Assessing the natural capital value of water quality and climate regulation in temperate marine systems using a EUNIS biotope classification approach

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Abstract

Using a natural capital framework to inform improvements to water quality and mitigation of climate change requires robust and spatially explicit ecosystem service data. Yet, for coastal habitats this approach is often constrained by a) sufficient and relevant habitat extent data and b) significant variability in baseline assessments used to quantify and value regulatory habitat services. Here, the European Nature Information System (EUNIS) habitat classification scheme is used to map seven key temperate coastal biotopes (littoral sediment, mat-forming green macroalgae, subtidal sediment, saltmarsh, seagrass, reedbeds and native oyster reefs) within the UK's Solent European Marine Site (SEMS). We then estimate the capacity of these biotopes to remove nitrogen (N) and phosphorus (P) and carbon (C), alongside monetary values associated with the resulting benefits. Littoral and sublittoral sediments (including those combined with macroalgae) were the largest contributors to total N, P and C removal, reflecting their large biotope area. However, our results also show considerable differences in relative biotope contributions to nutrient removal depending on how they are analysed and delineated over large spatial scales. When considered at a regional catchment level seagrass meadows, saltmarshes and reedbeds all had considerable N, P and C removal potential. Overall, we estimate that SEMS biotopes provide nutrient reductions and avoided climate damages equivalent to UK £1.1 billion, although this could be nearly £10 billion if water-treatment infrastructure costs and high carbon trading prices are utilised. Despite the variability in the final natural capital evaluations, the substantial regulatory value of N, P and C ecosystem services support a strong rational for restoring temperate coastal biotopes.

Key Words: Natural Capital, EUNIS mapping, Ecosystem Services, Nutrient Regulation, Carbon Burial, Economic Valuation.

1 Introduction

The valuation of services and goods provided by ecosystems underpins conservation and management processes leading to policy decisions based on natural capital (NC) (Norton *et al.,* 2018). NC refers to the stocks of renewable and non-renewable resources (e.g. water sediments, air and all living organisms) that combine to yield a flows of ecosystem services (ES) and is increasingly being used as a concept to support the development of strategic tools and general frameworks for restoring and maintaining high quality ecosystems (Guerry *et al.,* 2015; Hooper *et al.,* 2019). Mapping biologically and ecologically important areas is a fundamental first step in an NC decision framework (e.g. Burdon *et al.,* 2019; Rees *et al.,* 2019) followed by measuring the extent, status and value of NC stocks and the ES and benefits derived from them. Coastal habitats such as saltmarshes, seagrass meadows, bivalve reefs, reedbeds and mudflats (littoral and sublittoral) are documented to be particularly difficult to include in spatial NC assessments due to a lack of baseline information relating to their area and condition (Natural Capital Committee 2014), but are intricately involved in the provision of many ES, that act to regulate local water quality and climate conditions so that adverse impacts on human well-being and biodiversity are avoided. Nitrogen (N), phosphorous (P) and carbon (C) are three important nutrients involved in the biogeochemical processes regulating water quality and climate regulation (e.g. Pendleton *et al.,* 2012; Beaumont *et al.,* 2014; Watson *et al.,* 2016). Extraction of these nutrients from the coastal environment for example often takes place through three different pathway's (i) harvest/removal of the organisms (e.g. bivalves) – thereby returning N, P or C back to land; ii) through denitrification and its intermediates e.g. anaerobic ammonium oxidation (anammox), leading to loss of N to the atmosphere or iii) sequestration and burial of N, P or C in benthic marine sediments. These processes contribute to several final ES including; waste remediation, climate regulation, drinking water, fisheries provision and recreation (e.g. bathing waters). Yet despite the multiple efforts to measure these nutrient fluxes and processes, using both field and laboratory systems, significant variability exists in long term N, P and C sequestration estimates (e.g. Kellogg *et al.,* 2014; Krause-jensen and Duarte, 2016). Moreover, most valuation estimates of these services do not account for the fact the ecological functions underlying these ES can vary spatially and temporally (Spake *et al.,* 2019), which can greatly affect the benefits they provide.

The production of marine habitat maps typically relies on the use of habitat classification schemes, with the European Nature Information System (EUNIS) habitat classification scheme being the most extensively used for marine mapping and modelling efforts in Europe (see Strong *et al.,* 2019 and references therein). However, whilst EUNIS land-cover data is now used extensively to map the distribution of several ES in terrestrial systems, a comparable approach exploiting EUNIS marine and coastal habitat data is still lacking (Hooper *et al.,* 2019). For example, green macroalgal mat communities (e.g. *Ulva* and *Enteromorpha* spp.) are extensive and extremely common in coastal regions as a response to eutrophic conditions (Lyons *et al.,* 2012), yet these features are currently not included explicitly within NC assessments due to a lack of inclusion within routine EUNIS mapping exercises.

Here, a regional scale approach is used to map the distribution of seabed-associated ES of seven key coastal habitats by using a methodology that brings together i) a geospatial dataset representing the NC stocks of the region using an EUNIS mapping approach ii) information on each habitats capacity to remove common nutrients including N, P and C iii) the monitory value of benefits associated with avoided wastewater treatment costs and reduced climate change damages. The coastal habitats of the Solent European Marine Site (SEMS) located on the south coast of the England are under significant pressure from nutrient inputs, exploitation and coastal habitat loss. As such, the SEMS is a global exemplar of areas with the greatest need for NC assessment to improve the ES provided by key coastal habitats. Information surrounding the long term removal of N, P and C by coastal habitats are poorly represented and undervalued in the SEMS despite the need for accurate monetary valuation for policymakers (Keeler *et al.,* 2012). We therefore sourced relevant broad-scale habitat mapping data to generate habitat area coverage estimates, and for the first time calculate the contributions of littoral sediments with green macroalgal mats separately within an NC framework. We then collate from published sources biophysical N, P and C removal rates as a function of coupled nitrification-denitrification (N only) and long-term burial in sediments (N, P & C) for each of the key habitats linked to the ES of waste remediation (Watson *et al.,* 2016) and climate regulation (NEA 2011). As N, P & C are also accumulated into biogenic material (e.g. the shell of bivalves), we have incorporated this in to our calculations for oyster reefs.

Values of N and P removal rates for maintaining good water quality were then estimated using the replacement cost method (see Farber *et al.,* 2002). This captures the cost difference associated with reaching a nutrient reduction target by relying on the capacity of natural systems as opposed to utilising human-generated alternatives. However, just like the biophysical rates, replacement costs will vary depending on the data source, replacement type and regional context. To generate relevant estimates of economic value associated with N and P, we use the most current temperate habitat-specific costs of nutrient reduction measures (e.g. Bryan *et al.,* 2013; BPPDC 2017). These provide 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. To assess the sequestration value of C we use low, medium and high UK prices for CO_2 based on the marginal abatement cost method (DECC, 2011) which represents the maximum marginal abatement cost needed to meet a specific emission reduction target. Combining the calculated biophysical rates with N and P replacement costs and the CO_2 abatement costs we express the NC in a range of economic values for the region. Together, these regulatory services valuations provide a widely comparable methodology that can be applied to other temperate coastal marine systems to support meaningful national and local scale policy.

2. Materials and methods

2.1 Mapping natural capital using a EUNIS approach

To collate information on NC stocks a range of GIS broad-scale SEMS habitat maps and datasets were utilised (Table 1) and combined using Arc GIS (version 10.7). Seven habitats identified as having distinct contributions to nutrient fluxes and blue carbon stocks (Potts *et al.,* 2014) were established using the EUNIS biotope classification system as recommended by Hooper *et al.,* (2019). Based on the Potts *et al.,* 2014 scoring method — reedbed, saltmarsh and seagrass biotopes had the greatest potential to remediate nutrients and sequester carbon (while oyster reefs and subtidal sediments were considered unlikely to provide the same level of service delivery) (Table 1). As, by definition, habitat types at EUNIS level-4 and beyond are determined by both their biotic and abiotic features, the underlying assessment units 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. littoral soft sediments and soft subtidal sediments). Littoral and sublittoral coarse sediment classifications (A2.1 and A5.1) were not analysed further due to their small area coverage in the SEMS region. Ancillary datasets on seagrass extent, macroalgal mat communities and native oyster (*Ostrea edulis*) reefs were also sourced from local monitoring programmes. To estimate the native oyster biotope area, locations of commercial reefs, active oyster dredge areas and numbers of individuals caught yr^{-1} in key sections of the SEMS were combined. Seagrass (*Zostera spp*.), and macroalgal mat areas are based on the largest estimates of coverage within the SEMS using surveys conducted from 2006-2019.

The final biotope estimates were also assessed using a derived quality rating (Lillis, 2016) indicating the likelihood of a particular biotope being correctly mapped within a study area (Low: 0, 4: High; Table 1). This enables end-users to determine their adequacy for decision-making, and future survey effort can be directed to low scoring areas. The final combined biotope map and a disaggregated summary of biotopes present in the 12 Water Framework Directive (WFD) transitional and coastal assessment units of the SEMS are also provided in Appendix Figure S1 and Table S1. 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.

Table 1 EUNIS biotope area estimates (ha) and confidence assessment for the SEMS including Pagham Harbour) from the following data sources: Coastal Channel Observatory (CCO), Environment Agency (EA), Joint Nature Conservancy Council (JNCC) UKSeaMap, Hampshire & Isle of Wight Wildlife Trust (HIWW), The Southern Inshore Fisheries and Conservation Authority (SIFCA). JNCC (Lillis, 2016) qualitative confidence in mapping assessment scores (Low 0-4 High). Potts *et al.,* (2014) scale of ecosystem service supply by biotopes classification: 1= Low contribution, 2= Moderate contribution, 3= Significant contribution.

EUNIS assessment unit	EUNIS Code(s)	Data source	Waterbody -Survey year	Area (ha)	JNCC confidence score	Potts *et al.,* (2014) nutrient cycling/carbon sequestration
Littoral sediments	(A2.3, A2.4)	CCO	2013	6204	2.5	2.5/1.5
Littoral sediments (with macroalgae)	-	EA	2015-2019	1616	4	2.5/ 1.5 (A2.3/A2.4 mud and mixed sediments used as proxy)
Subtidal sediments	(A5.2, A5.3, A5.4)	UKSeaMap	2018	19486	4	2/No data
Saltmarsh	(A2.5)	CCO/EA	2013/2016	1261	2.5	3/3
Seagrass	(A5.53, A5.545, A2.61)	HIWW	2006-2014	698	1	3/2
Reedbeds	(C3.2, C32.1)	CCO	2013	273	2.5	3/3 (A2.5 saline reedbeds used as a proxy)
Native oyster (*Ostrea edulis*) reefs	(A5.435)	IFCA	2018	2839	2	2/1

2.2 Biophysical rates

Biophysical rates for N, P and C as a function of assimilation in oyster shells, coupled nitrification-denitrification (hereafter denitrification) and long-term burial in sediments, were quantified from previous studies of temperate estuarine and coastal biotopes (see Table 2), often using different methodologies. For example, in the case of N, many studies (e.g. Adams *et al.,* 2012) measured denitrification *via* total N_2 loss and burial *via* NH_3 which may also include estimates of other anaerobic process such as anammox or dissimilatory nitrate reduction to ammonium (DNRA). Therefore, although our N denitrification and burial estimates undoubtedly include contributions from these ecosystem processes, the paucity of measurements precludes meaningful extrapolation. Net rates of nutrient remediation and carbon sequestration (g m^{-2} yr^{-1}) are also 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). It is important to note, this study has not attempted to estimate these balances, but used the most relevant and appropriate values reported in the scientific literature for calculating means, medians and ranges (minimum and maximum). In addition, surprisingly no studies — to our knowledge — have examined the long-term sequestration capacity of native oyster reefs (*Ostrea edulis*) including properties such as denitrification, P burial and C assimilation in shell tissues. We, therefore, generated estimates of nutrient loss (denitrification) and sequestration (assimilation in shell and burial) for this biotope using studies with *Crassostrea virginica* (Eastern American oyster). By combining the assembled biophysical rates (Table 2) with data on the current area of the regions biotopes (Table 1), estimates of the relative quantities of N, P and C removed by these biotopes were derived.

Table 2 Nitrogen, phosphorus and carbon annual removal rates used for biotope types occurring in the Solent 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*).

EUNIS biotope	Ecosystem process/function	Nitrogen[1] (g N m^{-2} yr^{-1})				Phosphorous[2] (g P m^{-2} yr^{-1})				Carbon[3] (g C m^{-2} yr^{-1})				References
		Mean	Median	Min	Max	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	210	139.5	18	1713	(Adams *et al.,* 2012[1,2]; Burrows *et al.,* 2017[3])
	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	83	110	19	191	(Burrows *et al.,* 2017[3]; Duarte *et al.,* 2005[3]; Eyre *et al.,* 2016[1]; Holmer *et al.,* 2006[2]; Romero *et al.,* 1994[3])
	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	382	484.5	5.17	554	(Brix *et al.,* 2001[3]; 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 sediment	Burial	9.2	9	7	11.4	-4.2	0.6	-19.3	1.3	155.2	130	18.7	291.6	(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 sediment (macroalgal mats)	Burial	33.3	24.6	4.7	78.2	31.4	29.7	4.3	64.4	264.2	312.3	96.1	336.3	(Palomo *et al.,* 2004[2]; Trimmer *et al.,* 2000[1,3], 1998[3])
	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	50.6	35	4.6	150	(Burrows *et al.,* 2017[3]; 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*) reefs	Assimilation in shell (g/individual)	0.18	0.14	0.02	0.4	0.1	0.09	0.003	0.4	4.9	4.4	0.6	10	(Higgins *et al.,* 2011[1,2,3])
	Burial	2.1	0.6	0	7.8	2.3	0.7	0	8.4	-10.5	4	-71	21	(Fodrie *et al.,* 2017[3]; Newell *et al.,* 2005[1,2])
	Denitrification	16.4	3.7	2.7	55.6	-	-	-	-	-	-	-	-	(Kellogg *et al.,* 2014[1])

2.3 Valuation approach

To estimate the economic value associated with N and P removal, we rely on the actual costs of nutrient reduction measures undertaken on the UK's southeast coast. The value of bioremediation rates for maintaining clean water was estimated using a replacement cost valuation method (e.g. Farber *et al.,* 2002). This captures the difference in costs associated with reaching a nutrient reduction target by relying on the capacity of natural systems as opposed to utilising a manufactured alternative. Replacement costs for removing a kilogram of N vary substantially (Table 3) so were extracted from a combination of nutrient management and planning documents (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) a neighbouring catchment of the Solent. To incorporate cost variability, we also consider the lowest and highest cost values based on diffuse CSF initiatives and traditional water treatment to remove N and P from upstream point sources. Average abatement costs of reducing N and P from these sources are estimated as 295 [£/kg] for N and 282 [£/kg] for P. We use these costs as our mid-range conservative ecosystem replacement value estimates.

To estimate the provision of carbon sequestration by the biotopes we use the British Department of Energy and Climate Change (DECC) low, medium and high range of non-traded carbon prices 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 use the non-traded values which represent the maximum marginal abatement cost needed to meet a specific emission reduction target in the future. Combining the previously calculated biophysical rates from Section 2.2. with N and P replacement costs £/kg/yr^{-1} and the CO_2 marginal abatement costs, £/ tonnes /yr^{-1}, estimates can be made for the value of these ES.

Table 3 Summary of estimated replacement and abatement costs - nutrient and carbon removal

Value	Valuation references	Notes
£5-23 N or P (£/kg)	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)		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 or P (£/kg)		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).
£30-90 C (£/Tonnes)	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).

3. Results

3.1. Nitrogen, phosphorus and carbon removal

As expected, the extent of each biotope type within the SEMS varied dramatically (Figure 1a). Although dominated by sediments (sublittoral, littoral and littoral covered with algal mats), which make up 84% of the area, these soft sediment biotopes are interspersed with seagrass meadows, saltmarshes and oyster reefs with a small amount of reedbeds. Many of these biotopes are fragmented (Figure S1) within localised areas of the SEMS.

Across the biotopes the net effect on N, P and C removal varied by up to four orders of magnitude and sometimes included both positive and negative values even within the same biotope (Figure 1 [b-d]). Of the 21 biophysical rates measured here, 17 mean values and 20 median values were positive, generally indicating enhancement in nutrient removal rates. Negative biophysical values were recorded for C burial by native oysters and for P burial by seagrass and littoral sediments. Subtidal sediments also exhibited the greatest range of biophysical rates of all the biotopes with mean negative P burial values recorded in excess of -1000 tonnes yr^{-1}.

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 (24.2% Figure 1 [a]) and the high burial rates of the macroalgae (33.3 g N m^{-2} yr $^{-1}$; Table 2). 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 (60.3% Figure 1 [a]). Saltmarsh, seagrass and reedbed biotopes were also highly productive in removing N at the catchment level with mean rates between (11.6-36 m^{-2} yr $^{-1}$; Table 2), but their contribution to the overall nutrient budget was often small because of the limited combined extent of these biotopes (6.8% Figure 1 [a]). In general, native oysters enhanced N removal rates, but the biophysical values varied (Figure 1 [b]) 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 2) 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 (Figure 1 [c]). A combed negative efflux of -1455 P tonnes yr^{-1} was calculated for littoral and subtidal sediments using the mean biophysical values. 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 2) but, lagged behind macroalgal sediment biotopes in terms of total P burial due to the low area coverage. 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) (Figure 1 [c]).

Coastal and offshore sediments (including those overlain with macroalgae) had the largest burial totals for C in the region removing between 4270-9860 tonnes yr^{-1} based on mean values (Figure 1[d]). Saltmarsh biotopes were the next largest sinks of C removing 2646 tonnes yr^{-1} *via* burial in the underlying sediment. Native oysters were also considered to be a net small source of C to the overlying water (-159 tonnes yr^{-1}) when using the mean biophysical values but were considered to remove C (238 tonnes yr^{-1}) when using the median burial estimates.

By combining the mean values for all the biotopes, we estimate that 3831 N tonnes yr^{-1}, -813 P tonnes yr^{-1} and 27883 C tonnes yr^{-1} are currently removed by existing biotopes in the region. In contrast, the total median rates for the SEMS were lower for N and C, removing 567 & 3803 fewer tonnes of each nutrient per yr^{-1} respectively, while P burial was 1452 tonnes per yr^{-1} greater representing a region wide positive sink of P (639 tonnes P yr^{-1}). A summary of the N, P and C removal potential for individual catchments in the region are given in Appendix Figures S2 & S3.

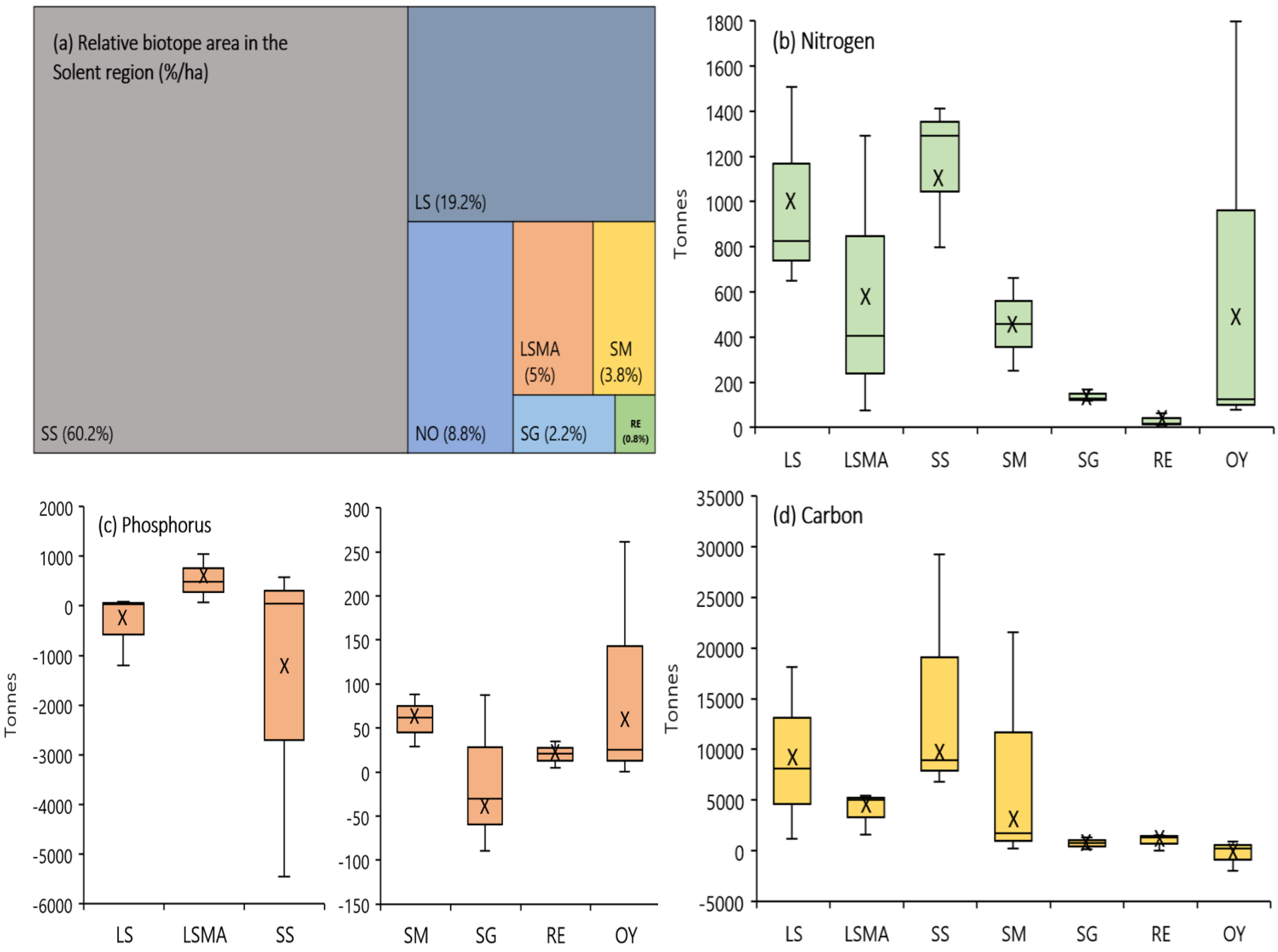

Figure 1 Relative biotope area (1a) and long-term nitrogen (1b), phosphorous (1c) and carbon (1d) removal by the SEMS coastal and subtidal benthic biotopes. Figures 1b, 1c and 1d show the lower (Q1) and upper (Q3) quartiles, median (-), mean (X) and range (error bars). LS (Littoral sediment), LSMA (Littoral sediment with macroalgae), SS (Subtidal sediment), SM (Saltmarsh), SG (Seagrass), RE (Reedbeds), OY (Native oysters).

3.2 Economic value

Based on the total economic benefits valuation, N, P and C removal by the SEMS biotopes are estimated to be: £962 million (N), £179 million (P) and £1.44 million (C) respectively (Table 4), when applying the median N, P and C biophysical rates, average replacement cost options and DECC (2011) non-traded carbon values. Removal of N represents 84.2% of the total value, followed by P at 15.7 %, while C represents only 0.1% of the total value. Economic benefit calculations using the total mean biophysical rates were slightly higher compared with using the median biophysical rates for N and C (£1.1 billion for N and £1.66 million for C) but lower and indeed negative for P (-£247 million) due to the high potential loss of P (tonnes) from the large areas of littoral and subtidal sediments. Further, we assume managers would select the least cost options from among the feasible alternatives for nutrient abatement, but if more costly options are selected due to a high dependence on new advanced wastewater treatment plants and associated drainage infrastructure to maintain water quality goals, the value of N and P removal in the SEMS region could be as high as £7.3 billion and £1.9 billion, respectively (Table S2 upper bound estimates). A full breakdown of (minimum, average and maximum) costs by biotope are given in Appendix Table S2. Generally, the average values (£) per hectare of vegetated biotopes were higher than comparative estimates for bare (littoral or sublittoral) sediment or oyster reefs Table 4) , with saltmarsh being the most important biotope for N (£111,009 per ha yr^{-1}), littoral sediments overlain with macroalgae were foremost for P (£83,853 per ha yr^{-1}) followed by reedbeds for C (£290.53). However, when considering the total biotope values littoral and sublittoral sediments were collectively more valuable for N and C removal, estimated to be worth 624 Million yr^{-1} and 0.90 Million yr^{-1} respectively (Table 4).

Table 4 Summary of the total estimated economic value provided by a hectare of biotope 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.

Unit	Biotope	Biophysical change valued in analysis (Median tonnes yr^{-1})	Economic value captured	Total average value per hectare (£ Annualized)	Total average value (£ Annualized)
Nitrogen	Littoral sediments (LS)	827	Based on the cost of replacing artificial substitutes with the ecological service of waste remediation where cost is a proxy for the nitrogen removal benefits of this regulation.	£39,300	£243.97 M
	Littoral sediments (with macroalgae) (LSMA)	403		£73,578	£118.89 M
	Subtidal sediments (SS)	1292		£19,559	£381.12 M
	Saltmarsh (SM)	475		£111,009	£139.98 M
	Seagrass (SG)	127		£53,607	£37.41 M
	Reedbeds (RE)	17		£18,869	£5.15 M
	Native oyster (*Ostrea edulis*) (OY)	123		£12,774	£37.44 M
	Total N	3264		Total N	£962.65 M
Phosphorus	Littoral sediments (LS)	34	Based on the cost of replacing artificial substitutes with the ecological service of waste remediation where cost is a proxy for the phosphorus removal benefits of this regulation.	£1,555	£9.65 M
	Littoral sediments (with macroalgae) (LSMA)	479		£83,853	£135.09 M
	Subtidal sediments (SS)	47		£677	£13.17 M
	Saltmarsh (SM)	62		£13,807	£17.39 M
	Seagrass (SG)	-30		-£12,239	-£8.53 M
	Reedbeds (RE)	21		£21,448	£5.84 M
	Native oyster (*Ostrea edulis*) (OY)	25		£2,483	£6.77 M
	Total P	639		Total P	£179.39 M
Carbon	Littoral sediments (LS)	8138	Based on avoiding the global economic damages of climate change (floods, droughts, famine, sea level rise, etc), as captured by DECC short-term non-traded carbon values	£78.70	£0.49 M
	Littoral sediments (with macroalgae) (LSMA)	5031		£187.37	£0.30 M
	Subtidal sediments (SS)	6820		£21.00	£0.41 M
	Saltmarsh (SM)	1758		£83.63	£0.11 M
	Seagrass (SG)	768		£66.00	£0.05 M
	Reedbeds (RE)	1322		£290.53	£0.08 M
	Native oyster (*Ostrea edulis*) (OY)	238		£28.80	£0.01 M
	Total C	24075		Total C	£1.44 M
	SEMS Total	**27978**		**SEMS Total**	**£1,143.48 M**

4. Discussion

4.1 Mapping natural capital stocks

Improvements in Earth Observation (EO) technologies have provided maps of unprecedented detail and spatial extent (see Strong *et al.,* 2019). The SEMS is one of the most studied systems in Europe, for example White (2019) collated over 57 studies using traditional macrobenthic sampling methods from the 1970's onwards. However, despite the scientific interest and EO improvements, many coastal biotopes are still difficult to map accurately. This reflects unresolved technical and logistical challenges combined with financial constraints for the local practitioners in collecting data. The use of the UKSeamap which uses a combination of physical data with information from biological sampling returns a high JNCC confidence score (4) because it is accompanied by an accompanying confidence map, whilst frequent aerial imagery collection in combination with satellite and ground truthing are used by the Environment Agency for macroalgal mat extent and density (JNCC 4). In contrast, low JNCC confidence scores (1-2) for seagrass *(Zostera spp.)* and native oyster reefs reflect significant spatial and temporal data gaps in the region. Technical limitations (e.g. *Zostera spp.* meadows cannot yet be easily delineated from other vegetation using EO imagery) mean they have to be sampled directly (by boat or walkover); inevitably reducing coverage. Other biotopes are also challenging for EO due to their sublittoral location. For example, the low confidence score for *Ostrea edulis* populations is due to a reliance on vessel point sampling for stock assessments (Gravestock 2016). The process of undertaking this assessment revealed a lack of accurate and reliable baseline data against which to assess changes in marine NC stocks even in an area such as the Solent which is highly studied. Similarly, the total extent of several biotopes 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 (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 (Maes *et al.,* 2013; 2018) to support full NC assessments in coastal regions.

There is also potential for the EUNIS classification process to focus at smaller spatial scales. Here, we 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, P and C 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 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 (e.g. disaggregate to A2.8 features of littoral sediments) related to the different functioning of these biologically mediated biotopes. Other EUNIS biotopes are also important to water quality and/or climate regulation. 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 assets available in a region would be important to allow the value of all biotopes to be considered in any future management decisions.

4.2 Biophysical rates

Alongside mapping and monitoring, the sensitivity surrounding the metrics used for assessing the flows of ES from stocks of NC also warrants discussion (Bright *et al.,* 2019). Even within the constraints of this analysis, there was considerable variation in methods used across studies. Fluxes of N_2, NO_2, PO_4^{3-}, CO_2 and CH_4 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 which in turn contribute to the ES of waste remediation and climate regulation (Hattam *et al.,* 2015). 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 the two ES by presenting upper and lower reported bounds alongside the mean and median values. The median and range values together provide important information regarding 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 practitioners 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.

The biophysical rates selected in this study generally indicate higher N and P sequestration rates (per m^2) in oyster reefs, coastal saltmarsh and seagrass meadows and higher C burial rates in reedbeds (*Phragmites australis*) and coastal saltmarsh. When considered at the catchment level, saltmarsh and seagrass biotopes were often the largest removers of some nutrients (e.g. saltmarsh biotopes contributed the largest removal of N in Lymington Harbour, the Beaulieu estuary, Pagham Harbour and Yarmouth estuary, Figure S2). Yet, when considered at the level of the whole SEMS intertidal and subtidal sediment systems (including those with macroalgae) were the most important biotopes for N and C removal while vegetated biotopes were generally more important for P removal. This is largely a consequence of the large area of sediment systems in the case of N and C accounting for 84% of the SEMS biotope area. Our findings therefore corroborate the focus to date on restoration and preservation of structured coastal biotopes such as salt marshes, seagrass and oyster reefs in temperate estuaries (e.g. van Katwijk *et al.,* 2015; Helmer *et al.,* 2019). Our findings also support evidence that because of their large area within coastal systems, intertidal and subtidal sediments can provide disproportionately large contributions to N and C removal in coastal systems (e.g. Piehler & Smyth 2011; Eyre *et al.,* 2016). These considerable differences in relative biotope contributions to nutrient removal illustrate how ratios of functionality for different biotopes dictate potential gains (or losses) in ES production depending on how they are analysed and delineated over large spatial scales.

The proportion of N, P and C that can be considered removed for the purposes of water quality or climate regulation will also depend on the material's fate and the time scale of interest (Beaumont *et al.,* 2014; Kellogg *et al.,* 2014). Our study has considered two of the primary mechanisms for N, P and C sequestration 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, P or C assimilated into oysters' shell, 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 oysters 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, P and C 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. N_2 and N_2O removed *via* denitrification is likely permanent, and beneficial from a water quality management perspective, but coastal margin biotopes may also emit other greenhouse gas (GHG) emissions such as CO_2 and methane (CH_4) to an unknown extent, potentially reducing the net C burial benefit. CH_4 emissions have previously been thought to be negligible in temperate saltmarsh biotopes (e.g. Adams *et al.,* 2012; Ford *et al.,* 2012), but more recent evidence suggests they can be locally high in some biotopes e.g. seagrass (Oreska *et al.,* 2020).

There are also other sequestration mechanisms that need to be considered in more detail, for example anammox and DNRA are important nitrate reduction processes in estuarine sediments (Kessler *et al.,* 2018). Denitrification, anammox and DNRA are thought to compete for available NO_3^- which indirectly will influence the amount of N_2 released or stored in sediments. However, the relationship between these processes remains poorly understood and, in all the studies reviewed here, the authors did not measure anammox or DNRA's relationship with N burial or denitrification. More research to understand the links between anammox, DNRA and dentification competition would begin to disentangle the overall effect of limiting N loss and increasing N retention in biotopes. Moreover, the range of biophysical rates collected in our review for certain biotopes — such as native oyster reefs — highlights the need to clarify the influence of local environmental factors and biogeographically relevant taxa (e.g. tidal regime, substrate, life history and climate factors) on denitrification and burial rates in order to refine our understanding of the role of these biotopes play in different regions.

For vegetated and angiosperm-based biotopes, there is increasing evidence that a large proportion of the N, P and C 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, P, C early in the growing season, to a net source of nutrients in late summer when productivity declines (e.g. Tyler *et al.,* 2001; Gao *et al.,* 2013). Intertidal macroalgal mat sediment systems may, therefore, be considered as temporary stocks of N, P and C, 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). Few studies have verified the potential role macroalgal mat N, P and C plays in both coastal and offshore food webs, limiting the inclusion of nutrient export and long-term storage as a removal mechanism in our calculations. Based on evidence of other macroalgal species (e.g. the brown seaweed (*Himanthalia elongata*) it could be deduced that as much as 22-36% (Queirós *et al.,* 2019) of the N and C calculated here *via* burial mechanisms could be released and re-exported from coastal regions as particulate organic N and C. Further measurements of dissolved organic nutrient production generated by macroalgal mat and other angiosperm biotopes, could further increase the global significance of the sequestration fluxes we estimate.

4.3 Economic values

NC accounts can provide a valuable support to policy making because they can help set standards and objectives and can give a measure of the effectiveness and cost-efficiency of the policies aiming at reducing pollution or improving the state of water bodies (Russi and ten Brink 2013). The range of economic values calculated in this study are designed for use around the UK's southeast coast to capture the current monetary value of coastal biotopes for maintaining water quality with respect to removing N and P and mitigating climate change *via* the removal of CO_2. Our analysis indicates that N uptake provides the highest value of the ES assessed (equivalent to £962 million yr^{-1}, based on median biophysical values and mid economic prices or 84.2% of total value). This is largely due to the fact that on a per unit basis, N abatement costs are currently of much higher value than C sequestration and its potential benefits are more localized. The current findings are consistent with those presented by Russel *et al.,* (2013), who documented biotope replacement cost N values from $24 million per year to $3 billion per year in the Tampa Bay watershed USA. Additionally, with regards to P, estuaries are generally heterotrophic and therefore their sediments often represent a net source of P to the ocean (e.g. Deborde *et al.,* 2007) 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 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.

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 catchments 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 angiosperm-dominated stands of vegetation and green macroalgal mats were the most efficient at removing N, P and C but, no single biotope was the best at removing all three. This information can aid in the discussion of trade-offs between different aspects of NC (and associated beneficiaries) when different policy and development options are considered (Keith *et al.,* 2017). 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. Therefore, truly robust regional replacement cost values would require more nuanced quantitative adjustments that consider the baseline condition of the resource and also the willingness to pay of beneficiaries to implement the replacement of the NC asset with another asset, either natural or manmade.

Other ES provided by coastal biotopes, such as sediment retention, the production of harvestable fish and invertebrates, and dampening of storm surges could also add significant ancillary biotope-related value to both local and globally connected human beneficiaries beyond those estimated here. Thus, the next step in improving the valuation approach could be to use targeted catchment modelling or indicator assessments to better understand the full range of ES provide by EUNIS biotopes (Vermaat *et al.,* 2016; Rees *et al.,* 2019), the potential risks to those biotopes (e.g. *via* the development of NC risk registers (*sensu* Mace *et al.,* 2015) 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 and a smaller but no less important benefit for the global community in terms of C sequestered.

4.4 Which baseline? The role of biotope changes in natural capital assessments

Establishing baselines of past biological condition is often a key issue for conservation (Papworth *et al.,* 2009). Therefore, applying comparable biophysical and economic metrics to historic changes' can be useful to underscore society's impact and consequent changes in ES at the regional level and show to policymakers what can be achieved. Our study has generated valuations designed for informing current and future policy decisions, but these are set against a backdrop of substantially shifted (and shifting) global baselines of coastal biotope extent and quality (e.g. Millennium, 2005; Deegan, *et al.,* 2012; Waycott, *et al.,* 2009; Krumhansl *et al.,* 2016; Murray *et al.,* 2019). For example, in the UK alone native oyster reefs, and seagrass meadows are estimated to have declined by at least 90% (Airoldi and Beck, 2007; Mackenzie *et al.,* 2007; Beck *et al.,* 2011) and 25-49% (Jones and Unsworth, 2016), respectively; while globally 25-50% of saltmarsh biotope is estimated to have been lost (Mcowen *et al.,* 2017). These dramatic and rapid changes are also replicated at the regional level, with a 65% loss in saltmarsh between 1946-2013 and an estimated 98% reduction in native oyster landings between 1978-2013 (Key & Davidson, 1981; Pogoda, 2019), recorded within the SEMS (Figure 2 [a & b]). The impact of these declines on regulatory services is almost certainly significant. For example, using the mean biophysical values in Table 2 and assuming no temporal change in denitrification and burial rates, the saltmarsh biotope loss across five of the main SEMS catchments results in a total reduction in service provisioning per year of approximately 520 tonnes of N, 68 tonnes of P and 3033 tonnes of C. Monetary values associated with these losses (applying average N and P replacement costs and mid DECC (2011) non-traded carbon value) are: £153 million (N), 19 million (P) and 0.18 million (C) per year. Although the cumulative values are likely substantial (e.g. £ billions in replacement and abatement costs), caution is required during interpretation. Firstly, historic estimates using current biophysical and monetary values may fail to capture historic contextual variables e.g. the social cost of nitrogen abatement. More importantly, the saltmarsh biotope will have been replaced with an alternative biotope (e.g. littoral sediment). Therefore, EUNIS biotope *substitutions* should be accounted for in any historical trend analysis to derive true net changes as such replacements could reduce or improve N, P or C removal processes. For instance, there have been large reductions in the extent of macroalgal mats in the Solent between the late 1990's and 2019 (Figure 2[c]). Assuming that these biotopes have reverted to bare littoral sediment, the 898-hectare replacement of littoral mudflat covered in green macroalgae has resulted in an 53% decrease in the N removal capacity of the intertidal soft sediment environment (removing around 105 fewer tonnes of N yr^{-1}). Thus, our data suggests that as the Solent recovers from eutrophication - the regulatory value of NC will therefore continue to decrease unless these areas are re-populated by biotopes with higher rates of bioremediation.

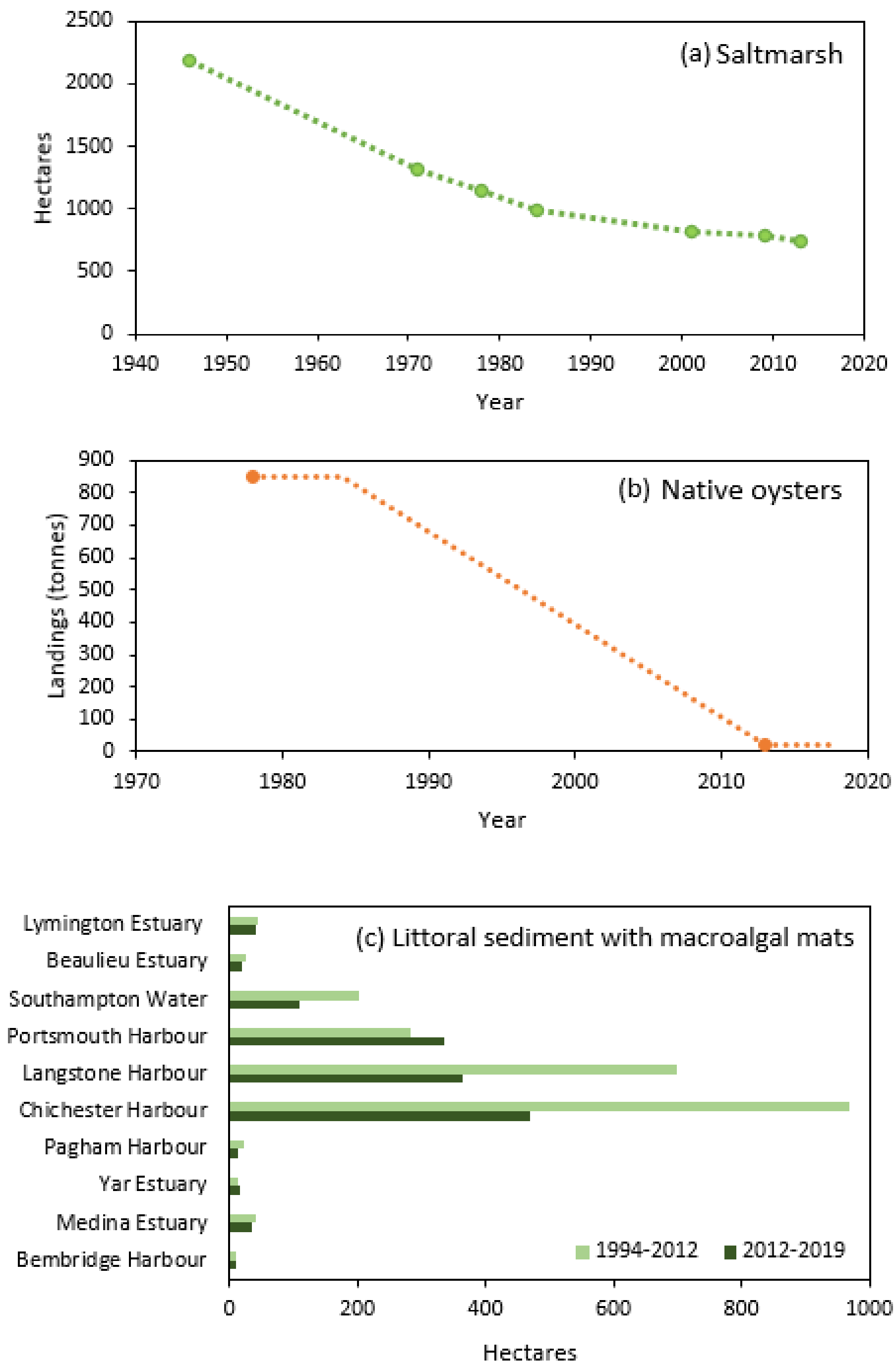

Figure 2 Historic changes in SEMS biotopes (3a) saltmarsh (3b) native oyster landings (3c) littoral sediments with macroalgal mats. Historic saltmarsh data (ha yr^{-1}) were sourced from: Haskoning (2004), Cope *et al.,* (2008) and combined for Lymington, Southampton, Portsmouth, Langstone and Chichester. Littoral mudflat with macroalgal mat comparison data (ha yr^{-1}) were sourced from the Environment Agency. Native oyster landings were predicted using a moving average based on data from (Key & Davidson, 1981; Pogoda, 2019).

5. Conclusions

In this study we used a three-step natural capital (NC) approach to estimate the capacity of temperate coastal biotopes to remove nitrogen (N) and phosphorus (P) and carbon (C) using the SEMS as a case study. Our approach considers the value of these three ecosystem functions by aggregating information on 1) the current extent of NC stocks using the EUNIS mapping framework 2) the biophysical rates that contribute toward the long-term removal of nutrients and 3) the monetary values associated with the resulting benefits. Even though we were not able to quantify all the different aspects of water quality and climate regulation, the available estimates indicate that the regulatory services proved by the Solent's coastal and nearshore biotopes are substantial (millions to billions of UK £) and in many circumstances could provide relatively cost-effective investment alternatives (e.g. NC restoration projects) to engineered solutions for maintaining healthy ecosystems. Our data provide a strong rational for investing in the natural capacity of such systems to reduce the impacts of N & P loading and offset C emissions back to the atmosphere. The estimates presented here reflect the best currently available information, but to further enhance the utility of a EUNIS approach to NC management and fully support the development of sustainable and relevant environmental policies to protect and enhance coastal biotopes we recommend:

- Using fine scale data (e.g. EUNIS Level 4 and lower) to create marine and coastal NC accounts, thus integrating benthic community function.
- Incorporating other dimensions of NC asset status (e.g. condition) would likewise be helpful in understanding how biophysical rates are affected by changes in the condition of biotopes.
- Explicitly including sediment covered with green macroalgal mats as separate biotopes for EUNIS NC assessments and, by extension in water quality accounting reports and in restoration strategies to mitigate climate change.
- Further research effort should be directed towards determining the mechanisms and ecological functions that underpin N, P and C processes in understudied species (e.g. *Ostrea edulis*).
- Adjusting historical and future accounting assessments for biotope replacements rather than just loss, including a consideration of the substitutability between ES provided by EUNIS biotopes.

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