Littoral sediments, Littoral sediments (with macroalgae), (A5.2, A5.3, A5.4) Subtidal sediments, (A2.5) Saltmarsh, (A5.53, A5.545, A2.61) Seagrass, (C3.2, C32.1), (A5.52) Kelp beds, Reedbeds, (A5.435) Native oyster (*Ostrea edulis*) reefs.	51 species	N/A	N/A
Recreation, leisure and tourism (km of water body available for marine recreation, leisure or tourism)	(X01) Estuaries (X03) Brackish coastal lagoons.	341.56	341.56	341.56

Table 34 SEMS marine and coastal ecosystem services physical monetary account. (N/A not assessed). *Only shellfish uplift due to improvements in water quality included. (M =Million)

Ecosystem service	EUNIS Habitat	Current WFD Status- 2016	Future scenario if all biotopes transition to "Good" WFD condition	Future scenario if all biotopes transition to "High" WFD condition
Waste remediation (nitrogen)	(A.23, A2.4) Littoral sediments, Littoral sediments (with macroalgae), (A5.2, A5.3, A5.4) Subtidal sediments, (A2.5) Saltmarsh, (A5.53, A5.545, A2.61) Seagrass, (C3.2, C32.1) Reedbeds, (A5.435) Native oyster (*Ostrea edulis*) reefs.	£1,059.16 M	£1,323.67 M	£1,685.04 M
Waste remediation (phosphorus)	(A.23, A2.4) Littoral sediments, Littoral sediments (with macroalgae), (A5.2, A5.3, A5.4) Subtidal sediments, (A2.5) Saltmarsh, (A5.53, A5.545, A2.61) Seagrass, (C3.2, C32.1) Reedbeds, (A5.435) Native oyster (*Ostrea edulis*) reefs.	£228.63 M	£258.03 M	£409.18 M
Climate regulation (carbon)	(A.23, A2.4) Littoral sediments, Littoral sediments (with macroalgae), (A5.2, A5.3, A5.4) Subtidal sediments, (A2.5) Saltmarsh, (A5.53, A5.545, A2.61) Seagrass, (C3.2, C32.1) Reedbeds, (A5.435) Native oyster (Ostrea edulis) reefs.	£2.30 M	£2.97 M	£4.87 M
Commercial, recreational and subsistence fisheries	(X01): Estuaries, (A5.435) Native oyster (*Ostrea edulis*) reefs.	£13.85 M	£13.98 M	£14.31 M
Nursery function and supporting the existence of biodiversity.	(A.23, A2.4) Littoral sediments, Littoral sediments (with macroalgae), (A5.2, A5.3, A5.4) Subtidal sediments, (A2.5) Saltmarsh, (A5.53, A5.545, A2.61) Seagrass, (C3.2, C32.1), (A5.52) Kelp beds, Reedbeds, (A5.435) Native oyster (*Ostrea edulis*) reefs.	N/A	N/A	N/A
Recreation, leisure and tourism	(X01) Estuaries (X03) Brackish coastal lagoons.	£2.73 M	£3.18 M	£3.59 M
	Total	**£1,304.38 M**	**£1,601.82 M**	**£2,116.99 M**

Part C: Assessing Multiple Stressors, and Impacts on Ecosystem Services Using a Biotope Sensitivity Approach

This section outlines the general principles that have allowed benthic biotope sensitivities and their risk of impact to be used to estimate a change in the level of ecosystem service as a result of a change in a stressor. By stressor we mean any natural or anthropogenic conditions that places stress on the health and functioning of an organism, population and/or ecosystem (akin to 'pressure' *sensu* Baird *et al.,* 2016) exerted on benthic biotopes. The methodology that has been developed with input from both the Project Steering Group (PSG) and the "Marine Pioneer Lessons Learned workshop" that was held on December 2019. More detail regarding specific aspects of the spatial model development per stressor is presented in Section 7.

7.1: Overall Methodology

In simple terms, a spatial model of marine ecosystem services can be conceived in terms of two components (Barbier 2012):

- A baseline assessment which seeks to provide a current map of biotopes and ecosystem service provision (i.e. Part A and B of this report); and
- A dynamic model which can represent changes in ecosystem service provision as a result of changes in human (or natural) stressors in space and time (Part C this report).

This project has thus far has focused on the changes in ecosystem service provision as a result of a ***positive*** change in the condition of benthic biotopes. The final section of this report will assess potential ***negative*** changes in the level of ecosystem service(s) that could result from changes in the quality of benthic biotopes, in the context of current and future anthropogenic stressors. We considered biotope impacts associated with four important stressors — physical abrasion from mobile fishing gears, introduction of microbial pathogens (*Escherichia coli*), increase in N or P inputs (eutrophication) and intertidal biotope loss due to sea level rise (coastal squeeze) — that directly or indirectly impact on water quality-related ecosystem services, both individually and cumulatively. The subsequent sections outline the methodology to develop the dynamic model, in a series of discrete steps.

7.2: Model Development

Step 1 - Identify ecosystem services

During the development of the cumulative marine ecosystem services assessment, the list of ecosystem services was edited further to include only the regulating services responsible for removing N, P and C. This took into account considerations of relevance to change in the condition of benthic biotopes as well as data availability and levels of uncertainty. The area of impacted biotopes linked to the ecosystem service of nursery function were also included but, were not considered in a monetary sense. This has resulted in a total of four ecosystem services for consideration:

- Waste remediation (nitrogen)
- Waste remediation (phosphorous)
- Climate regulation (carbon sequestration and storage)
- Nursery function and supporting the existence of biodiversity.

Step 2 – Biotope mapping and attribute sensitivities

The previously mapped biotype data (Figure 5) for the SEMS region were compiled according to the EUNIS system. As the present approach pertains only to sedimentary biotopes, all coastal biotopes above spring high tide limit (including rocky biotopes) were excluded from the analysis. Sensitivity information for the remaining EUNIS biotopes was extracted from the Marine Evidence-based Sensitivity Assessment (MarESA) database (Tyler-Walters *et al.,* 2019). MarESA compiles sensitivity information through a detailed literature review process of available evidence on the effects of stressors arising from human activities on marine habitats.

Table 35 Sensitivity of biotopes to selected stressors by applying a precautionary approach to link to the MarESA database to the EUNIS classification system. "Contains data provided by the MarLIN programme (www.marlin.ac.uk), the Marine Biological Association of the United Kingdom © copyright and database right 2018".

EUNIS code	Substrate	Abrasion	Introduction of microbial pathogens (*Escherichia coli*)	Nutrient enrichment	Emergence regime changes (Coastal squeeze/Sea level rise)
A2.11	Shingle (pebble) and gravel shores	Not sensitive	Not relevant (NR)	Not sensitive	Not sensitive
A2.3	Littoral mud	Low	Not sensitive	Not sensitive	Low
A2.3	Littoral mud (with mat forming macroalgae)	Low	Not sensitive	Not sensitive	Low
A2.5	Coastal saltmarshes and saline reedbeds	Low	No information	Low	Medium
A3	Rock or other hard substrata	Low	Not sensitive	Not sensitive	Not sensitive
A5	Sublittoral sediment	Low	Not sensitive	Not sensitive	Not sensitive
A5.12	Sublittoral coarse sediment in variable salinity (estuaries)	Not sensitive	Not sensitive	Not sensitive	Not sensitive
A5.2	Sublittoral sand	Low	Not sensitive	Not sensitive	Not sensitive
A5.3	Sublittoral mud	Medium	Not sensitive	Not sensitive	Not sensitive
A5.34	Infralittoral fine mud	Medium	Not sensitive	Not sensitive	Low
A5.4	Sublittoral mixed sediments	Medium	Not sensitive	Not sensitive	Not sensitive
A5.435	*Ostrea edulis* beds on shallow sublittoral muddy mixed sediment	High	High	Not sensitive	Not sensitive
A5.52	Kelp and seaweed communities on sublittoral sediment	Medium	Not sensitive	Not sensitive	Not sensitive
A5.53, A5.545, A2.61	Sublittoral seagrass beds and seagrass beds on littoral sediments	Medium	Not sensitive	Medium	Not sensitive (Seagrass beds on littoral sediments Medium)
B1.21	Unvegetated sand beaches above the driftline	Not sensitive	Not relevant (NR)	Not sensitive	Not sensitive
B2	Coastal shingle	Not sensitive	Not relevant (NR)	Not sensitive	Not sensitive
B2.2	Unvegetated mobile shingle beaches above the driftline	Not sensitive	Not relevant (NR)	Not sensitive	Not sensitive
B3	Rock cliffs, ledges and shores, including the supralittoral	Medium	Not relevant (NR)	No evidence (NEv)	Not sensitive
C3.21	*Phragmites australis* beds	Low	No information	Not sensitive	Medium
C2.3	Permanent non-tidal, smooth-flowing watercourses	No evidence (NEv)	No evidence (NEv)	No evidence (NEv)	Not sensitive

The assessments assign scores for habitat (biotope) sensitivity as a combination of resistance and resilience to particular stressors (see Table 35). The scores allocated are: Not Sensitive (NS), Low (L), Medium (M), High (H) and Not relevant (NR). These were coded numerically in Arc GIS (v10.7) and linked to the SEMS biotope data layer through a series of iterative joins, linking sensitivity information based on the most detailed biotope class information available (EUNIS levels 5 and 6), up to EUNIS level 3. At the higher EUNIS levels (3 and 4), MarESA assessments were aggregated, taking advantage of EUNIS' hierarchical structure and following a precautionary approach to assign the most sensitive score of all 'children' classes from existing MarESA assessments to their 'parent' class (e.g Rees *et al.,* 2019).

Step 3- Stressor models and scenarios

In order to address potential recovery of the ecosystem state when stressors are reduced, we developed a simple scenario-based framework. This builds on two methods: 1) development of a spatial model to predict the change in the level of ecosystem service provision as a result of physical abrasion from mobile fishing gears and 2) quantitative risk assessment scenarios, to test the effects of sea-level rise, nutrient and pathogen inputs on the current condition of the biotopes.

Approach 1 – spatial model (sediment abrasion)

A stressor layer representing surface sediment abrasion caused by bottom fishing activities in the Solent has been generated using guidance provided by (ICES 2016). Fishing vessel positional monitoring system (VMS) data based on the 2009-2012 UK Vessel Monitoring System (VMS) data for vessels under 15m (Vanstaen & Breen, 2014) and VMS data for vessels 15m and over for the 2016 period (MMO 2018) was used as a basis for the layer. The use of the <15m vessel 2009-2012 (VMS) dataset (as opposed to more recent data) facilitated the integration with inshore sightings data for England and Wales which is an amalgamation of data collected between 2007 and 2009 provided by Cefas as an output of MB0106 (Lee & Rogers 2010). The datalayer essentially forms a grid of fishing effort at the scale equal to a 200th of an ICES rectangle (equivalent to between 10km^2 to 22km^2), with each cell assigned an effort value according to number of hours fished within that cell. A limitation of this approach is that it only represents a snapshot of inshore fishing activity which can vary in location and intensity from year to year.

The fishing effort (in hours) per grid cell has also been classified according to four gear types in cells:

- Dredging
- Trawling
- Potting
- Netting

Exposure thresholds set for each effort type were then applied to the map symbology for each grid cell based on the methodology outlined by Enever *et al.,*(2017) who classified their dataset into low, medium or high exposure according to the four relative levels of fishing effort throughout English waters, based on quartiles of vessel counts per square nautical mile. This was then integrated with the VMS data to create a single surface abrasion stressor data layer (Figure 16). To enable direct comparison with other stressors the sediment abrasion stressor layer was combined spatially with the biotope sensitivity information (Figure 17). Combinations of sensitivity and exposure levels (Table 36) were then used to indicate the likely impacts to benthic biotopes, and their likely relative condition as a result. Biotopes protected under the following Southern IFCA Byelaws: Bottom Towed Fishing Gear Byelaw 16 (Figure S4), Prohibition of Gathering (Sea Fisheries Resources) in Seagrass Beds (Figure S5) and Sussex IFCA bylaws :Chichester Harbour Marine Site (specified Areas) Prohibition of Fishing Method Byelaw were also excluded from further analysis, due to all fishing being prohibited. The condition of the biotope then determines the level ecosystem service that the biotope is capable of providing (Table 37). To represent N, P and C ecosystem service provisioning under impacted conditions we used the previously calculated lower (Q1) quartile and minimum biophysical values for each impacted biotope (see Appendix Table S5 &6). A summary of the different classifications used in this analysis to represent condition are given in (Section 3.6 Table 7). The potential loss in economic activity from abrasion impacts were then derived using previously estimate "median" nutrient replacement costs and mid UK non-traded carbon prices (see sections 4.1 & 6.1)

Table 36 Combination matrix for Impacts due to habitats sensitivity and stressor exposure, and inferred likely condition of the benthic biotopes (adapted from Rees *et al.,* 2019).

Sensitivity	Exposure			
	None	Low	Moderate	High
NS	None	None	None	None
L	None	Low	Low	Moderate
M	None	Low	Moderate	High
H	None	Moderate	High	Very High

=

Sensitivity	Exposure			
	None	Low	Moderate	High
NS	Undisturbed (High)	Undisturbed (High)	Undisturbed (High)	Undisturbed (High)
L	Undisturbed (High)	Slightly disturbed (Good)	Slightly disturbed (Good)	Moderately disturbed (Moderate)
M	Undisturbed (High)	Slightly disturbed (Good)	Moderately disturbed (Moderate)	Heavily disturbed (Poor)
H	Undisturbed (High)	Moderately disturbed (Moderate)	Heavily disturbed (Poor)	Extremely disturbed (Bad)

Table 37 Level of ecosystem service from biotopes in various condition states

Site disturbance classification	WFD Ecological status	Level of Ecosystem Service
Undisturbed	High status	Maximum
Slightly disturbed	Good status	Upper quartile
Moderately disturbed	Moderate status	Median
Heavily disturbed	Poor status	Lower quartile
Extremely disturbed	Bad status	Minimum

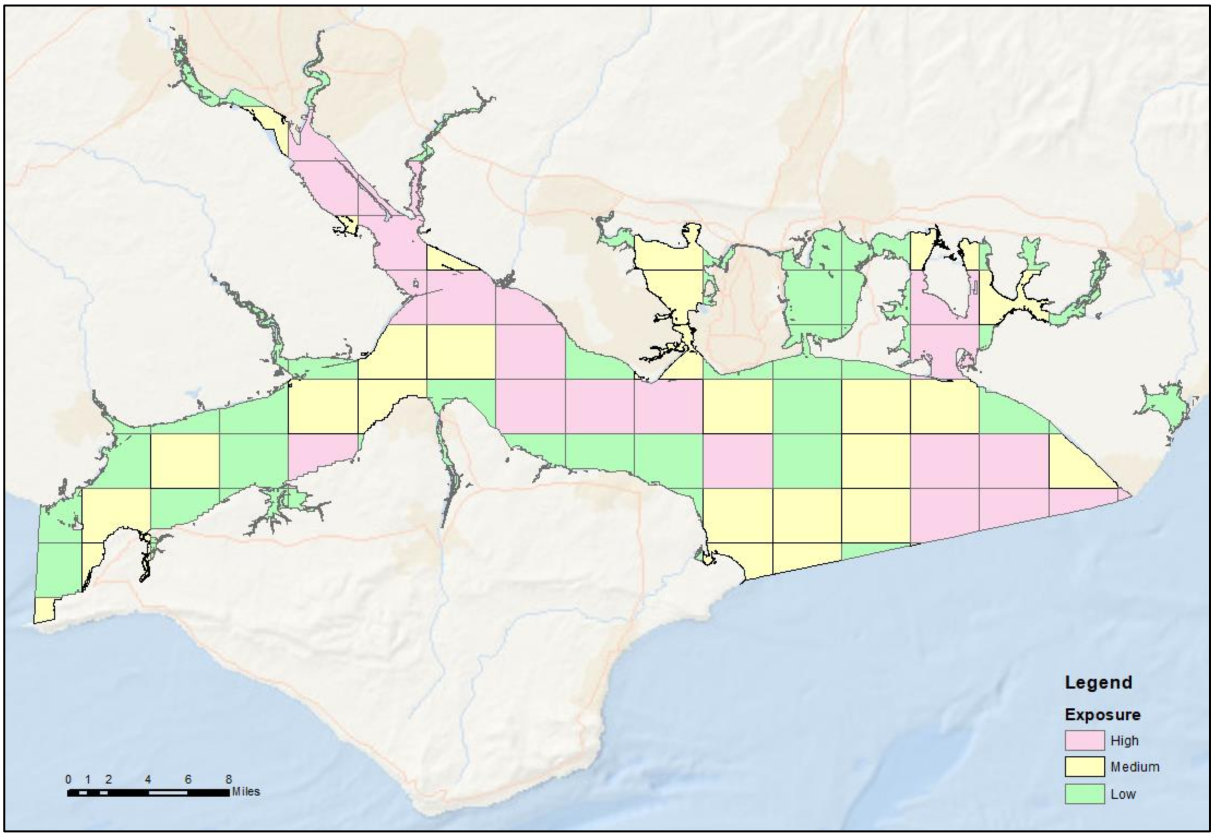


Figure 16 Intensity of abrasion as a function of exposure thresholds (High, Moderate and Low) by mobile gear type.

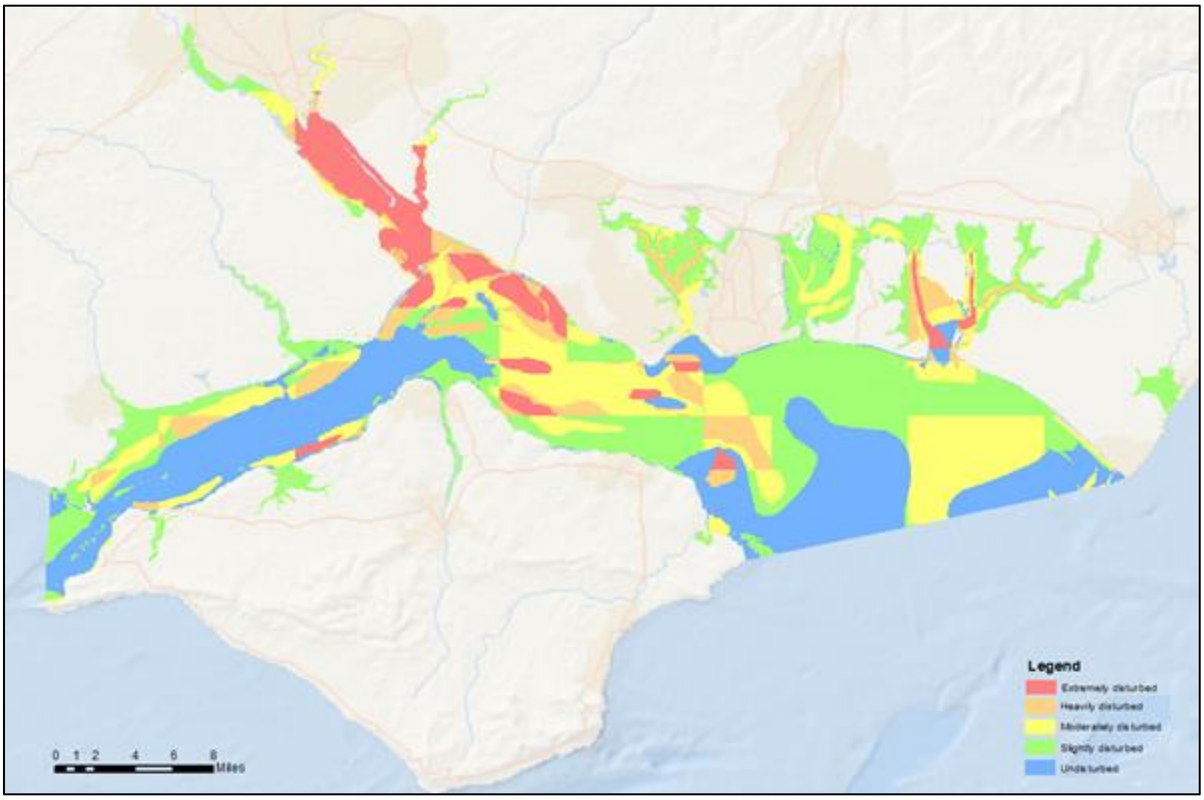


Figure 17 Likely relative condition due to impacts from abrasion, as inferred from the sensitivity-stressor approach

Approach 2 – Scenario analysis and expert judgement models (sea-level rise, nutrient and pathogen inputs)

Management options that cover a wide range of environmental factors, dynamic processes and interactions, geographical areas, and varying temporal scales are rarely quantitatively evaluated by one, coordinated research model (Uusitalo *et al.,* 2016). In these cases, expert judgement models can be used to help in risk decision making. Therefore, in order to get a compilation of the current best understanding of the whole-system responses to the selected stressors, we set up a series of scenarios representing an increase of nutrient inputs, pathogen inputs and coastal erosion impacts due to sea level rise. The basis of nutrient and pathogen scenarios was to compare an increase in the stressors to the point where MarESA "sensitive biotopes" where reduced to "poor" WFD (or class FSA prohibited) condition status. Using this framework, the level of ecosystem service provision made by each respective biotope was adjusted in a hierarchical process based on WFD and FSA condition data. The potential losses in economic activity from nutrient and pathogen impacts were then derived using the decision rules outlined in Table 37 and the previously estimated "median" nutrient replacement costs and mid UK non-traded carbon prices (see sections 4.1 & 6.1)

To create scenarios of biotope loss due to coastal squeeze and saline inundation impacts, we used previously calculated predicted biotope losses within the Solent Maine site, using modelled data (LiDAR flooding) from the Solent Dynamic Coast Project [SDCP] (Cope *et al.,* 2008). Habitats investigated by SDCP were set out using the following "EUNIS biotope groups" so ensuring consistent calculation of biotope losses and gains:

- Intertidal Mudflats (EUNIS A2.3)
- Intertidal Saltmarsh (EUNIS A2.5)
- Reedbeds (EUNIS C3.21)

The SDCP calculated biotope losses within the SEMS over 100 years. These estimates have subsequently been updated by the Regional Habitat Compensation Programme (RHCP) in the Solent and South Downs (SSD) Area (2020), which has identified the amount of biotope required to offset the adverse impacts to the Marine designated site(s) due to coastal squeeze and saline inundation impacts. The cumulative balance of biotopes across three epochs are summarised in table 37 below. By using this 'whole sites' approach — which considers biotope replacements and substitutions — it is possible to calculate the ***net*** future change in N, P and C ecosystem services values (£) resulting from sea level rise impacts, using the same method as described above for nitrogen and pathogen inputs. See Table 38 for a summary of the definitions of the scenarios in each case. It should be noted that the RHCP targets do not include restoration targets for other habitats e.g. seagrass, kelp, oysters.

Table 37 RHCP 2020 summary forecast biotope balance sheet. All values are rounded to the nearest hectare. (Updated February 2020 from the 2018 main report values see: https://southerncoastalgroup.org.uk/regional-habitat-creation-programme)

	Cumulative Biotope Balance (Ha)		
	Epoch 1	**Epoch 2**	**Epoch 3**
SDCP habitat group (EUNIS biotope group)	(2005 - 2025)	(2026 - 2055)	(2056 - 2105)
Intertidal Mudflats (EUNIS A2.3)	43	75	-32
Saltmarsh (EUNIS A2.3)	-20	-208	-392
Freshwater Habitats (EUNIS C3.21)	17	17	17

Table 38 The stressor scenarios implemented in the four evaluated models.

Stressor	EUNIS biotopes affected by stressor (based on MarESA sensitivity)	Scenarios implemented in the spatial and expert judgement models
Abrasion (mobile fishing gears)	A2.3, A2.5, A5, A5.2, A5.3, A5.34, A5.4, A5.435, A5.52, A5.53, A5.545, A2.61, C3.21	Abrasion/disturbance of the substrate on the surface of the seabed by current bottom fishing activities reduces the relative condition of biotopes.
Pathogen Input (*E. coli*)	A5.435	Increase of fecal indicator bacteria (*Escherichia coli* and enterococci) in shellfish-harvesting areas leading downgrading of all shellfish-harvesting areas.
Nutrient enrichment (N and P)	A2.5, A5.53, A5.545, A2.61	Increase of the limiting nutrient (nitrogen and/or phosphorus) loading from all sources (point and diffuse, all regions) leading to a downgrading of WFD water quality thresholds and standards for transitional catchments.
Climate change driven activities: Sea level rise/coastal squeeze (Emergence regime changes) including biotope net gain compensation.	A2.3, A2.5, C3.21	Requirements for replacement intertidal biotope as a result of tidal elevation and effects of coastal squeeze across the Solent were calculated, for sites where there was a sea defence or landfill inhibiting rollback of intertidal habitat.

7.3 Model Results

Direct links between MarESA sensitive biotopes and each stressors extent are shown in Table 39. It is notable that surface sediment abrasion caused by bottom fishing activity extends across ~35% marine nursery biotope area. Biotopes impacted by the other stressors e.g. nutrient enrichments, pathogen inputs and sea-level rise all had much smaller spatial extents (1-17%) compared to that of sediment abrasion. However, it is notable in the case of sea-level rise that much of the impact of this stressor is largely confined to coastal and inshore areas. Most of the abrasion impact is directed towards littoral and subtidal sediment biotopes (including native oyster reefs) but within this there are identifiable hot spots of sediment abrasion (Figure 16). Areas of high abrasion include Southampton Water and the central Solent Channel. There were also notable absences of sediment abrasion across saltmarsh and seagrass biotopes mainly due to recently introduced IFCA bylaws (Prohibition of Gathering (Sea Fisheries Resources) in Seagrass Beds) in 2016 which act effectively as No Take Zones (NTZ) which are Marine Protected Areas (MPA) permanently set aside from direct human disturbance, where all methods of fishing and extraction of natural materials, dumping, dredging or construction activities are prohibited. Interestingly, if these hadn't been included these more recent bylaws, we estimate around 682 ha of seagrass would have been vulnerable to abrasion activities, which would have resulted in an additional reduction of -£1,736,995 in natural capital value.

The spatial modelling suggests that removing benthic abrasion stress in Solent waters would increase levels of nutrient bioremediation, through recovery of littoral and subtidal sediment biotopes (Table 40 & 41). The total predicted increase in this service is for approximately £182.5 M per year of additional N and P remediated as a result of removing benthic abrasion stress. The change to this service mainly arises from littoral sediments for N and subtidal sediments for P, abrasion of which would reduce their level of contribution to this ecosystem service. In contrast, the scientific modelling predicted only a small increase in climate regulating services as a result of alleviating abrasion stress on benthic biotopes (Table 42). This is partly because three of the largest contributors to C sequestration in the marine environment (storage in reedbeds, saltmarsh and seagrass) were not directly impacted by abrasion. The total predicted increase in this service is for approximately £0.17M per year if abrasion pressure is removed.

To put the impacts of this service (and waste remediation) in context, two other stressors on marine water quality (nutrient enrichment and pathogen inputs) are examined through data on WFD and FSA water body status. In the case of pathogens, it is currently assumed that poor water quality is causing the closure of some shellfish fisheries in the Solent due to inadequate waste water infrastructure. Hygiene classifications have been performed for the native oyster populations in several areas in the SEMS and the majority have been classified as a long-term B. This classification means that post-harvest, oysters must be re-laid or purified by cooking by an approved method. The FSA microbiological standard is:

'live bivalve molluscs from these areas must not exceed the limits of a five-tube, three dilution MPN test of 4,600 E. coli 100g FIL in more than 10% of samples. No sample may exceed an upper limit of 46,000 E. coli 100g FIL.'

However, following consideration of the best available evidence, which indicates severe depletion of the native oyster reefs in the Solent, a decision was made for the 2019/2020 season by SIFCA to apply the Temporary Closure of Shellfish Beds Byelaw ('Temporary Closure Byelaw') to all native oyster beds in the Solent. Based on this evidence, we calculate that approximately £1.45 M per year could be saved in terms of water treatment costs and avoided climate change damages if these reefs were restored to class B status (Class B is used here rather than long term class B as an average of all the Solent native oyster reefs). Recovery of the population is likely dependent on serval factors beyond pathogen levels, e.g. improved larval recruitment, predation intensity and hydrographic conditions. Nonetheless, the calculations presented here are useful to exemplify how unfavourable environmental conditions may impact this biotope's ability to provide regulatory ecosystem services.

Available evidence also suggests that if future nutrient levels (N and P) increased in sufficient amounts to reduce all individual waterbody WFD status from "moderate to "poor", there is potentially a relatively large change in the amount of nutrients that could otherwise be remediated. Scrutiny of the changes in Tables 40 & 41 shows that changes in N and P remediation, through which the main impact on this ecosystem service is expected, mainly occur through changes in littoral sediment (including those with macroalgae) condition. The total predicted cost savings of maintaining the current 2016 WFD status designation for this service are £286.44 million per year. Overall, the ecosystem benefits of increased C sequestration as a result of maintaining WFD status is less substantial at around £0.35 million per year. Taken together this gives a hypothetical central estimated value of approximately £286.80 million per year in total replacement and abatement costs savings. In general, this result suggests that the impact of additional nutrient impacts would very likely have significant economic value in Solent waters. These estimates are indicative of potential areas where an increase in waste remediation and climate regulation services due to reducing nutrient loading stress could improve local water quality and assist with WFD compliance. However, more detailed local analysis would be required to establish this link (e.g. some catchments are likely much closer to transitioning to "good" status rather than "poor"). The valuations arrived at would be dependent on assumptions derived from the local policy context involved. As water quality conditions have been pre-set under legislation, then in a cost-effectiveness appraisal it is legitimate to compare (value) this ecosystem service with the cost of other abatement options.

The coastal squeeze scientific modelling concluded that the cumulative biotope balance for mudflat and reedbed biotopes at the end of the century would yield a net gain of £7.94 million per year in regulatory ecosystem service values. However, the annual values presented here do not adequately reflect the true long-term value of these biotopes with regard to removing N, P and C. Thus, in addition to calculating values for the annual flow of these services the Net Present Value (NPV) of these services are also calculated. A 3.5% discount rate is applied, as the recommended social discount rate from the HM Treasury Green Book (2011), to sum the amount of N, P and C remediated/ sequestered each year in order to produce a 100-year total value. Based on these estimates the total NPV of mudflat and reedbed biotopes would be £766.55 million (3.5% discount rate) over the period 2005-2105. If, however current projections of saltmarsh biotope loss are followed then £-5.5 billion (3.5% discount rate) in terms of regulatory service will not be realised as N, P or C will remain either remain in either the atmosphere, or water column. Taken together the gains and losses in biotopes related to coastal squeeze impacts equate to a net potential loss of £-4.75 billion (3.5% discount rate) by the year 2105. The biotope loss projections made in Table 37 may turn out to be conservative but, are based on expert judgement projections of coastal erosion, and do not account for other MarESA sensitive biotope losses due to land conversion (e.g. into residential and commercial uses) which are difficult to predict. As a result, the difference in future value could be less or greater, depending on progress of future biotope compensation schemes within the SEMS area.

Overall, the results of this study reveal spatial and temporal trends in relative seabed impacts resulting from different human activities, based on the sensitivity of the biotopes to a given human activity and stressor type. The estimated total annual water quality related replacement and abatement cost savings from reducing the impacts all four stressors is estimate here at **£516.25 million per year**. The requirements to achieve this saving include: 1) removing" high" intensity abrasive fishing stress across all "medium" to sensitive "high" sensitive biotopes 2) restoring native oyster beds to at least class "B" condition 3) maintaining "moderate" WFD status across all of the Solent's catchments and 4) compensating for the loss of saltmarsh over the next century using saltmarsh biotope creation schemes. The assumptions used are regarded as realistic, and possibly slightly conservative. This approach is taken in order to mitigate the risks of over-estimating values as a result of the significant uncertainties involved in the work, bearing in mind the need for the results to inform policy decisions.

Table 39 The spatial extent (ha) of each stressor associated with each MarESA "sensitive "biotope category as assessed in the present study.

Stressor	Littoral sediment	Littoral sediment (macroalgae)	Subtidal sediments	Saltmarsh	Seagrass	Reedbeds	Native oyster reefs	Total stressor area (ha)	% of nursery biotope area impacted
Abrasion	2705	1810	11234	0	0	0	2839	19893	34.85
Pathogens (*E.coli*)	0	0	0	0	0	0	2839	2839	4.97
Eutrophication	3535	1612	1834	682	340	125	1428	9556	16.74
Sea level rise (Coastal squeeze)	86	0	0	620	0	17	0	723	1.26

Table 40 Cumulative impact of the stressors on the value (£) of nitrogen remediation

Nitrogen	Current stressors (£/yr)		Future stressors (£/yr)		Ecosystem benefit (£/yr)
	Abrasion	Pathogens (E coli) (Class B > Prohibited)	Nutrient enrichment (Moderate>Poor)	Sea level rise (Coastal squeeze)	Potential total value (£) loss
Littoral sediment	-£33,936,795	£0	-£60,650,844	£7,012,150	-£87,575,489
Littoral sediment (macroalgae)	-£21,764,508	£0	-£65,067,209	£0	-£86,831,717
Subtidal sediment	-£10,951,686	£0	£0	£0	-£10,951,686
Saltmarsh	£0	£0	-£27,492,437	-£48,442,481	-£75,934,918
Seagrass	£0	£0	-£677,025	£0	-£677,025
Reedbeds	£0	£0	£0	£451,350	£451,350
Native oyster beds	£0	-£1,357,000	£0	£0	-£1,357,000
				Total	**-£263,876,484**

Table 41 Cumulative impact of the stressors on the value (£) of phosphorous remediation

Phosphorous	Current stressors (£/yr)		Future stressors (£/yr)		Ecosystem benefit (£/yr)
	Abrasion	Pathogens (*E coli*) (Class B > Prohibited)	Nutrient enrichment (Moderate>Poor)	Sea level rise (Coastal squeeze)	Potential total value loss
Littoral sediment	-£4,731,608	£0	-£103,281,270	£358,281	-£107,654,598
Littoral sediment (macroalgae)	-£21,261,439	£0	-£17,735,262	£0	-£38,996,701
Subtidal sediment	-£89,885,719	£0	£0	£0	-£89,885,719
Saltmarsh	£0	£0	-£7,485,363	-£4,836,300	-£12,321,663
Seagrass	£0	£0	-£4,046,136	£0	-£4,046,136
Reedbeds	£0	£0	£0	£107,301	£107,301
Native oyster beds	£0	-£5,632	£0	£0	-£5,632
				Total	**-£252,803,148**

Table 42 Cumulative impact of the stressors on the value (£) of carbon sequestration and storage

Carbon	Current stressors (£/yr)		Future stressors (£/yr)		Ecosystem benefit (£/yr)
	Abrasion	Pathogens (E coli) (Class B > Prohibited)	Nutrient enrichment Moderate>Poor)	Sea level rise (Coastal squeeze)	Potential total value loss
Littoral sediment	-£88,650	£0	-£264,633	£13,962	-£339,321
Littoral sediment (macroalgae)	-£25,808	£0	-£47,888	£0	-£22,390
Subtidal sediment	-£54,278	£0	£0	£0	-£54,278
Saltmarsh	£0	£0	-£27,791	-£29,295	-£57,086
Seagrass	£0	£0	-£9,282	£0	-£8,348
Reedbeds	£0	£0	£0	£494	£494
Native oyster beds	£0	-£88,346	£0	£0	-£88,346
				Total	**-£569,275**

7.4 Model Limitations

There are a number of assumptions and limitations associated with the development of the spatial and expert judgement models to predict the change in the level of ecosystem service provision. These broadly fit into the following five categories:

Applying ecosystem service and sensitivity assessments at broad-scale levels

There is frequently limited evidence available to define quantitative levels of regulatory ecosystem service provision across large spatial scales. In some cases, the majority of evidence may not relate to UK biotopes and species (e.g. the native oyster; see section 3.2) and judgments have to be made with regard to their applicability. Correspondingly, the evidence base to assess how changes in condition of a biotope will lead to changes in the level of ecosystem service is further limited. The MarESA sensitivity assessments are conducted on the lower levels of the EUNIS hierarchy that pertain to biotopes (e.g. L4 & L5). This potentially overestimates sensitivity in most cases of areas mapped at L3 or lower. However, the evidence base for sensitivity of biotope types is scarce, and unlikely to change due to the variation of biological responses to stressors within L4 and L5 types. These limitations would ideally be overcome through increased data availability at lower EUNIS levels, which is not currently possible for all SEMS waters. At the time of writing this report, JNCC has developed an automated process to aggregate sensitivity information at all EUNIS levels (Last *et al.,* 2020) in accordance with the approach taken here, however this process has so far only been applied to offshore biotopes. Future efforts to include coastal and transitional biotopes using a similar automated methodology would greatly facilitate ecosystem service and sensitivity assessments at broad-scale levels if applied elsewhere. Similarly, given more time or as a follow-on exercise it would be possible to run some sensitivity testing around the estimates that have been used within the model(s) here.

Stressors data and temporal resolution

Fishing activity is one of the better documented activity datasets, especially across SEMS waters. However, there are a number of limitations/ assumptions in developing the fisheries effort data layer. The main limitation of combining the different fisheries datasets is that they are all presented at different scales, in different units and levels of detail. In grid cells which have both VMS and inshore sightings data, for example, the gear types can only be split into mobile and static. Where it has not been possible to separate out the mobile gear types this will distort the levels of abrasion as not all the gear types included within this definition would result in the same type and/or degree of abrasion stress. Moreover, the fishing data layer used here for vessels under 15m in length covers the period 2009-2012, primarily due to the best-available data on fishing effort of the inshore fleet that covered different gear types and the entire Solent area. A number of changes in the vessels operating across the Solent since then are known to have occurred and so the proxy exposure to abrasion stress will likely have changed in intensity and distribution. For a more relevant assessment of likely condition due to abrasive impacts, more recent data from an appropriate timeframe in relation to biotope recoverability knowledge would be useful, both for smaller vessels of the inshore fleet and larger commercial vessels. The classification of the levels of abrasion intensity that have been assigned within this project have been based on scientific review and expert judgment. If a different banding or classification system had been used this would have resulted in different model outputs. It would therefore be possible to test the sensitivity of the model to these bandings by repeating some of the analysis using different categories of abrasion intensity. It is important to recognise there are many other activities that may also cause abrasive impacts to the seabed, such as anchoring, municipal dredging and capital dredging. In a complete accounting process these additional activities and resulting stressor impacts could also be mapped in the future and integrated into cumulative layers given more time and data.

Spatial resolution

A further limitation is that fishing effort is assumed to be undertaken across an entire 200th of an ICES rectangle which means that there is a spatial mismatch between the level of stressor that is actually exerted on the underlying biotopes. This resolution is coarse relative both to the level of spatial accuracy of much of the available biotopes data and to the movements (and subsequent impacts) of individual vessels. This point is offset somewhat by overlaying more recent bylaw legislation (e.g. IFCA bylaws (Prohibition of Gathering (Sea Fisheries Resources) in Seagrass) when modelling the changes in ecosystem service provisioning. At this time, issues around privacy and consent prevent access to more recent/detailed VMS records for vessels <15m or data products based on VMS pings, such as interpolated vessel tracks, from being used in this approach. We suggest that future work should be repeated in a more localised area, such as an individual estuary or WFD area for example, as it might be possible to more accurately define the fishing intensity based on the VMS point data. There would still be limitations as a result of this approach (due largely to the temporal frequency of pings), however, it would allow a better definition of the actual fishing intensity compared to the current grid cell-based approach. Recent high-profile discussions in the literature (Enever *et al.,* 2017; Amoroso *et al.,* 2018; Rijnsdorp *et al.,* 2018) have highlighted the wide-ranging interpretations of fishing intensity that arise from the resolution used to report by, and implicitly impacts (and condition) information can be overestimated as a result.

Scenario testing and combining stressors

When projecting ecosystem effects of external drivers' certain uncertainty always originates from the scenarios chosen. In the case of the spatial model, ecosystem indicator results are dependent on the model configuration (i.e. how many types of inshore fishing activity are included in the model). In this report we only considered four types of abrasion stressor on the seabed, yet other types of disturbance activity (e.g. (extraction, dredging, disposal, construction or hand gathering and bait collection) may also have significant impacts on the seabed. Reductions of stressors (e.g. nutrient inputs and fishing stress) also aim at recovery of the ecosystem and reaching a specific environmental objective. Stressor risk to the natural capital asset-benefit relationships within this context were tested against proposed UK targets for achieving Good Ecological Status (GES) for all water bodies derived in Annex V of the Water Framework Directive, and Regional Habitat Compensation targets for the Solent and South Downs (SSD) area. Current thresholds modelled here for GES of seafloor integrity are largely precautionary as there is limited evidence that SEMS WFD catchments will transition to overall "poor" condition. Additionally, potential options for meeting future habitat compensation targets are likely to change and evolve as more locally specific policy thresholds are designed that can support a 'net gain' for the marine systems. Any update/refinement of coastal squeeze calculations, and therefore habitat compensation baseline targets, using the most up-to-date LiDAR and sea level rise predictions would need to be considered as part of any future updates to three Shoreline Management Plans (SMPs); the North Solent SMP, the Isle of Wight SMP and the Beachy Head to Selsey Bill SMP.

The combed results of the spatial and expert models also assume additive relations of stressors, lacking estimates of synergistic and antagonistic effects (Côté *et al.,* 2016), which both have been shown to be prominent in marine ecosystems (Crain *et al.,* 2008) e.g. significant interaction between nutrients, microorganisms and particulates can occur in the water column making prediction of the impacts difficult to ascertain. The strength of the cumulative impact analysis is that it can be used to illustrate particular geographic biotope areas where stressors are many and where the impacts on ecosystem benefits are most apparent. As data on activities and pressures become more readily available and research on how to combine multiple stressors matures, we expect to see improvement in this modelling approach in the near future (see Manning *et al.,* 2018).

Future economic valuations

The valuation of future impacts on marine N, P and C sequestration and storage is intended to be consistent with the valuation of ecosystem benefits provided in Parts A and B of this report. Hence, the central N and P replacement costs and DECC (Department of Energy and Climate Change) central non-traded price of C is therefore used to value C sequestration from the marine environment. However, this price reflects non-traded N, P and C prices which are most closely aligned to current UK policy, but are not currently used for policy appraisal. Whilst non-traded N, P and C used here is indeed not traded within a market, the values are calculated based on market principles related to marginal abatement cost curves and provide a useful benchmark to compare against costs of emission reduction policies. The price of nutrients in existing markets may become more representative of the value of non-traded N, P and C in the future as the institutional framework of markets becomes more established (e.g. UK-specific Emissions Trading Schemes (EUETS)). The decision of which price to use will therefore need to be re-evaluated in the future. The application of the different discount rates used here may also cause a significant difference in the value of the ecosystem service. In addition, the calculations presented here assume that N, P and C removal rates remain the same over time at each WFD location. In the future additional information may become available to show how the nutrient rates (and by proxy their value £) vary with factors such as temperature, CO_2 concentrations and water table depth, but currently these data do not exist.

Conclusions and Recommendations

A natural capital approach to policy and decision making considers the value of the natural environment for people and the economy and is a keystone of the UK governments 25-year plan for the environment. This is because natural capital is a unifying concept as it brings legal requirements about the status of habitats, species and ecosystems and the capacity of ecosystems to provide ecosystem services into a common framework, including their potential value. Securing clean, healthy, productive and biologically diverse seas and oceans are key to achieving different targets of the 25-year plan (HM Government, 2018). Of particular importance is spatially explicit knowledge about ecosystem extent, condition and the different stressors that act on ecosystems for deciding on a prioritisation framework for ecosystem restoration. Meanwhile ecosystem service indicators are also becoming essential to monitor the successful implementation of many other policies including agriculture, fishery, water, climate or public health.

Further research to develop the analysis can be undertaken for both the ecosystem service valuation work and the scientific stressor modelling. Part A, B and C of this report each includes a discussion of research conclusions, limitations and potential for further work. These are combined and expanded on here to give an integrated view of how to take forward analysis following this study.

Part A Mapping, estimating and valuing nutrient removal

The process of undertaking this assessment revealed a lack of accurate and reliable baseline data against which to assess changes in marine natural capital stocks even in an area such as the SEMS which is highly studied compared to other Marines Sites in the UK. It is recognised that existing habitat (biotope) maps in the SEMS are variable in terms of spatial coverage and resolution. For example, the total extent of several biotopes (e.g. seagrass and native oysters) in the SEMS could be assumed to be an underestimate of total area as little or no survey work has been conducted in some locations to confirm presence and extent.

- This considerable information gap for spatial coastal assessments is already acknowledged at a national level (Natural Capital Committee, 2014; Drakou *et al.,* 2017; Strong *et al.,* 2019), but there is a pressing need to undertake consistent monitoring beyond normal policy requirements to support full NC assessments in coastal regions. The coordinated completion of inter-annual, holistic, monitoring surveys at discrete time intervals would assist in the production of robust habitat/biotope maps of the SEMS, thus informing the management measures required to meet the conservation objectives of the site. Application of mapping in the future should make use of the future best available habitat/biotope maps, taking account improvements in Earth Observation (EO) technologies.

Similarly, several EUNIS biotopes (e.g. kelp beds, polychaete reefs, maerl beds, epiphyte and sponge communities) only comprised small areas in the SEMS but could potentially be important contributors to N and P removal.

- Future efforts to include the full breadth of SEMS habitats/biotopes available in a region would be important next step to allow the value of all biotopes to be considered in any future management decisions.

We found high rates of N remediation (on a per m^2 basis) in vegetative and structured coastal habitats such as saltmarshes, reedbeds, seagrass and oyster reefs, while P burial was highest in littoral sediments (overlain with macroalgal mats) and reedbed habitats (on a per m^2 basis).

- Our findings therefore support the need for the conservation and restoration of vegetative and structured habitats in the SEMS, many of which (e.g. saltmarsh and native oyster beds) have been in serious decline over the last few decades. Strategic approaches to restoring saltmarsh and reedbed habitats for example could more than triple the N and P removal capacity of alternate "bare" littoral sediment habitat (on a per m^2 basis). This could help reduce pressure on agricultural areas of land and help reduce point sources of nutrient pollution from wastewater treatment outfalls.

Our findings also support evidence that because of their large area within coastal systems, intertidal and subtidal sediments can provide disproportionately large contributions to nutrient removal in coastal systems when considered at Marine Site or catchment scales.

- Littoral and subtidal sediments are often overlooked in a nutrient removal context yet substantial removal of N by burial and denitrification (which is the only known process that permanently removes N from the ecosystem) also occurs in these sediments. Further research could be done to refine the sequestration fluxes we estimate including:

1) Looking at seasonal patterns of N and P removal *via* burial in UK sediments (including in sediments under vegetative and structured habitats).
2) Considering other mechanisms of nutrient removal for example anammox and dissimilatory nitrate reduction to ammonium (DNRA).
3) Clarify the influence of local environmental factors and biogeographically relevant taxa (e.g. tidal regime, substrate, life history of fauna and climate factors) on denitrification and burial rates in order to refine our understanding of the role of these habitats play in different regions.

The effect of green macroalgal mat habitats/biotopes is also substantial in terms of removing and storing N and P.

- There is clear evidence from EA data that macroalgal mats are beginning to decline across several areas of the SEMS (see Watson *et al.,* 2020). New research programmes such as the Nutrients in Transitional waters (RansTrans) project could also help rapidly remove excessive algal mats and nutrients helping to increase ecological status of coastal systems but, this could also reduce the natural capital value and nitrogen removal capacity of these systems, depending on what habitat algal mat systems are replaced with. Ecosystem services are interdependent, that is, decreasing the production of a service such as nutrient removal (as algal mat extent and biomass decreases) may increase that of another water quality dependent service such as human leisure activities in coastal areas. More research into the different ecosystem services (and disservices) provided by macroalgal mat assemblages would be important to allow management trade-offs to be made.

From a management perspective our N and P loading vs habitat uptake analyses suggest that even if "nutrient neutrality" is achieved for new developments in the SEMS (e.g. *via* nitrogen credits or offsetting), greater nutrient reductions (i.e. not just maintaining the status quo) will be required if habitats and species in the Solent are to recover from the impacts of eutrophication.

- Given the high input levels of N and P to the SEMS waterbodies the best available evidence suggests that focusing on nutrient reductions is still a priority. The results of the assessment, undertaken at a waterbody level, identified that strong N limitation was only found in Langstone Harbour (watershed N bioremediation rate of 92%). We recommend a focus across all SEMS estuaries and harbours to be on N reductions and for developments that are impacting on Southampton Water, Chichester Harbour and Pagham Harbour, a P budget may be required.

The nutrient removal valuations provide compelling evidence that the natural environment of the SEMS provides significant and sustained economic value (potentially billions of £) to society

- Our results indicate that existing habitats/biotopes in the Solent could remove 3,590 tonnes of N yr^{-1} and 811 tonnes of P yr^{-1} estimated to be worth at just over £1.1 billion (N) and £228 million (P) respectively based on each biotopes current Water Framework Directive (WFD) condition status. Given the continuing need to reverse historic environmental declines and prepare for new developments and climate change, this report shows where investment will enhance the nutrient removal services provided by the Solent's waterbodies; and thus, increase the SEMS (and indeed the UK's) overall natural capital.

Part B Potential refinement of the Solent marine natural capital accounts

Water quality assessments may best inform policy if are presented not just as concentrations of N or P removed but also in terms of bundles of other ecosystem services e.g. carbon sequestration and storage, fish and shellfish catches, or recreational opportunities. To date there has been a lack of methods to inform decision makers on how changes in water quality would affect these valuable services.

- We have addressed this knowledge gap by introducing a generalisable natural capital asset register for the assessment and valuation of the SEMS ecosystems services. The total monetary benefits accrued for the ecosystem services investigated under "good status" water quality conditions are estimated at £1,304.38 Million yr^{-1}. However, this is almost certainly an underestimate of the true "value" of water quality related services in the SEMS and a greater understanding of individual and bundled ecosystem services relating to water quality is still required. Specific ecosystem services not addressed here but, that would be useful to value in a water quality context include: fresh water provisioning, sediment stabilisation, natural hazard protection (e.g. floods, storms), raw materials (e.g. biofuels) and other marine "wastes" (e.g. heavy metals, persistent pollutants, microplastics, radioactive substances).

Many exploited fish and macroinvertebrates utilize Solent coastal zone (e.g. bass, eels, polychaete bait species) yet the degree to which coastal habitats/biotopes are important as nursery and spawning grounds had not been quantified. Thus, we reviewed and synthesized literature on the ecological value of 51 commercially important species that use the coastal zones of the Solent.

- Our results suggest that representatives of the Solent species utilized most habitat that we investigated with intertidal soft bottom sediment habitats used heavily as nursery ground (71% of species). Sediment habitats were particularly important for young bass which is an important commercial and recreational fishing target in the UK and Europe. These findings will aid in defining key habitats for protection and restoration in the SEMS and provide baseline information needed to define knowledge gaps for quantifying the habitat value for exploited fish and invertebrates. It is also clear from our analysis that many non-commercially important species in the Solent utilize coastal biotopes. For most species, however, there is insufficient information to judge whether these coastal habitats are essential and limiting to population growth and fishery production.

Overall, the conclusion from the initial natural capital accounts is clear that restoring and/or improving the existing condition of biotopes should be seen as a major consideration for marine management in the Solent.

- The total monetary benefits accrued by improving all habitats/biotopes investigated in this study to "good WFD status" are estimated at £1,601.82 Million yr^{-1}, approximately a £297.45 Million yr^{1} increase on current water quality conditions. The increase in total economic activity is notably larger between moderate ('the status quo') and "high" status conditions (£2,116.99 Million yr^{-1}), with an increase of approximately £812.62 Million yr^{-1}. Although the findings are not presented in terms of a cost-benefit ratio, the estimates provided here have the capacity to do so should those cost estimates become available in the future.

Part C Policy relevance of cumulative impacts on water quality related ecosystem services findings

There are significant gaps in the scientific understanding of how ecosystem services flowing from habitats/biotopes change in response to multiple stressors. The conclusions reached through this part of the study are heavily caveated due to uncertainties in the scientific modelling (such as inaccuracies in benthic habitat and fisheries abrasion data). Nevertheless, these values are indicative of how stressors such as sea level rise are affecting the productivity and health of the marine environment, and the water quality derived services society receives from it.

- Overall, the estimated total annual water quality related nutrient (N and P) replacement and CO_2 abatement cost savings from reducing the impacts all four stressors is estimate here at £516.25 million y^{-1}. The information developed here can be used to better understand the potential value (£) risk of losing ecosystem services of the coastal margins, which will be under increasing threat as climate change proceeds. Including biophysical trade-offs, of different habitat compensation requirements will also enable environmental net gain approaches to address key issues such as climate change, waste, nutrient pollution and natural hazards.

Finally, two factors that the study has (explicitly) not covered would be expected to have a significant influence on benthic ecosystem services are therefore require further research:

- **Displacement of stressors** (e.g. fishing effort, building additional man-made sea defences and diverting nutrient loading): future policies that aim to alleviate stressors that result in displacement of that effort create an additional complication for predicting future impacts.
- **Habitat creation (natural or induced by management)**: The scientific stressor modelling looked at restoration of ecosystem services *via* the removal of multiple stressors on the current habitats/biotopes understood to be present on the Solent's seabed. However, it only considers how mudflat and reedbed biotopes may increase (*via* the Solent Regional Habitat Compensation Programme (RHCP)) in extent over the next one hundred years. It does not cover the possibility that other biotopes (e.g. seagrass, saltmarsh, native oyster beds) would increase in extent due to protection policies allowing habitat/species recovery. Future studies investigating how "net gain" environmental policy's or ongoing restoration schemes (e.g. The Solent Native Oyster Restoration Project, Natural England's seagrass ReMEDIES project or the Solent saltmarsh restoration work undertaken by the Beneficial Use of Dredging in the Solent Project (BUDS)) may affect the flows of ecosystem services would be a useful to enhance the modelling work conducted here.

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Appendices

Table S1 EUNIS biotope area estimates (ha) for the Solent. (N/a) biotope/species known to be present but data not available, blank space (-) biotope not present.

EUNIS assessment unit and code	Littoral sediments (A2.3, A2.4)	Littoral sediments (with macroalgae)	Subtidal sediments (5.2, A5.3, A5,4)	Saltmarsh (A2.5)	Seagrass (A5.53, A5.545, A2.61)	Reedbeds (C3.2, C32.1)	Native oyster (*Ostrea edulis*) beds (A5.435)
Data source	**CCO**	**EA**	**UKSeaMap**	**CCO[(1)]/EA[(2)]**	**HIWW**	**CCO**	**IFCA**
Waterbody-Survey year	**2013**	**2015-2019**	**2018**	**2013 [(1)] 2016[(2)]**	**2014**	**2013**	**2018**
Lymington Estuary	177	42	82	84[(2)]	-	17	-
Beaulieu Estuary	152	21	240	98[(2)]	-	25	N/a
Southampton Water	755	110	1380	264[(2)]	-	78	576
Portsmouth Harbour	591	337	612	72[(1)]	86	1.53	279
Langstone Harbour	1233	364	348	62[(2)]	104	22	491
Chichester Harbour	1641	470	635	305[(2)]	116	25	435
Pagham Harbour	125	14	38	126[(1)]	5	11	-
Yar Estuary	27	15	0	43[(2)]	10	20	-
Newton Harbour	94	77	36	69[(2)]	-	0.09	-
Medina Estuary	21	34	56	11[(2)]	27	0.68	N/a
Bembridge Harbour	24	9	12	4[(1)]	2	0.54	-
Solent (open water)	1374	123	16047	123[(2)]	348	72	1058

Table S2 Total number of native oysters caught in 2018 per m^2 at each station location. Data not available (N/a). Station number refers to the location of native oyster beds and shellfish zones in the Solent (Figure S1 below)

Solent Region	Native oyster (*O. edulis*) ind per m^2.	Native oyster (*O. edulis*) shellfish beds (ha)	Station number
Eastern Solent			
Bramble	0.34	407.79	**8**
Browndown	0.34	417.16	**12**
Chilling	0.30	770.54	**6**
Lee-On-The-Solent	0.145	388.81	**11**
North Channel	0.52	861.18	**10**
Osbourne	0.35	1273.31	**16**
Ryde Middle	1.08	482.29	**9**
Spit Sand	0.22	322.34	**13**
Sturbridge	0.29	186.01	**14**
Western Solent			
Lepe	0.34	935.21	**3**
Newtown	0.30	513.15	**17**
Thorn Knoll	0.01	417.56	**7**
Lymington (Pennington)	0.13	843.94	**1**
Sowley	0.18	1426.02	**2**
Stanswood	0.32	815.18	**4**
Yarmouth	0.22	522.61	**18**
Warner	N/a	N/a	**15**
The Harbours			
Beaulieu	N/a	N/a	**A**
Medina	N/a	N/a	**B**
Southampton water (including Hamble)	0.45	5761.86	**C (5)**
Portsmouth	0.76	2964.89	**D**
Langstone	0.50	1112.82	**E**
Chichester	0.56	6036.35	**F**

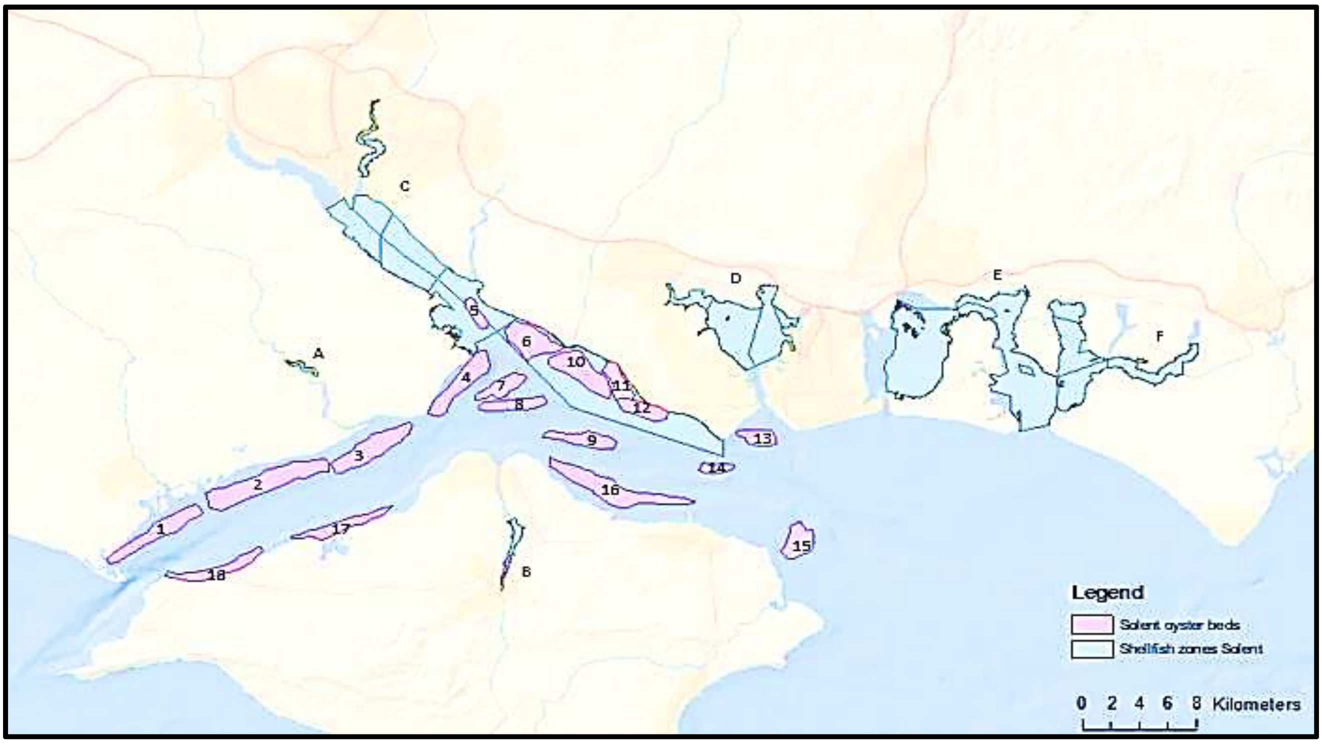

Figure S1 Location of native oyster beds and shellfish zones in the Solent

Table S3 Final List of Ecosystem Services to be Assessed in this Project

Ecosystem service		Definition
Regulating	Climate Regulation	A series of biogeochemical processes regulated by living marine organisms: The regulation of the volatile organic halides, ozone, oxygen and dimethyl sulphide, and the exchange and regulation of carbon, by marine organisms.
	Waste Remediation, Detoxification and Storage	Water and air purification, waste treatment arising naturally as a result of human action: 1) Bioremediation: remediation using plants, remediation using micro-organisms; and 2) Dilution and sequestration: dilution, filtration, sequestration and absorption.
Provisioning	Commercial Fish and Shellfish Harvesting.	Fish, shellfish and seaweed for consumption both from wild capture and aquaculture
Cultural	Recreation, Tourism and Leisure	Specific recreational activities that are dependent on different features within the natural environment. All benefit from general environmental quality and species abundance and marine health, in addition to different activities relying on different marine features.
Supporting	Nursery Function- Biodiversity, Species and Habitat Conservation	Non-use value of maintaining diversity of marine species.

Table S4 Percentage of N and P loads removed from each WFD catchment in the Solent (extent only condition not included)

	N Total Unremediated	N Total remediated	P Total Unremediated	P Total remediated
Lymington Estuary	50	50	0	100
Beaulieu Estuary	34	66	0	100
Southampton Water	81	19	87	13
Hamble Estuary	72	28	0	100
Portsmouth Harbour	68	32	0	100
Langstone Harbour	8	92	0	100
Chichester Harbour	58	42	11	89
Pagham Harbour	68	32	49	51
Yar Estuary	30	70	0	100
Newton Harbour	55	45	0	100
Medina Estuary	80	20	0	100
Bembridge Harbour	89	11	0	100

Table S5 lower (Q1) and upper (Q3) quartile estimates for nitrogen removal (tonnes m^2 yr^{-1})

	Littoral soft sediments		Littoral soft sediments with macro		Subtidal soft sediments		Saltmarsh		Seagrass		Reedbeds		Native oysters	
Nitrogen	**Q1**	**Q3**	**Q1**	**Q3**	**Q1**	**Q3**	**Q1**	**Q3**	**Q1**	**Q3**	**Q1**	**Q3**	**Q1**	**Q3**
Lymington Estuary	21.03	33.24	6.23	22.03	4.40	5.69	24.48	38.64	0	0	0.85	2.49	0	0
Beaulieu Estuary	18.06	28.55	3.12	11.01	12.86	16.66	28.56	45.08	0	0	1.25	3.66	0	0
Southampton Water	85.20	134.70	5.16	18.25	68.79	89.07	67.21	106.10	0	0	0.67	1.96	12.41	119.46
Hamble Estuary	5.08	8.04	10.42	36.82	5.18	6.70	9.72	15.34	0	0	3.23	9.47	7.84	75.46
Portsmouth Harbour	70.21	111.00	50.01	176.74	32.80	42.47	20.98	33.12	15.05	18.19	0.08	0.22	9.81	94.41
Langstone Harbour	146.48	231.58	54.02	190.90	18.65	24.15	18.07	28.52	18.19	22.00	1.10	3.22	17.26	166.15
Chichester Harbour	194.95	308.21	69.75	246.49	34.04	44.07	88.59	139.84	20.29	24.53	1.25	3.66	15.29	147.20
Pagham Harbour	14.85	23.48	2.08	7.34	2.04	2.64	36.72	57.96	0.87	1.06	0.55	1.61	0	0
Yar Estuary	3.21	5.07	2.23	7.87	0	0	12.53	19.78	1.75	2.12	1.00	2.93	0	0
Newton Harbour	11.17	17.66	11.43	40.38	1.93	2.50	20.11	31.74	0	0	0	0.01	0	0
Medina Estuary	2.49	3.94	5.05	17.83	3.00	3.89	3.21	5.06	4.72	5.71	0.03	0.10	0	0
Bembridge Harbour	2.85	4.51	1.34	4.72	0.64	0.83	1.17	1.84	0.35	0.42	0.03	0.08	0	0
Solent (open water)	163.23	258.06	18.25	64.51	860.12	1113.66	35.84	56.58	60.88	73.60	3.60	10.55	37.19	358.03
SEMS Total	**738.82**	**1168.05**	**239.07**	**844.89**	**1044.45**	**1352.33**	**367.16**	**579.60**	**122.12**	**147.63**	**13.64**	**39.98**	**99.79**	**960.72**

Table S6 lower (Q1) and upper (Q3) quartile estimates for phosphorous removal (tonnes m^2 yr^{-1})

	Littoral soft sediments		Littoral soft sediments with macro		Subtidal soft sediments		Saltmarsh		Seagrass		Reedbeds		Native oysters	
Phosphorous	**Q1**	**Q3**	**Q1**	**Q3**	**Q1**	**Q3**	**Q1**	**Q3**	**Q1**	**Q3**	**Q1**	**Q3**	**Q1**	**Q3**
Lymington Estuary	-16.56	1.67	7.16	19.77	-11.37	1.30	3.02	5.01	0	0	0.81	1.73	0	0
Beaulieu Estuary	-14.22	1.44	3.58	9.88	-33.28	3.82	3.53	5.84	0	0	1.19	2.55	0	0
Southampton Water	-67.09	6.78	5.93	16.38	-177.94	20.41	8.30	13.75	0	0	0.64	1.36	1.56	17.79
Hamble Estuary	-4.00	0.40	11.96	33.04	-13.39	1.54	1.20	1.99	0	0	3.07	6.60	1.01	11.38
Portsmouth Harbour	-55.29	5.58	57.42	158.62	-84.85	9.73	2.59	4.29	-7.36	3.50	0.07	0.16	1.33	14.58
Langstone Harbour	-115.35	11.65	62.02	171.33	-48.25	5.53	2.23	3.70	-8.90	4.23	1.05	2.25	2.15	24.61
Chichester Harbour	-153.52	15.51	80.08	221.22	-88.04	10.10	10.94	18.12	-9.93	4.72	1.19	2.55	1.72	20.86
Pagham Harbour	-11.69	1.18	2.39	6.59	-5.27	0.60	4.54	7.51	-0.43	0.20	0.52	1.12	0	0
Yar Estuary	-2.53	0.26	2.56	7.06	0	0	1.55	2.56	-0.86	0.41	0.95	2.04	0	0
Newton Harbour	-8.79	0.89	13.12	36.24	-4.99	0.57	2.48	4.11	0.00	0.00	0	0.01	0	0
Medina Estuary	-1.96	0.20	5.79	16.00	-7.76	0.89	0.40	0.66	-2.31	1.10	0.03	0.07	0	0
Bembridge Harbour	-2.25	0.23	1.53	4.24	-1.66	0.19	0.14	0.24	-0.17	0.08	0.03	0.06	0	0
Solent (open water)	-128.54	12.98	20.96	57.89	-2224.92	255.15	4.43	7.33	-29.79	14.16	3.42	7.35	4.30	51.28
SEMS Total	**-581.79**	**58.77**	**274.47**	**758.26**	**-2701.73**	**309.83**	**45.36**	**75.10**	**-59.75**	**28.41**	**12.97**	**27.84**	**12.08**	**140.49**

Table S7 Percentage (%) of N and P loads removed from each WFD catchment in the Solent (extent and condition)

	N Total Unremediated	N Total remediated	P Total Unremediated	P Total remediated
Lymington Estuary	37	63	0	100
Beaulieu Estuary	25	75	0	100
Southampton Water	78	22	81	19
Hamble Estuary	73	27	0	100
Portsmouth Harbour	60	40	0	100
Langstone Harbour	1	99	0	100
Chichester Harbour	57	43	12	88
Pagham Harbour	63	37	53	47
Yar Estuary	30	70	0	100
Newton Harbour	51	49	0	100
Medina Estuary	80	20	0	100
Bembridge Harbour	84	16	0	100

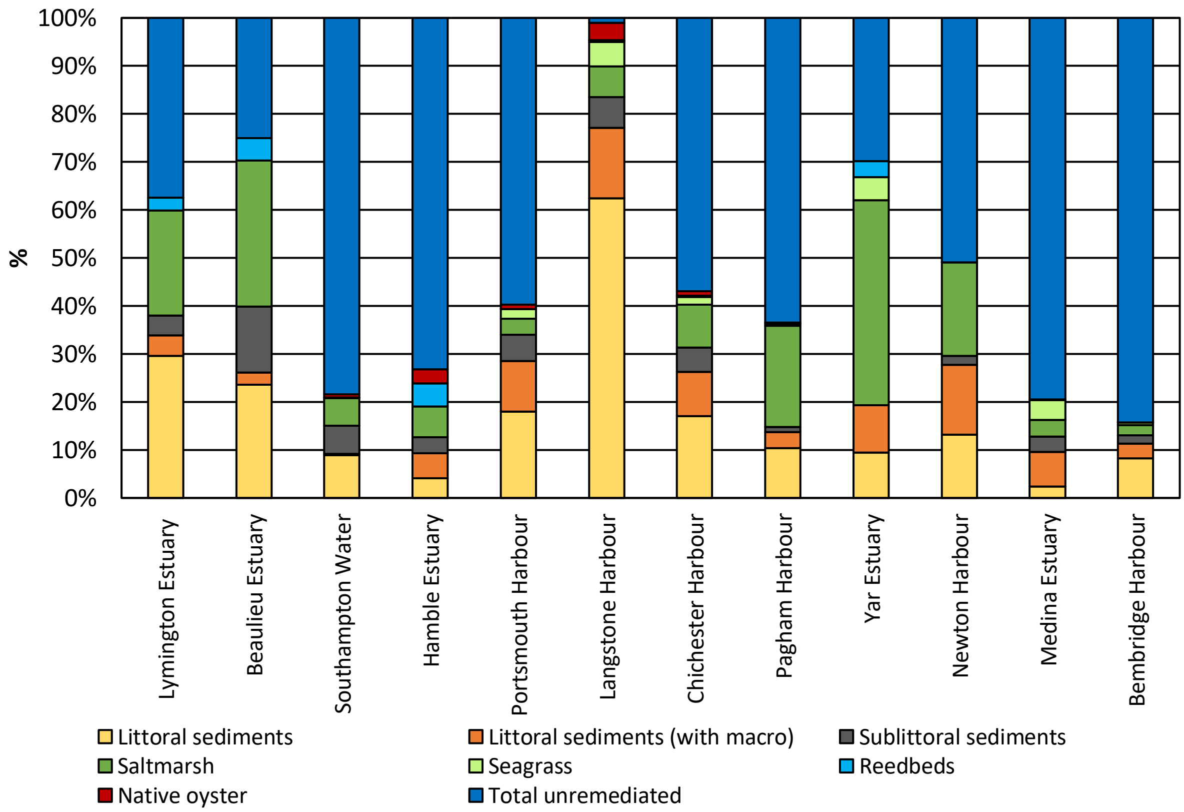

Figure S2 Percentage of Nitrogen remediated by biotopes in the Solent (with extent and condition)

Table S8 Estimates of total cost for nitrogen mitigation options (Bryan *et al.,* (2013); RSPB (2013); BPPDC (2017)).

Measure	Option	Average potential cost (£ kg yr^{-1})
Catchment sensitive farming (CFC)	Application of CSF across whole catchment	24
	Establishment of cover crops following winter wheat	48
	Baling and removal of Oilseed Rape straw	66
	Moving from Oilseed Rape to spring beans	75
	Move from Oilseed Rape to winter oats	92
	Use of clover in place of nitrogen fertiliser on all managed land	137
	No tillage and reduction in livestock numbers to achieve 100% N reduction	175
	10% reduction in fertiliser applied to oilseed rape	197
	Reduced 20% application of N to managed grassland	21
	Allow field drainage systems to deteriorate including land adjacent to watercourses, natural wetlands and ribbon areas.	404
Payments for ecosystem services (PES)	Local conservation body purchases farm holding and over time changes land use	539
	Provide grants for farmers to change land use to commercial woodland	888
	Change of use of public owned land from agriculture to sparsely treed landscape.	48
	Purchase and reversion (ceasing fertiliser use) of arable land	527
	Purchase and reversion (ceasing fertiliser use) of managed land	557
	Land and change use to sparsely treed sparsely treed	539
Upgrades to existing wastewater treatment plants and associated drainage infrastructure	Improve the discharge quality at treatment plants *via* Introduction of N stripping measures.	24.2 -1100

Table S9 Estimates of total cost for phosphorus mitigation options (RAWG (2019))

Measure	**Option**	**Average potential cost (£ kg yr^{-1})**
Catchment sensitive farming (CFC)	Regulatory controls on agricultural phosphorus	5
	Storing and transporting excess P from dairy farms to arable farms	115
	Make available compost to improve soil condition	115
Payments for ecosystem services (PES)	Change in land use from intensive to less intensive grass production	536
	Creation of wetlands	127
	Taking out agricultural land (arable or grass) Production through offsetting	890
Upgrades to existing wastewater treatment plants and associated drainage infrastructure	Reducing flows through sewage network through water efficency measures	250
	On site treatment with disposal systems (e.g. P stripping or wetlands)	218

Table S10 Variability in the estimated annual ecosystem goods production value by biotope type. As determined by the low, mid and high replacement costs and low, mid and high DECC (2011) non-traded carbon values.

	Remediation (£ yr^{-1})	Littoral sediments	Littoral sediments (macroalgal mats)	Subtidal sediments	Saltmarsh	Seagrass	Reedbeds	Native oyster (*Ostrea edulis*)	Total
Nitrogen (Low)	Mean	£32.57 M	£17.83 M	£37.48 M	£14.79 M	£4.55 M	£1.03 M	£17.30 M	£125.55 M
	Median	£26.92 M	£13.18 M	£42.12 M	£15.47 M	£4.13 M	£0.57 M	£4.14 M	£106.53 M
	Min	£21.21 M	£2.45 M	£25.98 M	£8.47 M	£3.83 M	£0.32 M	£2.52 M	£64.78 M
	Max	£49.17 M	£42.08 M	£46.06 M	£22.32 M	£5.49 M	£2.04 M	£59.00 M	£226.16 M
Nitrogen (Mid)	Mean	£294.77 M	£161.37 M	£339.15 M	£133.81 M	£41.16 M	£9.34 M	£156.53 M	£1,136.13 M
	Median	£243.82 M	£118.90 M	£381.12 M	£139.98 M	£37.42 M	£5.15 M	£36.26 M	£962.65 M
	Min	£191.93 M	£22.22 M	£235.11 M	£76.64 M	£34.63 M	£2.90 M	£22.78 M	£586.21 M
	Max	£444.97 M	£380.76 M	£416.76 M	£201.98 M	£49.69 M	£18.44 M	£533.91 M	£2,046.50 M
Nitrogen (High)	Mean	£1,099.13 M	£601.72 M	£1,264.64 M	£851.80 M	£153.48 M	£34.81 M	£583.67 M	£4,589.26 M
	Median	£908.42 M	£444.76 M	£1,421.11 M	£282.74 M	£139.51 M	£19.21 M	£139.59 M	£3,355.35 M
	Min	£715.67 M	£82.84 M	£876.68 M	£169.09 M	£129.14 M	£10.80 M	£84.94 M	£2,069.16 M
	Max	£1,659.22 M	£1,419.77 M	£1,554.01 M	£449.06 M	£185.27 M	£68.76 M	£1,990.85 M	£7,326.94 M
Phosphorus (Low)	Mean	-£1.30 M	£2.54 M	-£5.97 M	£0.30 M	-£0.08 M	£0.10 M	£0.34 M	-£4.08 M
	Median	£0.17 M	£2.40 M	£0.23 M	£0.31 M	-£0.15 M	£0.10 M	£0.11 M	£3.18 M
	Min	-£5.98 M	£0.35 M	-£27.25 M	£0.14 M	-£0.45 M	£0.03 M	£0.00 M	-£33.16 M
	Max	£0.42 M	£5.20 M	£2.86 M	£0.44 M	£0.44 M	£0.17 M	£1.25 M	£10.79 M
Phosphorus (Mid)	Mean	-£73.60 M	£143.23 M	-£336.85 M	£16.77 M	-£4.41 M	£5.76 M	£18.97 M	-£247.50 M
	Median	£9.65 M	£135.09 M	£13.17 M	£17.39 M	-£8.53 M	£5.84 M	£6.77 M	£179.37 M
	Min	-£337.50 M	£19.78 M	-£1,536.97 M	£8.17 M	-£25.16 M	£1.46 M	£0.02 M	-£1,870.19 M
	Max	£23.48 M	£293.48 M	£161.55 M	£24.94 M	£24.57 M	£9.85 M	£70.45 M	£608.32 M
Phosphorus (High)	Mean	-£232.28 M	£452.04 M	-£1,063.10 M	£52.93 M	-£13.92 M	£18.16 M	£59.88 M	-£726.28 M
	Median	£30.42 M	£427.66 M	£41.62 M	£54.95 M	-£26.96 M	£18.48 M	£19.96 M	£566.13 M
	Min	-£1,065.17 M	£62.42 M	-£4,850.71 M	£25.79 M	-£79.39 M	£4.61 M	£0.08 M	-£5,902.37 M
	Max	£74.11 M	£926.23 M	£509.87 M	£78.72 M	£77.53 M	£31.08 M	£222.35 M	£1,919.89 M

Fish Species		Habitat								
Common Name	Scientific name	Seagrass	Intertidal soft bottom	Subtidal soft bottom	Kelp	Shallow open water	Saltmarsh	Shellfishl beds	Macroalgae	Rocky shore
Anchovy	*Engraulis encrasicolus*					N		N,F		
Bass	*Dicentrarchus labrax*	N					N	N,F		
Black Bream	*Spondyliosoma cantharu*									
Blacck Goby	*Gobius niger*									
Brill	*Scophthalmus rhombus*		N	S,N						
Cod	*Gadus morhua*	N		N	N, F				N	N
Common Blenny	*Lipophrys pholis*									
Dover Sole	*Solea solea*		N,F			S,M				
Dragonet Reticulated	*Callionymus reticulatus*									
European eel	*Anguilla anguilla*	N, F		N	N, F	M	N, F	N,F	N, F	N, F
Flounder	*Platichthys flesus*		N	N,F			N	N,F		
Garfish	*Belone belone*									
Gilthead bream	*Sparus aurata*									
Goby - Common	*Pomatoschistus microps*									
Goby - Sand	*Pomatoschistus minutus*									
Goby - Two spotted	*Gobiusculus flavescens*									
Goby - Black	*Gobius niger*									
Goby-Painted	*Pomatoschistus pictus*									
Transparent Goby	*Aphia minuta*									
Goby - Rock	*Gobius paganellus*									
Herring	*Clupea harengus*	S				N, F		N,S,F	S	S
Long spined sea scorpic	*Taurulus bubalis*									
Mullet - Golden Grey	*Chelon aurata*									
Mullet - Thick Lipped	*Chelon labrosus*									
Mullet - Thin Lipped	*Liza ramada*									
Pipefish - Greater	*Syngnathus acus*									
Pipefish - BroadSnout	*Syngnathus typhle*									
Plaice	*Pleuronectes platessa*		N	N,F			N			
Pollack	*Pollachius pollachius*			N	N				N	N
Sandeel	*Ammodytes tobianus*			S,N,F		F				
Sandeel-smooth	*Gymnammodytes semisquamatus*									
Snandeel -lesser	*Ammodytes marinus*									
Sandeel -greater	*Hyperoplus lanceolatus*									
Sand Smelt	*Atherina presbyter*									
Smelt	*Osmerus eperlanus*									
Short Spined Sea Scorpi	*Myoxocephalus scorpius*									
Snake Pipefish	*Entelurus aequoreus*									
Lemon Sole	*Microstomus kitt*									
Solenette	*Buglossidium luteum*									
Sprat	*Sprattus sprattus*		N			N,F	N			
Stickleback - Three spin	*Gasterosteus aculeatus*									
Stickleback - Fifteen spi	*Spinachia spinachia*									
Wrasse - Ballan	*Labrus bergylta*									
Wrasse -Cuckoo	*Labrus mixtus*									
Wrasse - Corkwing	*Symphodus melops*									
Shellfish Species										
Common Name	Scientific name									
Native Oyster	*Ostrea edulis*		S, N, F	S, N, F						
Manilla clam	*Ruditapes philippinarum*		S, N, F	S, N, F						N, F
Hardshell clam	*Mercenaria mercenaria*		S, N, F	S, N, F						N, F
Cockle	*Cerastoderma edule*		S, N, F	S, N, F						
Polychaete Species										
Common Name	Scientific name									
King Ragworm	*Alitta virens*	F	S, N, F	S, N, F	F	F	F	F	F	
Lugworm	*Arenicola marina*	F	S, N, F	S, N, F	F	F	F	F	F	
Common Ragworm	*Hediste diversicolor*	F	S, N, F	S, N, F	F	F	F	F	F	

Figure S3 Presence (1)/absence (blank cell) of fish and mobile invertebrates in each region of the Solent

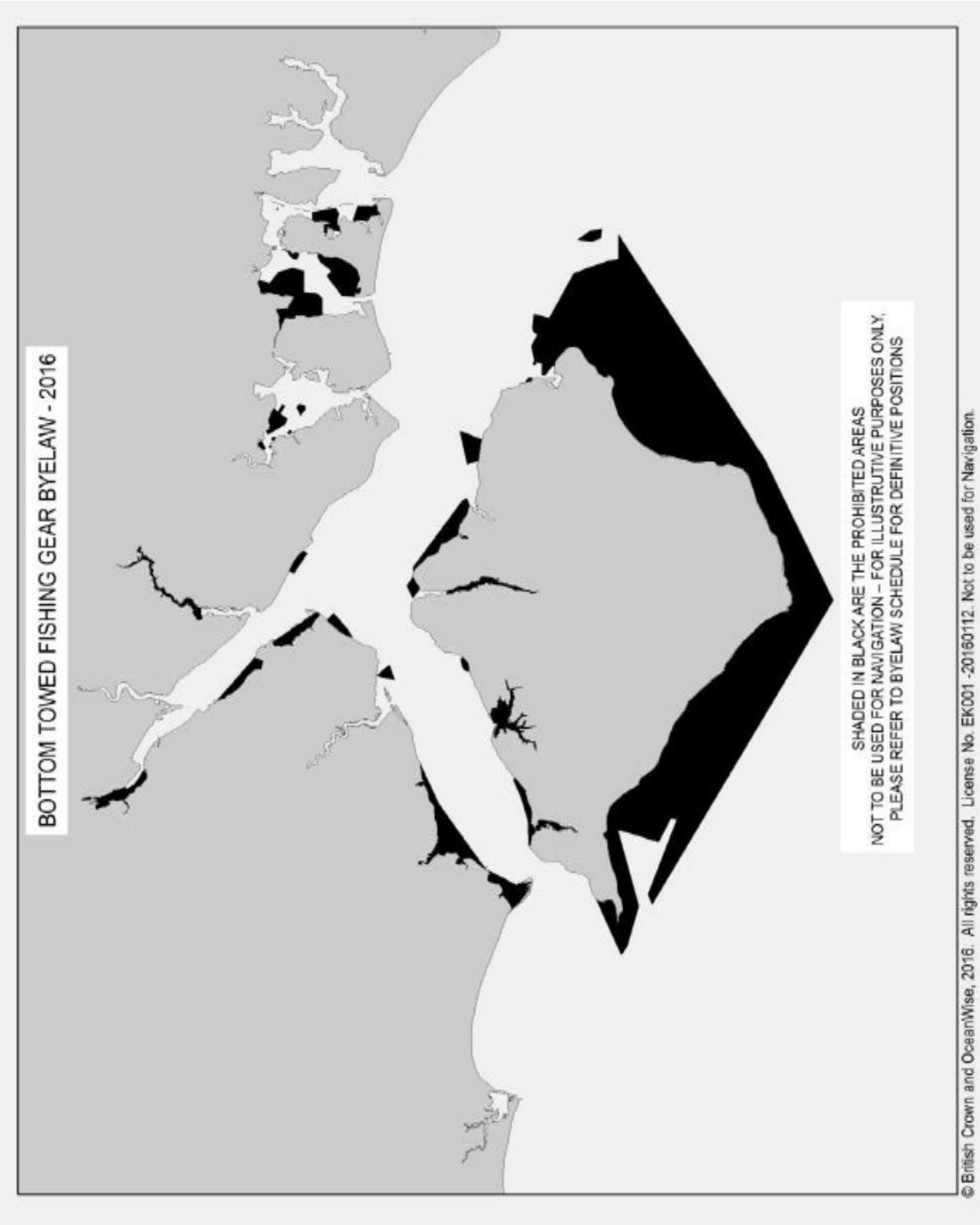

Figure S4 Southern IFCA Bottom Towed Fishing Gear Bylaw 2016 area.

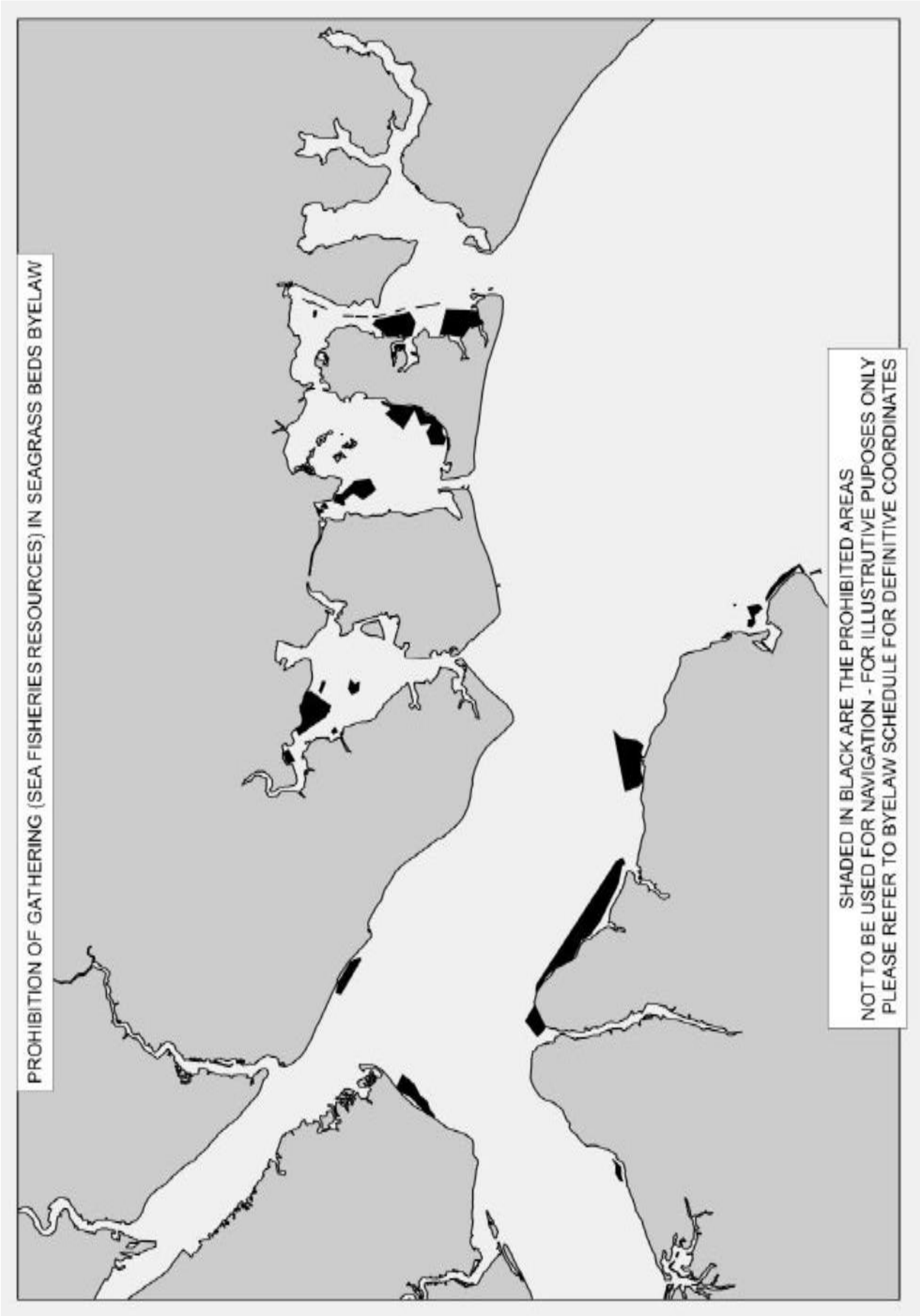

Figure S5 Southern IFCA Prohibition of Gathering (Sea Fisheries Resources) in Seagrass Beds Bylaw area.