



Bivalves at Work: Quantifying Nutrient Removal Services in UK Coastal Waters

Konstancja Woźniacka^{1,2} · Heather Moore³ · Eunice Pinn^{4,5} · Suzanne B. Bricker⁶ · Alhambra M. Cubillo⁷ · João G. Ferreira^{7,8} · Matthew Service³

Received: 11 July 2025 / Revised: 12 February 2026 / Accepted: 28 February 2026
© The Author(s) 2026

Abstract

Nutrient loading from agriculture, urban discharge, and industrial effluents contributes to eutrophication, a critical environmental challenge impacting water quality. This research evaluated the role of bivalves in nutrient removal as a bioremediation tool under aquaculture practices in the UK, focusing on current key commercial species: blue mussels *Mytilus edulis*, Pacific oysters *Magallana gigas*, (formerly *Crassostrea gigas*), native oysters *Ostrea edulis*, and Manila clams *Ruditapes philippinarum*. The removal of nutrients by bivalve aquaculture was estimated based on the Nitrogen (N) and Carbon (C) fixed in their tissue and shell at the point of harvest using two complementary methods: (i) proximate analysis based on nutrient content in shellfish tissue and shell (for N and C), and (ii) using the FARM model (N only). The monetary value of these services was estimated through comparison to costs of alternative mitigation methods. The results show that the UK bivalve aquaculture sector contributed to the removal of an estimated ca. 127 to 286 tonnes of N in 2019. The harvested bivalves also fixed an estimated ca. 1763 tonnes of C in their tissue and shell. Mussels accounted for the majority of N removed, ca. 116.7 to ca. 264.5 tonnes, due to higher yields. Together, oysters and clams potentially removed an estimated ca. 9.9 to ca. 21.9 tonnes of N. The potential annual value of N removal from the UK bivalve aquaculture sector ranged from £33,000 to over £314 million, depending on estimated harvest and type of alternative removal measure. The results of this work highlight that the expansion of UK bivalve aquaculture could deliver water quality improvements, enhance ecosystem services gains, provide economic benefits and strengthen its role as part of holistic nutrient management strategies.

Keywords Bivalve aquaculture · Ecosystem services · Nutrient bioextraction · Ecosystem management · Eutrophication · FARM model

Communicated by David Gillett

✉ Eunice Pinn
eunice.pinn@nature.scot

¹ Seafish, 18 Logie Mill, Logie Green Road,
Edinburgh EH7 4HS, UK

² Biosciences, University of Exeter, Exeter, UK

³ Agri-Food and Biosciences Institute (AFBI), 18a Newforge
Lane, Belfast BT9 5PX, UK

⁴ NatureScot, Inverdee House, Baxter Street,
Aberdeen AB11 9QA, UK

⁵ School of Law, University of Aberdeen,
Aberdeen AB24 3UB, UK

⁶ Retired. Formerly NOAA NOS NCCOS Cooperative Oxford
Laboratory, Oxford, MD 21654, USA

⁷ Longline Environment Ltd, 63, St. Mary Axe,
London W1G 8TB, UK

⁸ DCEA, Faculdade de Ciências e Tecnologia, Universidade
Nova de Lisboa (NOVA), Quinta da Torre,
Monte de Caparica 2829-516, Portugal

Introduction

One of the foremost causes of coastal and estuarine water quality degradation is eutrophication, resulting from nutrient pollution, which is expected to increase throughout the 21st century due to climate change and human population increase (Dai et al., 2023; Rabalais et al., 2009; Sinha et al., 2017). While current eutrophication management frameworks focus on bottom-up control through reduced land-based nutrient inputs, the growth of coastal populations, increased resource consumption, and difficulties experienced in monitoring and management strategies can make this approach insufficient (Boesch & Brinsfield, 2000; Boesch, 2019; García-García et al., 2019). This highlights the growing need for more nature-based solutions that can supplement traditional mitigation methods.

There has been a surge of interest in bivalve-mediated nutrient bioextraction, and how it can be incorporated into nutrient management measures (Bricker et al., 2018, 2020; Cornwell et al., 2016, 2023; Cubillo et al., 2023; Ferreira & Bricker, 2016; Jennings et al., 2016; Petersen et al., 2012; Pradeepkiran 2019; Reitsma et al., 2017; Rose et al., 2015, 2021; Taylor et al., 2019) whilst still providing a sustainable protein source. Whilst the bivalves filter-feed, they ingest and fix organic forms of nutrients (e.g. phytoplankton, detritus), which make up their tissue and shell (Cubillo et al., 2023). Bivalve biomass is extracted from the sea at harvest, which is an important consideration when comparing cultured to wild populations and when assessing their ecosystem services, especially when considering carbon balances and that the shell of wild populations will eventually dissolve in the sea, as opposed to harvested bivalves which shells can remain on land (Filgueira et al., 2019). A review of worldwide studies between 1980 and 2011 summarised that bivalves can remove between 1% and 15% of total annual nitrogen (N) loads, depending on the density, nutrient input levels, and local environmental conditions (Carmichael et al., 2012). For Manila clams (*Ruditapes philippinarum*) in Jiaozhou Bay, China, the removal of external N and phosphorus (P) input was estimated at 28.7% and 43.3% respectively (Zan et al., 2014). More recently, a study of six bays in New Brunswick, Canada proposed that mussels and oysters could remove between as little as 0.1% to as much as 86.2% of the total annual N load (Clements & Comeau, 2019). An assessment of N removal in Europe estimated that in 2018 bivalve aquaculture was able to remove between 0.12% and 0.17% of the total N loads (Cubillo et al., 2023). The potential of bivalve bioremediation and their usage in nutrient trading has been explored as early as the Lysekil trial (2005–2011) where mussels were used to compensate for the discharge of N from a sewage treatment plant (Lindahl, 2011; Lindahl & Kollberg, 2008,

2009). Research across the Danish fjords further highlighted the nutrient extractive potential of bivalves and that high removal efficiency could be achieved with mussel mitigation cultures (Petersen et al., 2014; Taylor et al., 2019). In the wider Baltic Sea, mussels were shown to be a cost-effective measure for nutrient bioextraction and a complement to the land-based nutrient management measures (Kotta et al., 2020; Petersen et al., 2020). The value of oyster aquaculture in a water quality improvement capacity has also been recognised, and lead to the potential inclusion in nutrient credit trading programmes, where growers can be compensated for the N removal provided by their oyster farms (Ferreira & Bricker, 2019; Parker and Bricker, 2020; Rose et al., 2021).

The UK has committed to the sustainable development of the blue economy and the implementation of nature-based solutions (NBS) under the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR), the Convention on Biological Diversity, and the UK Marine Strategy. In this context, shellfish cultivation could be promoted as a NBS and contribute to the safeguarding of the UK's marine ecosystems' natural capacity to sequester nutrients. Furthermore, the UK has ranked in the top 20 oyster and mussel producers in the world and the top 10 in Europe between 2015 and 2021, with the bivalve aquaculture industry valued at nearly £29 million in 2022 (FAO, 2023 c; Seafish, 2023b, 2023a; Wijsman et al., 2019), demonstrating potential for bivalve nutrient mitigation research. However, though the concept of bivalve bioextraction is gaining support, translation into policy has been slow. Many questions remain regarding national capacity for bivalve nutrient removal, the cost and logistics of implementing a nutrient credit trading (NCT) system, societal acceptance of the approach and resulting seafood products, and the financial benefits to be gained.

To address some of these questions, in this work, we (i) summarised the nitrogen loadings into the UK waters, (ii) evaluated the scale of nitrogen and carbon removal services by key commercially farmed bivalves (Pacific oysters, Native oysters, Manila clams, and blue mussels) for the UK as a whole and individually for England, Wales, Northern Ireland, and Scotland using two complementary methods, and (iii) assessed the monetary value of the nitrogen removal services provided by the UK bivalve aquaculture in water quality improvement.

Methodology

Bivalve Aquaculture Production in the UK

Bivalve aquaculture production data on a national and regional scale was sourced from Cefas (Cefas, 2023, pers.

comm.) and Marine Scotland (Scottish Government, 2023) for years 2013 to 2020. Commercial production in the country focuses on blue mussels (*Mytilus edulis*) and Pacific oysters (*Magallana gigas*) while other species, such as scallops (*Pecten maximus*, *Aequipecten opercularis*), cockles (*Cerastoderma edule*), native oyster (*Ostrea edulis*) and Manila clams (*Ruditapes philippinarum*) are commercially harvested from wild stock, some of which are managed stock. Cultivation methods include bottom culture, suspended culture, subtidal and intertidal culture (off-bottom intertidal, trestles), with farms ranging from small family-run operations focused on local markets to larger commercial enterprises, reaching international markets.

Here four key commercial bivalve species that represent over 90% the UK bivalve production were considered: blue mussel, Pacific oyster, native oyster, and Manila clam (Table 1). For most recent estimates of production used in the nutrient removal analysis and comparison of nutrient removal between the species and countries, year 2019 data are used due to considerations that the 2020 data were not representative of the general activities in the sector due to influences of the Covid-19 pandemic on the operations of farms and market demand.

Nitrogen Loading in the UK Waters

The primary sources of nutrient influx into water bodies include agricultural activities (fertilizers, manure and slurry runoff), urban runoff (stormwater runoff, pet waste, domestic wastewater), inadequate wastewater treatment systems, and industrial discharges (manufacturing and mining).

The N loadings to UK waters (Celtic Seas, Channel, Atlantic, Irish Sea and North Sea) between 2000 and 2021 were sourced from the OSPAR RID dataset (OSPAR, 2023). These loadings were categorised by source into point and diffuse sources. Point sources comprise industry effluents

(wastewater discharges), sewage effluents (domestic and municipal wastewater), and aquaculture discharges (mainly from fish farming), and were summarised in Total Direct Discharges (TDD). Diffuse sources include all riverine inputs, both monitored and unmonitored, that transport contaminants from various origins within a river's catchment and discharge them into tidal waters. These inputs are classified as Total Riverine Inputs (TRI), while the TDD refer to contaminants that are discharged directly into the sea without passing through a river system before entering the sea. These categories followed the data classification utilised by OSPAR in their data collection. After initial handling, considerable gaps in the dataset were identified as a result of changes in sampling and record-keeping across the different administrations, as well as interruption caused by COVID-19 (2020–21). As such, the final analysis made use only of 2014 data (Table 2). This was the most recent year with most regions reporting their total nitrogen loads. Although TRI and TDD were recorded for nearly every year between 2000 and 2021, the individual sources (sewage, industry, and aquaculture effluents) were not always provided. This was particularly evident for aquaculture discharges and reports for Welsh waters, which recently comprised only the information on TRI. The most recent data (2015 onwards) was subject to reduced reporting spatially in England and Wales, and to changes in the reporting methods for Northern Ireland, making them unsuitable for this analysis.

Nutrient Removal Methodology

To assess the potential of bivalves for nutrient removal, various methods can be employed, including field and laboratory studies, historical data analysis, and modelling. While laboratory and field studies provide precise measurements, they are limited in scale and often costly. One such approach, elemental analysis (also referred to as proximate analysis, PA) of tissue and shell, allows for upscaling estimates to the harvested population (Kotta et al., 2020; Olivier et al., 2021). However, it does not account for the contribution of bivalves to nitrogen removal by enhancing denitrification (the microbial driven process of bioavailable nitrogen transformation to di-nitrogen gas).

To address the limitation of laboratory and field studies, modelling approaches can extend these estimates to broader spatial scales and include multiple species. Among these, the Farm Aquaculture Resource Management (FARM) model has been widely applied to simulate aquaculture production and nutrient removal at the farm scale in various settings (Bricker et al., 2020; Cubillo et al., 2023; Dvarskas et al., 2020; Ferreira et al., 2007; Parker & Bricker, 2020; Rose et al., 2015; Silva et al., 2012). By integrating physical, biogeochemical, growth, and eutrophication

Table 1 Bivalve species considered in the proximate analysis with the percentage content of nitrogen (N) and carbon (C) in their shell and flesh (% of live weight, in fresh mass) presented originally in the Green Aquaculture Intensification in Europe (GAIN) project report as mean elemental composition

Species	Latin name	Origin	% of N in TFW	% of C in TFW
Blue mussel	<i>Mytilus edulis</i>	Belfast Lough (Northern Ireland)	0.88	11.1
Pacific oyster	<i>Magallana gigas</i>	Dundrum Bay (Northern Ireland)	0.37	10.9
Native oyster	<i>Ostrea edulis</i>	Lough Foyle (Northern Ireland)	0.29	8.80
Manila clam/ Japanese carpet shell	<i>Ruditapes philippinarum</i>	Venice Lagoon (Italy)	0.32	8.50

Table 2 Nitrogen (N) loadings and sources for England, Northern Ireland, Scotland, and Wales for 2014*. Loadings are provided in 103 tonnes y⁻¹. Total Direct Discharges are the sum of Aquaculture discharges, Industry and Sewage effluent. Total of all inputs represents the sum of Total Direct Discharges and Total Riverine Inputs (monitored and unmonitored)*The data in the table should be considered with caution. Although it is supplied by OSPAR RID, the data is collected by many bodies across the UK and collated on many levels, which could introduce error. Some nations do not collect and report data on certain sources which is reflected in the gaps within the table and the graphs. Moreover, certain regions are not always included in the reporting, depending on the yearly monitoring capabilities of the country, as such not all OSPAR regions might be included in this summary

Country	Source	N-Total (10 ³ tonnes y ⁻¹)
England	Industry effluents	1.20
	Sewage effluents	35.35
	Total Direct Discharges	36.25
	Total Riverine Inputs (monitored and unmonitored)	229.84
	Total of all inputs	266.09
Northern Ireland	Industry effluents	0.001
	Sewage effluents	0.96
	Total Direct Discharges	0.99
	Total Riverine Inputs (monitored and unmonitored)	8.22
	Total of all inputs	9.21
Scotland	Aquaculture discharges	11.11
	Industry effluents	2.11
	Sewage effluents	12.21
	Total Direct Discharges	25.32
	Total Riverine Inputs (monitored and unmonitored)	46.99
	Total of all inputs	72.31
Wales	Industry effluents	-
	Sewage effluents	0.99
	Total Direct Discharges	1.39
	Total Riverine Inputs (monitored and unmonitored)	20.17
	Total of all inputs	21.56
UK	Total Direct Discharges	63.94
	Total Riverine Inputs (monitored and unmonitored)	305.22
	Total of all inputs	369.16

models, FARM assesses the physiology, growth, and harvest of bivalve populations at the local (farm) scale (Ferreira et al., 2007). The model also assesses changes in the eutrophication status at the farm, through a mass balance of intake (filtration) and loss (faeces, pseudofaeces, excretion, and mortality) of N in organic material by the modelled population. FARM estimates nutrient removal by explicitly simulating bivalve physiology and goes beyond a harvest-based approach by determining net N removal of the whole cultivated population. Similarly to the proximate analysis, FARM estimates do not include removal through denitrification processes.

Despite its mathematical simplicity, upscaling elemental analysis can help identify shellfish operations with high bioremediation potential and determine the minimal harvest required for water quality improvement. To better estimate N removal capacity, both approaches can complement each other. Therefore, this analysis employs both proximate nutrient removal estimates and FARM model outputs, upscaled to current UK production levels, to provide a comparative range of nutrient removal potential from lower-bound to upper-bound estimates (Tables 3 and 4).

Proximate Analysis

The first approach in estimating N and C removal by bivalves was through a function of their weight and N or C content. The elemental composition of the bivalves, sampled between May 2019 and January 2021, was procured from combustion elemental analysis performed as a part of the Green Aquaculture Intensification in Europe (GAIN) project (Cubillo et al., 2023; Ferreira et al., 2020). In the process, the elemental composition is determined for a pre-weighed sample by flash combustion and separation of the resulting gaseous products. The full description of the methodology can be found in the project's report (Cubillo et al., 2023; Ferreira et al., 2020). The estimated nutrient removal (both N and C) refers to the amount of the nutrient fixed in bivalve body (tissue and shell) at the point of harvest. In the context of C removal, it is important to note that the contribution of bivalves to atmospheric CO₂ sequestration is still debated in literature, as during calcification the inorganic carbon removed is not the CO₂, and as such bivalves may not consistently contribute to climate change mitigation (Álvarez-Salgado et al., 2022; Chen et al., 2025; Jansen & Van Den Bogaart 2020; Pernet et al., 2025a, b). To calculate

Table 3 Nitrogen and carbon removed (tonnes in 2019) by each species in the UK in their tissue and shell, estimated through scaling up of proximate analysis and FARM results based on production from 2019 reported by Cefas. For Pacific oysters, the estimates are available for diploid and triploid organisms (t)

Proximate analysis										
Country	Blue mussel		Native oyster		Manila clam		Pacific oyster		Total N	Total C
	N	C	N	C	N	C	N	C		
England	25.90	326.73	0.03	0.96	0.02	0.60	4.51	132.94	30.47	461.22
Northern Ireland	6.12	77.15	-	-	-	-	3.89	114.61	10.01	191.76
Scotland	58.95	743.59	0.02	0.70	-	-	1.30	38.26	60.27	782.55
Wales	25.75	324.79	-	-	-	-	0.07	2.18	25.82	326.97
UK Total	116.72	1472.25	0.05	1.66	0.02	0.60	9.78	287.99	126.57	1762.49

FARM						
Country	Blue mussel	Native oyster	Manila clam	Pacific oyster	Total N	
	IBM	IBM	IBM	IBM	IBM	
England	58.69	0.10	0.08	9.43–10.00 (t)	68.31–68.87 (t)	
Northern Ireland	13.86	-	-	8.13–8.62 (t)	21.99–22.48 (t)	
Scotland	133.57	0.07	-	2.71–2.88 (t)	136.36–136.53 (t)	
Wales	58.34	-	-	0.15–0.16 (t)	58.50–58.50 (t)	
UK Total	264.46	0.18	0.08	20.43–21.66 (t)	285.15–286.39 (t)	

Table 4 Nitrogen loadings (tonnes) and nitrogen loading per unit area of the country (kg km⁻² y⁻¹) in the UK waters (2014) and nitrogen removal (tonnes) by each of the countries (with average percentage of the total nitrogen loads for the respective UK waters in brackets) calculated through proximate analysis (PA) and FARM software (t stands for estimates where triploid oysters were used for Pacific oyster instead of diploid oysters) using 2019 production data. Population equivalent (PEQ) at 3.3 kg of Nitrogen per capita, based on average N-removal

Country	N total input	N per unit area	N-removal		PEQ	
			PA	FARM	PA	FARM
England	266 088.31	2041.96	30.47 (0.01%)	68.59 (0.03%)	9233	20,785
Northern Ireland	9 206.83	651.58	10.01 (0.11%)	22.24 (0.24%)	3033	6738
Scotland	72 305.60	928.18	60.27 (0.08%)	136.45 (0.19%)	18,264	41,347
Wales	21 561.33	1037.65	25.82 (0.12%)	58.51 (0.27%)	7824	17,729
Total	369 162.10	1518.44	126.57 (0.03%)	285.78 (0.08%)	38,355	86,598

removal on a national scale, the percentage value of N content in the live weight of each species (Table 1) was scaled up to the corresponding national production (Eq. 1):

$$N_{removed(p)} = N_{bivalves} \times Production_{National} \quad (1)$$

FARM Model Implementation

The second approach used was to provide the nutrient removal estimates made through the application of the Farm Aquaculture Resource Management (FARM) model. A full description of the modelling approach can be found in previous publications (Ferreira et al., 2007, 2020, 2021).

The model framework consists of (i) individual-based Net Energy Balance models for each bivalve species, called AquaShell, used to simulate physiological processes determining growth and the mass balance of substances of interest with respect to ecosystem services (i.e., nitrogen) for each animal, and (ii) an Individual-Based Model (IBM) population approach (see: Cubillo et al., 2023; Ferreira et al., 2021), with an advection–diffusion model for transport of water

properties, including settling of suspended particles. Both models use local water-quality data to simulate growth. The AquaShell model simulates physiological processes of feeding and energy intake, metabolic expenditure and ammonia excretion rates, as well as reproductive behavior for individual bivalves. The FARM model simulates processes at the local scale on a population level, i.e. at the typical farm (Ferreira et al., 2007). FARM inputs also include water current speeds and farm operational details (i.e., farm size, length of the culture cycle, seeding densities, etc.) to simulate growth, nutrient removal, and changes in water quality at the farm level. In the IBM version of FARM, each individual in the population is stochastically assigned a fitness parameter at seeding in terms of assimilation efficiency (AE, ±0–5% of the mean). The objective is to simulate the typical genetic variance in the cultivated population. This model simulates various attributes related to growth performance and interactions with the environment, such as food consumption, organic waste production, and mortality. After integrating the individual bivalve models into FARM, the production and environmental effects at the local scale of a farm are simulated over one culture cycle and the model estimates the

removal of suspended particles, particulate organic waste, excretion of dissolved nitrogen, and oxygen consumption, as well as removal of phytoplankton and detrital C and N by bivalves. The net removal is calculated by subtracting losses caused by pseudofaeces, faeces, excretion, mortality, and spawning. To better reflect real-world shellfish farm operations, the ‘Harvest When Ready’ option was selected for the model runs, so that the model is configured to harvest the shellfish as soon as the user-defined threshold weight is reached (i.e. precision aquaculture), rather than harvesting shellfish only at the end of the culture period.

The AquaShell models for the different shellfish species were parameterised in the framework of previous projects (e.g. Ferreira et al., 2007, 2009). Farm operational details - including information such as typical cultivation methods, average culture cycle duration, seeding densities, seed size and harvest size - were sourced from conversations with UK-based bivalve growers as well as literature. Environmental data required to run the simulations - including water temperature, salinity, chlorophyll concentration, particulate organic matter, total particulate matter and current speeds - were sourced from AFBI (2023), SMILE (Ferreira, 2007; Ferreira et al., 2007), Cefas (Morris, 2016), DAERA (2023), ICES (Gonzalez-Pola et al., 2023), Marine Scotland Science (Bresnan, 2016) and Copernicus Marine Services (CMEMS, 2023). Population dynamics in the simulation were defined by culture strategies using pre-defined “typical farms” selected to be representative of each species, thereby providing an estimate of potential removal each for UK jurisdiction (Table S1). In the case of Pacific oysters, model runs for both diploid and triploid oysters were performed as the use of unfertile triploid oysters is widely encouraged in the UK as means to prevent proliferation of this non-native species.

The results of the model were scaled up to the national production (for 2019) (Eq. 2):

$$N_{removed(m)} = \frac{Production_{national}}{TPP} \times N_{removal} \quad (2)$$

where $N_{removed(m)}$ represents the total N removed by the estimated live weight production in tonnes. $Production_{national}$ (tonnes) is the corresponding species production for each nation, while Total Physical Product (TPP) (tonnes) is the estimated FARM-modelled production for each species at the farm scale, and $N_{removal}$ ($kg\ y^{-1}$) is the N removed by each species at the local (farm)-scale.

Substitution Cost Analysis

The value of nutrient removal by shellfish farms can be estimated by placing it in context with other remediation approaches for water treatment. For the valuation, the

approach with the highest potential, due to its simplicity and wider availability of necessary data, is the avoided or replacement cost method, where the monetary value is estimated based on the cost of an alternative, least-cost mitigation methods and nutrient removal strategies such as stormwater control measures, chemical and manual wastewater treatment and approved agricultural Best Management Practices (BMP) (Barrett et al., 2022). By applying the cost required to remove one kilogram of N (£/kg N) by a known nutrient removal strategy to the amount of nutrient removed by bivalves, the cost savings achieved through bivalve bioextraction can be estimated. This information was sourced for a Dorset-based water treatment facility, in addition to costs related to alternative mitigation measures (Wessex Water, 2023; Table 5). Moreover, the costs for various aspects of N and eutrophication removal, monitoring, and mitigation measures were summarised to expand the valuation and showcase the potential range of savings.

Results

Shellfish Production

Scotland was the top shellfish producer in the UK for the 2013–2019 period, contributing between 26.3% (6,935 tonnes in 2013) and 44.3% (7,061 tonnes in 2019) of total shellfish production. A notable decline was observed for Wales, where the total shellfish production dropped from 31.6% (8,344 tonnes in 2013) to 18.5% (2,946 tonnes in 2019). England contributed between 29.0% (7,648 tonnes in 2013) to 26.2% (4,181 tonnes in 2019) of total shellfish production. Northern Ireland also experienced a decline in production, from 13.1% (3,464 tonnes in 2013) to 11.0% (1,747 tonnes in 2019), although the country did increase its Pacific oyster production during this period.

There was an overall decline in bivalve production in the UK, from 24,149 tonnes in 2015 to 15,936 tonnes in 2019. Between 2013 and 2019, blue mussel, Manila clam and native oyster production experienced an overall decline of 41.26%, 36.36% and 62% respectively. At the same time, Pacific oyster production increased by 112% (Figs. 1 and 2). However, while discernible patterns were evident for blue mussels, Pacific oysters and native oysters, production of Manila clam exhibited greater variability in the year-to-year yields.

Nitrogen Loadings in UK Waters

The overview of the total annual mean N loadings into the UK waters for 2014 are presented in Table 2. With consideration of the limitations and gaps in data records, there were

Table 5 Costs for various aspects of nitrogen and eutrophication removal, monitoring, and mitigation measures. Valuation is provided for the UK nitrogen (N) removal: 126.57 tonnes from proximate analysis (PA), and 285.77 FARM estimates

Issue addressed	Method	Cost	Source	Location	Valuation (PA)	Valuation (FARM)
Nitrogen	Methanol dosing; tertiary denitrifying sand filters	£58,300/tonne of N yr ⁻¹	Wessex Water, 2023	Poole Harbour	£7,378,051 yr ⁻¹	£16,660,391 yr ⁻¹
	Catchment management	£8,750/tonne of N yr ⁻¹			£1,107,487 yr ⁻¹	£2,500,487 yr ⁻¹
	Mixed methods	£0.26 kg ⁻¹ N	Brink et al., 2011 (adapted from Pretty et al., 2003)	England and Wales	£32,884	£74,300
	Change of agricultural land use to less intensive grass production through direct land purchase or Payments for Ecosystem Services schemes	£295–895 kg ⁻¹ N	Watson et al. 2020a, b, c	Solent	£37,351,150–113,325,150	£84,302,150 - 255,764,150
	Upgrades to existing wastewater treatment plants and associated drainage infrastructure	£282–1100 kg ⁻¹ N			£35,781,540–139,227,000	£80,587,140 - 314,347,000
	Application of Catchment Sensitive Farming measures	£5–23 kg ⁻¹ N			£632,850 - 2,913,110	£1,428,850 - 6,572,710
Other cost estimates						
Not Eutrophication	Treatment of algal blooms and in-water preventative measures	£0.5 m yr ⁻¹	Pretty et al., 2003	England and Wales		
	Adopting new farm practices that emit fewer nutrients	£3.39 m yr ⁻¹				
	Monitoring costs for water and air	£0.44 m yr ⁻¹				
	Developing eutrophication control policies and strategies	£0.2 m yr ⁻¹				
	Stormwater control measures	£2,914 kg ⁻¹ N	Rose et al., 2015	USA-based estimates		
Nitrate	Approved agricultural BMP	£374 kg ⁻¹ N				
	Wastewater treatment upgrades	£7047 kg ⁻¹ N				
Nitrate	Costs to the agricultural sector of complying with the NVZ regulations	£44 m - £65 m yr ⁻¹	Environment Agency, 2021	UK		
	Company-led catchment schemes to reduce nitrate concentrations	£95 m - £115 m yr ⁻¹				

still marked variations in nitrogen loadings in the UK waters. The biggest UK contributors to N loadings were England and Scotland. Based on N loadings normalised per unit of surface area of each country (Table 4), the highest loads per km² were attributed to England (2,042.0 kg km⁻² y⁻¹) and Wales (1,037.7 kg km⁻² y⁻¹). For the entire UK, riverine inputs (TRI) were higher than direct discharges (TDD) (305.2 × 10³ tonnes and 63.9 × 10³ tonnes, respectively), reaching a total of 369.2 × 10³ tonnes in 2014. Overall, in 2014, the highest N loadings were recorded for the North Sea South (106.7 × 10³ tonnes y⁻¹ for TRI), Celtic Sea (58.3 × 10³ tonnes y⁻¹ for TRI) and Irish Sea (44.8 × 10³ tonnes y⁻¹ for TRI).

Nutrient Removal Estimates

The estimated amount of N and C fixed in tissue and shell of bivalves at the point of harvest using the scaled-up proximate analysis are available in Table 3 as well as Figs. 3 and 4. Estimated N removal for both the proximate analysis and FARM IBM model are presented in Table 3 for the most recent year (2019). Mussels were responsible for most of the removal, followed by Pacific oysters. Based on the 126.6 tonnes of N removed in most recent year of the analysis (2019), mussels were responsible for 92.2% of the N removed by shellfish, followed by 7.7% removed by Pacific

Fig. 1 Total aquaculture production (tonnes) of UK bivalve species (based on blue mussel, Native oyster, Pacific oyster, Manila clam) for England, Northern Ireland, Scotland, and Wales together between 2013–2020. The red dashed line indicates 2019, after which the reported production might be biased due to Covid-19 pandemic and should be considered with caution

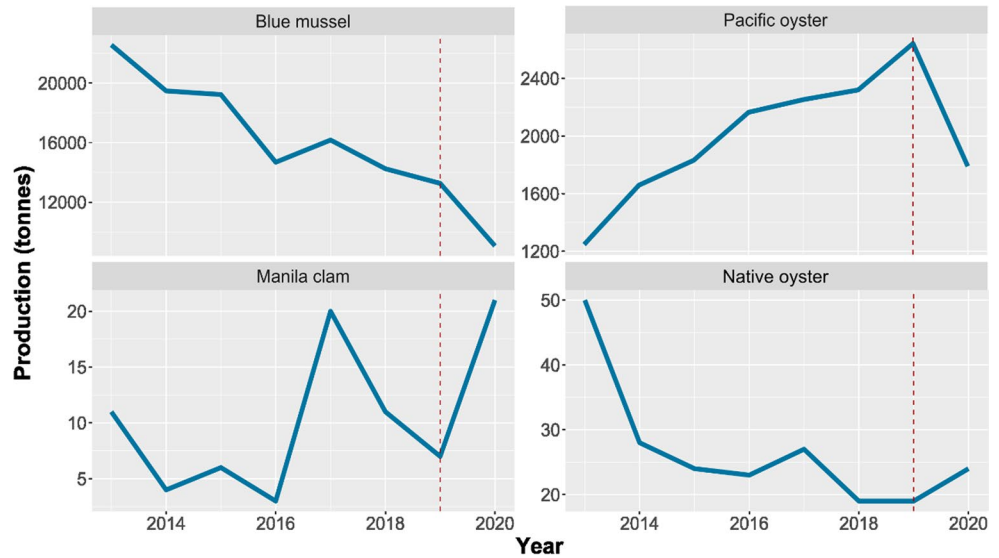
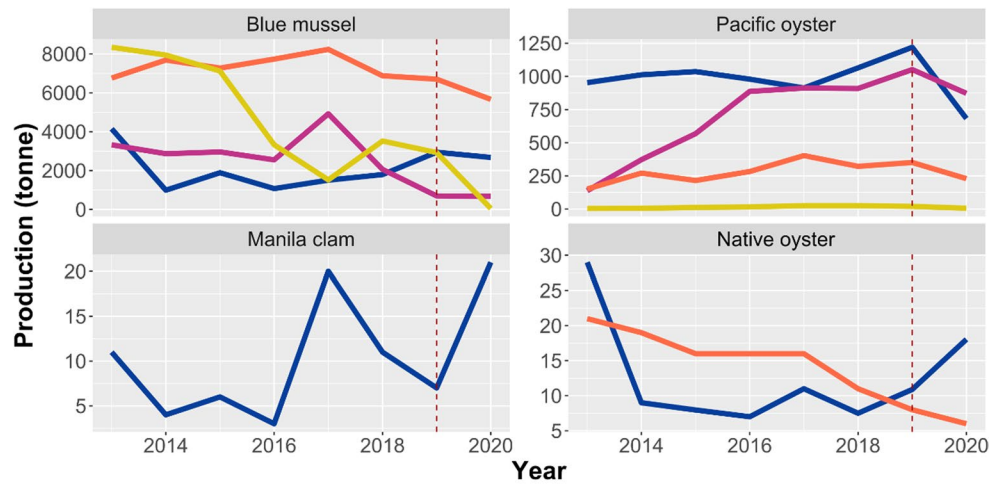


Fig. 2 Aquaculture production (tonnes) of UK bivalve species across the countries between 2013–2020 (note the different scales). The red dashed line indicates 2019, after which the reported production might be biased due to Covid-19 pandemic and should be considered with caution



Region — England — Northern Ireland — Scotland — Wales

Fig. 3 Total nitrogen removal (tonnes) offered by aquaculture harvest of UK bivalve species (based on blue mussel, Native oyster, Pacific oyster, Manila clam) for England, Northern Ireland, Scotland, and Wales between 2013–2020, estimated by proximate analysis (note the different scales). The red dashed line indicates 2019, after which the reported production might be biased due to Covid-19 pandemic and should be considered with caution

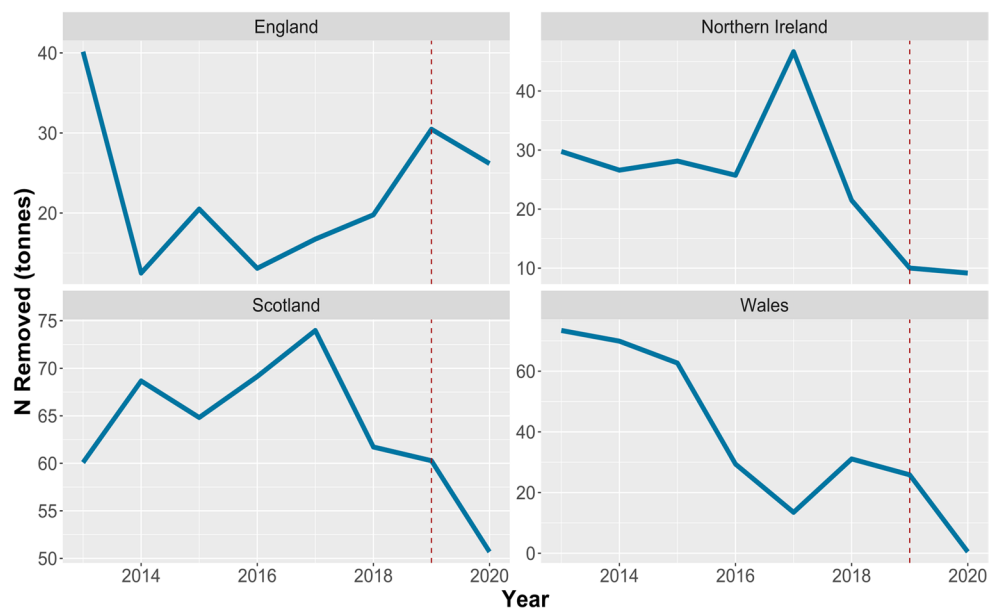
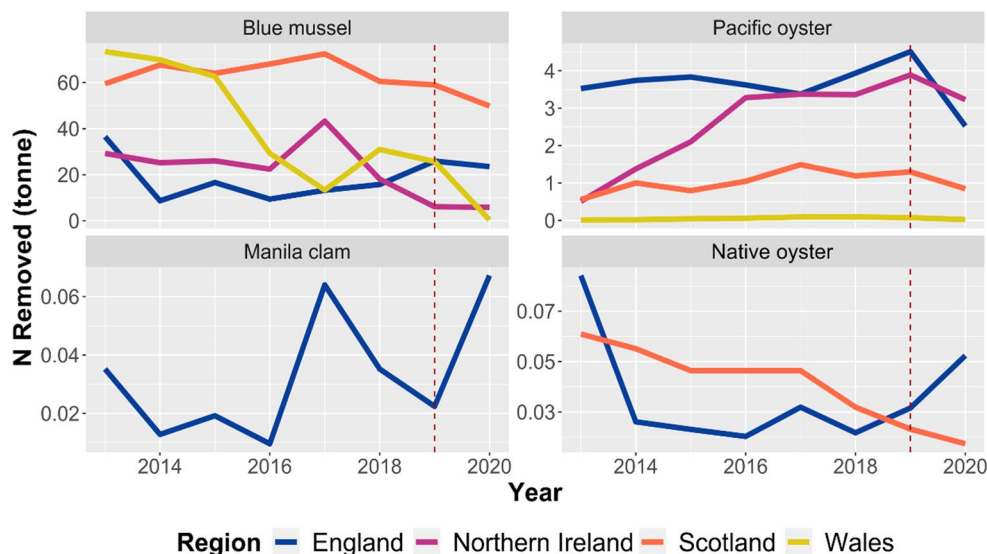


Fig. 4 Nitrogen removal (tonnes) offered by aquaculture harvest of UK bivalve species for England, Northern Ireland, Scotland, and Wales between 2013–2020, estimated by proximate analysis (note the different scales). The red dashed line indicates 2019, after which the reported production might be biased due to Covid-19 pandemic and should be considered with caution



oysters. The removal of C as part of bivalve tissue and shell was also estimated, reaching 1762.5 tonnes in 2019 (Table 3). Similarly, mussels dominated in terms of the proportion of total potential C removed by shellfish (83.5%), with Pacific oysters removing nearly five times less (16.3%). Scotland and England contributed 47.7% and 24.1% of N and 44.4% and 26.1% of C removed respectively. FARM estimated the removal of ca. 285.8 tonnes of N. In comparison to the proximate analysis, the estimated N removal is over two times higher for FARM outputs.

Mass Balance and Monetary Valuation

The comparison of estimated N removal for 2019 and N loadings that enter the marine environment for 2014 in UK waters is available in Table 4. Bivalve aquaculture was estimated to remove between 0.03% and 0.08% (FARM) of the N loadings in the UK waters. In terms of removal of their individual N loads, Wales (0.12% and 0.27% (FARM) of N loading into local waters) and Northern Ireland (0.11%; 0.24% FARM) have shown the biggest potential. England removed below 0.05% of their total N-input based on proximate analysis and FARM analyses, while Scotland removed between 0.08% and 0.19% for proximate analysis and FARM model, respectively.

The assessment of N removal costs and the potential value of bivalve bioextraction is available in Table 5. Considering the cost of traditional water treatment from the Dorset-based facility (£58 thousand t N⁻¹ y⁻¹), the cumulative avoided cost value of removal by harvested shellfish on a national scale was estimated to be over £7 million (or £16 million; FARM) per year for the 126.6 tonnes (ca. 285.8 tonnes; FARM) of N removed in 2019. The value of N removal by bivalve bioextraction can also be compared to the costs associated with alternative N reduction strategies.

Catchment management efforts that support local farmers, landowners, and businesses in reducing N influx were estimated to provide a comparable N reduction benefit worth approximately £1.1 million (£2.5 FARM). Using the average cost of reducing a tonne of N of £295,000 from the Solent case study, the value of bioextraction was calculated at over £37 million (£84 million; FARM). When applying the highest avoided cost estimate of £1,100,000 per tonne, the total estimated value increases to £139 million (£398 million; FARM).

Discussion

Nutrient Loadings in UK Waters

This analysis relied on available N loading data, though limitations in reporting and data consistency influenced the selection of the dataset as detailed above. Despite these constraints, marked variations in the loadings can be observed. The contribution of nutrient influx varies across the country, with England and Scotland leading in total nitrogen loadings. However, human population size and density, agricultural intensity, land use, and N management policies also contribute to these regional differences. When considering loadings per unit area (Table 4), England and Wales exhibited the highest N loadings, both of which have higher population density than Scotland. Wales has also the highest percentage of land used for agricultural purposes (88%) compared to England (69%), Scotland (68%), and Northern Ireland (73%) (Armstrong, 2016).

Although more precise data are currently not available, there have been reports of improvements in water quality and reduced N loadings, which has been attributed to the measures taken by the UK government to reduce nutrient

loadings into surrounding waters (DEFRA, 2012; DEFRA, 2019; Maier et al., 2009; Painting et al., 2018). Analysis conducted by OSPAR also revealed a significant reduction in nutrient inputs to the OSPAR Maritime Areas, particularly in the Greater North Sea area (Axe et al., 2022). However, inputs of nitrogen oscillate year to year, and a lag caused by nitrogen reservoirs in soils and sediments as well as confounding effects of environmental change, may result in future shifts, though the direction and magnitude of these changes remain uncertain (Basu et al., 2022; Causse et al., 2015; Foden et al., 2011). Considering further land developments, offshore aquaculture farms and environmental changes, implementing more bioremediation methods such as bivalve bioextraction would be beneficial.

Shellfish Production

Traditionally, the UK's for-consumption production focuses on blue mussels and Pacific oysters, reaching nearly 16 thousand tonnes in 2019. In comparison, native oysters, Manila clams and scallops contributed less than 312 tonnes combined between 2013 and 2019, peaking at 69 tonnes in 2013 (CEFAS, 2023, pers. comm.). The combined worth of all harvested bivalve species was placed at £22 million in 2019 (CEFAS, 2023, pers. comm.), with mussels, oysters and Manila clams responsible for over 99% of this value. Over the past decade, the UK's shellfish industry has consistently placed in the top 20 oyster and mussel producers in the world and the top 10 in Europe between 2015 and 2021 (FAO, 2023; Seafish, 2023b, 2023a; Wijsman et al., 2019). This was achieved despite the overall decline in bivalve production, from 24,149 tonnes in 2015 to 15,936 tonnes in 2019, which was largely due to decreased yields of mussels (Figs. 1 and 2). Between 2013 and 2019, blue mussels experienced a 41% drop in yields, partially attributed to biological events such as spread of diseases, Harmful Algal Blooms (HABs), insufficient spat supply and retention, predation and enhanced biosecurity protocols which have contributed to the closing of mussel ventures in locations across Scotland (Crown Estate Scotland & Maritek, 2019). This was also observed in Wales and Northern Ireland, which experienced the most visible decline in total shellfish production, dropping from 8,344 tonnes to 2,946 tonnes and from 3,464 tonnes to 1,747 tonnes respectively (2013–2019), again mostly due to reduced mussel yields. The decline in production can be in part due to administration and technical issues such as hindered access to finance for smaller companies due to a long wait (often greater than three years) before a company starts gaining revenues, exacerbated by unexpected costs, slow domestic market, difficulty in finding employees (particularly in the remote regions of Highlands), as well as regulatory constraints.

This is in stark contrast to continuous growth of the Pacific oyster sector, leading to a 111% increase in production, mostly due to higher yields in England and Northern Ireland. Harvest levels in Scotland and Wales have remained limited, possibly due to unsuitable environmental conditions, ecological concerns about their invasiveness, and strict regulatory barriers on non-native species.

The expansion of the Pacific oyster production in England and Northern Ireland can be attributed to the improved availability of hatchery-reared seed and the high value of the species, which ranks second only to Manila clams and native oysters (CEFAS, 2023, pers. comm.). However, despite its high value and unlike Pacific oysters, native oyster production has seen a 62% decline since 2013. Recently, though, there has been a resurgence in native oyster harvests, driven by the growing recognition of their biodiversity and ecosystem services (Bertolini & Pastres, 2022). This recovery can be attributed to the widespread conservation and restoration efforts, aimed at expanding natural beds and supporting the success of commercial production. Initiatives such as the protection of Loch Ryan, Scotland's only commercially exploited wild bed, and the Fal Fishery in Cornwall, where traditional, low-impact methods support native oyster recovery, contribute to both ecological restoration and commercial viability (Baggett et al., 2014; FalFishery, 2025; Kaspar, 2014; NatureScot, 2023; Preston et al., 2020; Zu Ermgassen et al., 2020).

Nutrient Removal Estimates for the UK

These results highlight how bivalve bioremediation, when integrated thoughtfully into catchment management systems, may be able to contribute to water quality improvements where it matters most. Despite recent declines in bivalve production in the UK, by 2019 shellfish aquaculture still contributed to nutrient removal, with mussels accounting for the greatest proportion (Figs. 3 and 4; Table 3). The removal services of fixed N and C provided by bivalves are closely linked to production levels; the greater the production volumes, the greater the nutrient removal service. However, specifically for C removal, it is also important to note that the estimated 1762 tonnes potentially removed does not represent long-term overall sequestration, but rather the amount of particulate C from the water column fixed into bivalve body (tissue and shell) that is then harvested. The contribution of bivalves to carbon sequestration is highly debated in literature, along with how to include the biological processes in the CO₂ budget associated with aquaculture of bivalves (Álvarez-Salgado et al., 2022; Chen et al., 2025; Filgueira et al., 2015, 2019; Jansen & Van Den Bogaart, 2020). Moreover, while calcification

removes inorganic carbon, it is not CO₂ from the water column (Ware et al., 1992). As such shell CaCO₃ does not reduce the carbon footprint and it does not contribute to reduced atmospheric CO₂ and climate change mitigation (Cunningham & Hunt, 2023; Ware et al., 1992). While this falls outside of the scope of this work, crucial future work to investigate not only the CO₂ fluxes in shellfish ecosystems and how these relate to C balance, but also the difference in the CO₂ budget between the tissue and shell is suggested (Chen et al., 2025; Filgueira et al., 2019; Pernet et al., 2025b).

Similar nutrient removal results have been seen for the UK based on mussel production in the UK (2018), where 14,247 tonnes of mussels removed an estimated 125 tonnes of N (Cubillo et al., 2023; Ferreira et al., 2020). The removal efficiency is 0.0088 tonnes of N per tonne of mussel is slightly higher than the removal proposed by Olivier et al., (2020), but still shows the greatest bioremediation potential per tonne of shellfish. Comparably, the average oyster removal here was 0.0037 tonnes of N per tonne of oysters, higher than the estimated 0.00233 based on global data (Olivier et al., 2020). Variability in nutrient removal estimates is in part due to site- and species-specificity and the true potential of any farm can only be ascertained with the inclusion of the particular circumstances of the planned farm (Rose et al., 2015). Nutrient removal further varies between culture systems, with rope-cultured blue mussels capturing more N than bottom-cultured ones (Olivier et al., 2021). Bivalves are known to allocate different proportions of energy to shell versus soft tissues depending on the farming system. Intertidal and bottom-cultured mussels invest a greater share of energy in shell formation compared to those grown in suspended systems, with implications for the carbon cycle (Filgueira et al., 2015). While this was outside of the scope of this work, it is an important area of investigation for future research.

A similar pattern was observed in results from the FARM model, where mussels were the primary contributors to nutrient removal. The FARM model results estimated an average N removal of 285.8 tonnes, over two times higher than proximate analysis results. This discrepancy arises from methodological differences: while proximate analysis quantifies N content solely in harvested individuals, the FARM model mass balance incorporates the filtering capacity of the entire shellfish population, including undersized and unharvested individuals, as well as those lost to natural mortality. This broader scope explains the higher N removal estimates provided by the model, although direct comparison between these two approaches is challenging. Further disaggregating the contributions of mortality and unharvested biomass within the model could provide additional insights into their respective roles in N removal.

Efficiency on National Scale and Economic Potential

Based on the 126.6 to 285.8 (average) tonnes of N removal estimates using proximate analysis and the FARM model, bivalve aquaculture could potentially remove the equivalent of between 0.03% and 0.08% of the N loadings in the UK waters, respectively. While this may seem modest, even small reductions can have a pronounced impact in areas prone to eutrophication. Under the Water Framework Directive (WFD), for example, water bodies are classified into five status bands, and even small improvements can be enough to shift a water body into a higher (better water quality) category, which can ease regulatory pressures and support better environmental outcomes. The removal rates of N by bivalves also compare favourably to other management strategies such as agricultural best management practices and stormwater control measures (Rose et al., 2015).

Removal results vary considerably among the countries included in this analysis, as well as among different years. Comparing individual removal of local N loads, Wales (0.12%–0.27% of N loading into local waters) and Northern Ireland (0.11%–0.24%) have shown the greatest potential. In contrast, results showed potential removal in England of <0.05% of their total N-input, based on proximate analysis and FARM analyses, while removal estimates for Scotland were between 0.08% and 0.19% for proximate analysis and FARM model, respectively. While these two countries have greater production than Wales and Northern Ireland, their N loadings are significantly higher. However, other than in some specific areas, bivalve-mediated bioextraction is not expected to fully offset nutrient loadings but rather support traditional land-based strategies as part of a comprehensive nutrient management plan. Inclusion of bivalve bioextraction in the soil to sea catchment management process, working with other habitat restoration processes, could help expedite fulfilment of nutrient management goals. For example, in England and Scotland, the harvest-related bivalve N bioextraction represents between 2.5% and 6.5% of their respective N loads from industry effluents, while in Northern Ireland it was greater than the N loadings from industry effluents. Although the offsetting of industrial effluents is not the goal of bivalve aquaculture, this comparison highlights the tangible benefits for water quality improvement, in addition to the wide range of other ecosystem services of bivalves and gains from the food and by-products of the industry (Bateman & Bishop, 2017; Gallardi, 2014; Lillebø et al., 2017; Olivier et al., 2020; Pinn, 2021; Smaal et al., 2019). The estimated N removal by bivalves of between 126.6 and 285.8 tonnes can be interpreted as an ecosystem service equivalent to water treatment for 38–86.5 thousand people, considering a per-person annual load of 3.3 kg N per year (Ferreira et al., 2007). Moreover, it is a cost-efficient

mitigation measure compared to most of the other mitigation or purification measures and other best management practices (Petersen et al., 2012, 2014; Pretty et al., 2003; Rose et al., 2015; Watson et al., 2020a, b, c).

However, it should be noted that N removal from 2019 was compared to N loadings from 2014. While this comparison allows for the use of the most recent complete data, it also introduces potential inaccuracy to the mass balance as the changes in loadings and shellfish production can be significant year-to-year. In this case, the production in 2014 was over 5,000 tonnes higher, and the estimated N removal ranged from 177.7 to 402.3 tonnes (PA and FARM, respectively). This would mean a removal of between 0.05% and 0.11% of the total nitrogen loadings in the UK, compared to the 0.03% to 0.08% estimated for 2019. While the removal overall is greater, in relation to significant loadings in the UK waters the difference is not substantial, and the 2014–2019 comparison still illustrates the benefit of the bivalve shellfish bioextraction.

To further quantify the benefits of bivalve bioextraction, an alternative cost method was employed. While the valuation of any ecosystem service is a challenging task with numerous caveats from economic and biological perspectives, this approach has been successfully applied in coastal regions of the United States for oysters (Bricker et al., 2020; Dvarskas et al., 2020; Parker & Bricker, 2020) and in the Baltic Sea for mussels (Gren, 2019). In the UK, such valuation has been conducted for oyster populations of the Solent (southern England) (Watson et al., 2020a, b, c) and shellfish populations in Dundrum Bay (Northern Ireland) (Cubillo et al., 2023; Ferreira et al., 2020). Here, the cost was estimated based on the assumption that the bivalves would eventually be harvested, taking into account the costs associated with nutrient removal at water treatment facilities, catchment management, changes in agricultural land use, support for farmers and agricultural programs, as well as upgrades to existing wastewater treatment plants and their associated drainage infrastructure (Table 5). This method also assumes that the nutrient removal services provided by shellfish are equivalent to those of other strategies and that there is a willingness to pay for these services due to nutrient-related water quality goals (Freeman et al., 2014).

This study estimated that integrating cultivated bivalves into nutrient management programs could provide benefits valued between £32 thousand to £314 million annually, depending on the assessment method and N removal approach. While the range is substantial, it highlights the great potential of bivalve bioextraction and its role supporting other water management strategies.

On the conservative end, using the proximate analysis, it would cost £7 million to remove the estimated 126 tonnes of N (based on PA) through methanol dosing and tertiary denitrifying sand filters (Wessex Water, 2023). With the higher FARM-based removal estimates, the cumulative savings on a national scale could surpass £16 million annually for the average ca. 285.8 tonnes of estimated N removed in 2019. Outside of the traditional wastewater treatment methods, bivalve bioextraction could be worth between £1.1 million to £2.5 million when compared to collaborative efforts with local landowners and businesses to reduce N influx. A different regional valuation based on Solent abatement costs estimates the economic benefits of N removal through oyster filtration. Using an average reduction cost of £295,000 per tonne calculated by Watson et al., (2020a, b, c), the estimated value of N removal in this study exceeds £37 million (£84 million FARM). If their highest potential avoided cost of £1,100,000 per tonne is considered, this value could rise to £139 million (£314 million; FARM). Moreover, there is also a consideration for the damage costs associated with nutrient enrichment and eutrophication, placed at £82.5 million to £125.8 million annually for England and Wales (Pretty et al., 2003).

While it is crucial to remember that any potential remuneration for the services provided by bivalves would be most likely based on the least cost approach and adjusted for the needs of the market, it can nonetheless become an innovative revenue stream for shellfish farmers in the UK and lead to a nutrient credit trading (NCT) policy framework. The use of NCT has been considered for the USA and has already been implemented in some states (Jones et al., 2010; Newell & Mann 2012; Rose et al., 2015; Stephenson et al., 2010; Wheeler, 2020). The monetary value of bivalve aquaculture N removal was placed between £82,522 (current) and £318,299 (prospective) for oysters in the Great Bay Piscataqua (Bricker et al., 2020), with annual revenue between £440 to £9.82 million in Chesapeake Bay (Parker & Bricker, 2020) and £129 to £325 per kilogram of N annually at Greenwich Bay (USA) (Dvarskas et al., 2020). Evaluations from the EU also highlight the great potential of NCT, with mussel farming in Limfjorden (Denmark) providing avoided costs in the range of £1.34 million to £1.62 million (Filippelli et al., 2020). However, it is important to note that these valuations do not represent compensations for the growers. Assessments for the USA oyster aquaculture highlight that the prices of credit in the NCT are key in determining the economic incentives, and high credit prices are required to result in significant production increases (Weber et al., 2018). Notwithstanding, recognition of the bivalve ecosystem services can serve as a valuable incentive,

help open doors to additional support mechanisms, such as certification schemes, policy incentives, and financing opportunities which can enable delivery of positive impacts at a larger scale (Alleway et al., 2019).

Due to the intricate dynamics of aquatic ecosystems and biochemical processes involved in N cycles, a national NCT in the UK would most likely be based on harvested biomass, such as the Chesapeake example (Cornwell et al., 2016; Ferreira & Bricker, 2019; Rose et al., 2021). However, the inclusion of processes such as denitrification and burial could be implemented on a regional, catchment basis using modelling approaches outlined in the GAIN framework (Cubillo et al., 2023; Ferreira et al., 2020; Ferreira & Bricker, 2019) and help account for a higher proportion of nutrient loadings. Studies conducted in the Chesapeake Bay revealed that bivalve-mediated denitrification can remove significantly more N than bivalve harvest, and a best management practice for oyster denitrification was approved in 2023 (Cornwell et al., 2023; Golen, 2007; Mykoniatis & Ready, 2020). The stimulation of denitrification has been also demonstrated in a recent meta-analysis of oyster habitats in comparison to bare sediments (Ray & Fulweiler, 2021). Oyster reefs consistently showed denitrification efficiency above 50%, typically exceeding 75%, indicating net N removal with minimal N₂O emissions (Onorevole et al., 2018). High denitrification rates have also been reported within and between natural green-lipped mussel (*Perna canaliculus*) beds (Sea et al., 2021), suggesting a wider spatial impact of bivalve presence. Moreover, when comparing natural reefs and beds to cultivated ones, it is important to take into account the higher tissue: shell ratio of farmed bivalves which translates to higher N removal (Ayvazian et al., 2022).

While bivalve-mediated nutrient management control cannot fully address excessive nutrient influx into coastal waters, it can help optimise nutrient management strategies and provide recognition and monetary incentives to shellfish farmers (Olivier et al., 2020; Rose et al., 2015). However, there are numerous considerations, such as the partial offset of the nutrient removal associated with high stocking densities or excessive biodeposition, which can increase localised nutrient regeneration, resuspension, or sediment oxygen demand (Alleway et al., 2025; Burkholder & Shumway, 2011). As net nutrient extraction efficiency is reduced, more biomass or area would be required to achieve the same removal targets, increasing operational costs while reducing the marketable value of the ecosystem service (e.g., nutrient credit trading, improved water quality, co-benefits for aquaculture). Therefore, optimizing stocking densities and spatial planning of aquaculture is essential to not only

control environmental impact on the seabed, but also maintain positive net removal and the economic potential of bioextraction. Moreover, inclusion of bivalves in nutrient management will require a comprehensive regulatory framework, encompassing credit calculations, monitoring protocols, and enforcement mechanisms, which demands the collaboration of governmental bodies, aquaculture operators and environmental advocates. These efforts need to be supported by fostering public awareness and acceptance to maintain interest in the food provisioning services of bivalves. It is crucial to ensure that consumers understand that bivalve bioremediation stands as a safe and well-regulated process, that does not compromise food safety. While bivalves play a key role in improving water quality by filtering excess nutrients, this process does not introduce new health concerns—rather, it highlights their environmental value alongside their established role as a safe and nutritious food source. Strict health and safety requirements already govern bivalve production, ensuring that harvested shellfish meet rigorous standards for human consumption and does not in itself introduce health concerns. As NCT frameworks in the UK broaden their scope to encompass bivalve producers, they would promote to the eco-intensification of existing sites, enhanced yields, improved profitability, and the generation of employment opportunities across the country. This inclusion would also imply affiliated benefits for the ecosystem, extending across multiple industries and cultural services.

Conclusions

As nutrient enrichment and elemental imbalance promote eutrophication, bivalve bioextraction offers a promising nature-based solution to help resolve the issue. The UK, with its extensive coastline and significant bivalve aquaculture potential, is a prime location for implementing bivalves into water management frameworks. The present study provides encouraging results for the bioextraction potential of UK bivalves. The estimates provided show a wide range, showing strong country- and species-specific differences affecting the amount of nutrients removed. These results are consistent with reviewed literature highlighting the positive effects of bivalves on water quality and net nutrient removal. Moreover, bivalve-mediated nutrient removal costs compared favourably with best management practices and stormwater treatment. Nitrogen removal cost assessments reveal substantial savings potential, ranging from £33 thousand to £314 million associated with the total harvest of all bivalve shellfish species. As this study focused on harvest-based removal, further

research will be required on the impact of biochemical processes, denitrification, nitrification, biodeposition and excretion which contribute to the net balance of nutrients. Research suggests that bivalves' presence stimulates denitrification, although high densities and biodeposition may partially offset nutrient removal.

As one of the top 20 global bivalve aquaculture producers, the UK stands to benefit from further research into the bioextraction potential of bivalves and the development of effective management policies. Successful implementation hinges on robust regulations, stakeholder collaboration, and public awareness, fostering economic benefits and sustainable practices. There is potential in using harvest removal estimates, combined with widely tested methods like the FARM model for a more comprehensive understanding of site-specific nutrient removal capacity. Future work should focus on catchment-level case studies and adapting the nutrient management strategies to the local context, ensuring that the potential of the UK's shellfish sector is fully realised. Including bivalve bioremediation ecosystem services in the NTC could not only achieve substantial cost savings but also address critical environmental concerns, propelling the UK towards a more sustainable and economically efficient future.

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s12237-026-01689-3>.

Acknowledgements This article was written under the Ecosystem Services (ES) project commissioned by the Shellfish Stakeholder Working Group. This work was funded by Fishmongers' Company and the Sea Fish Industry Authority (Seafish). We would also like to express gratitude to Tim Ellis (Cefas) for the UK shellfish production data and to all the organisations that helped us gain UK data: Defra, DAERA, Cefas, Environmental Agency, Natural Resources Wales, SEPA and OSPAR. This work was informed by Green Aquaculture Intensification in Europe (GAIN) Project [www.gain2020.com], which served as a key reference for both data and methodology.

Author Contributions Konstancja Woźniacka: Writing – original draft, Writing – review & editing, Visualization, Investigation, Formal analysis, Data curation. Eunice Pinn: Supervision, Project administration, Writing – review & editing, Methodology, Funding acquisition, Conceptualisation. Heather Moore: Supervision, Writing – review & editing, Methodology, Funding acquisition, Conceptualisation. Suzanne B. Bricker: Supervision, Writing – review & editing, Methodology, Conceptualisation. Matthew Service: Supervision, Writing – review & editing, Funding acquisition, Conceptualisation. Alhambra Martínez Cubillo: Writing – review & editing, Software, Methodology. João G. Ferreira: Writing – review & editing, Software, Methodology.

Data Availability All data utilised in this project was publicly available and/or sourced from government bodies. The data that support this study can be made available from the corresponding author upon reasonable request.

Declarations

All authors have approved of and have agreed to submit the manuscript to the Estuaries and Coasts journal.

Ethics Approval and Consent to Participate Not applicable.

Competing interests The authors declare that they have no known competing interests.

Open Access This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if changes were made. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit <http://creativecommons.org/licenses/by/4.0/>.

References

- AFBI (2023). *Agri-Food and Biosciences Institute*. <https://www.afbi.ni.gov.uk/>
- Alleyway, H. K., Gillies, C. L., Bishop, M. J., Gentry, R. R., Theuerkauf, S. J., & Jones, R. (2019). The ecosystem services of marine aquaculture: Valuing benefits to people and nature. *BioScience*, 69(1), 59–68. <https://doi.org/10.1093/biosci/biy137>
- Alleyway, H. K., Gentry, R. R., Smart, L., Jones, A. R., & Mackay, J. (2025). Managing bivalve aquaculture to enhance blue carbon ecosystems and carbon sequestration. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 35(7), Article e70160. <https://doi.org/10.1002/aqc.70160>
- Álvarez-Salgado, X. A., Fernández-Reiriz, M. J., Fuentes-Santos, I., Antelo, L. T., Alonso, A. A., & Labarta, U. (2022). CO2 budget of cultured mussels metabolism in the highly productive Northwest Iberian upwelling system. *Science of the Total Environment*, 849, Article 157867. <https://doi.org/10.1016/j.scitotenv.2022.157867>
- Armstrong, E. (2016). The Farming Sector in Wales. *Research Briefing, National Assembly*. <https://research.senedd.wales/media/aixb5t4/16-053-web-english2.pdf>
- Axe, P., Sonesten, L., Skarbövik, E., Leujak, W., & Nielsen, L. (2022). *Inputs of Nutrients to the OSPAR Maritime Area. In OSPAR, 2023: The 2023 Quality Status Report for the North-East Atlantic*. OSPAR Commission. <https://oap.ospar.org/en/ospar-assessment/s/quality-status-reports/qsr-2023/indicator-assessments/inputs-nutrients/>
- Ayvazian, S.G., Ray, N.E., Gerber-Williams, A., Grabbert, S., Pimenta, A., Hancock, B., Cobb, D., Strobel, C., Fulweiler, R.W. (2022). Evaluating connections between nitrogen cycling and the macrofauna in native oyster beds in a New England estuary. *Estuaries and Coasts*, 45, 196–212.
- Baggett, L. P., Powers, S. P., Brumbaugh, R., Coen, L. D., DeAngelis, B., Greene, J., Hancock, B., & Morlock, S. (2014). Oyster habitat restoration monitoring and assessment handbook The Nature Conservancy: Arlington. *VA, USA*.
- Barrett, L.T., Theuerkauf, S.J., Rose, J.M., Alleyway, H.K., Bricker, S.B., Parker, M., Petrolia, D.R., Jones, R.C. (2022). Sustainable growth of non-fed aquaculture can generate valuable ecosystem

- benefits. *Ecosystem Services*, 53, 101396. <https://doi.org/10.1016/j.ecoser.2021.101396>
- Basu, N. B., Van Meter, K. J., Byrnes, D. K., Van Cappellen, P., Brouwer, R., Jacobsen, B. H., Jarsjö, J., Rudolph, D. L., Cunha, M. C., Nelson, N., Bhattacharya, R., Destouni, G., & Olsen, S. B. (2022). Managing nitrogen legacies to accelerate water quality improvement. *Nature Geoscience*, 15(2), 97–105. <https://doi.org/10.1038/s41561-021-00889-9>
- Bateman, D., & Bishop, M. (2017). The environmental context and traits of habitat-forming bivalves influence the magnitude of their ecosystem engineering. *Marine Ecology Progress Series*, 563, 95–110. <https://doi.org/10.3354/meps11959>
- Bertolini, C., & Pastres, R. (2022). Identifying knowledge gaps for successful restorative aquaculture of *Ostrea edulis*: A bibliometric analysis. *Open Research Europe*, 1, Article 103. <https://doi.org/10.12688/openreseurope.14074.3>
- Boesch, D. F. (2019). Barriers and bridges in abating coastal eutrophication. *Frontiers in Marine Science*, 6, Article 123. <https://doi.org/10.3389/fmars.2019.00123>
- Boesch, D., & Brinsfield, R. (2000). Coastal Eutrophication and Agriculture: Contributions and Solutions. In E. Balázs, E. Galante, J. M. Lynch, J. S. Schepers, J. P. Toutant, D. Werner, & P. A. T. J. Werry (Eds.), *Biological Resource Management Connecting Science and Policy* (pp. 93–115). Springer. https://doi.org/10.1007/978-3-662-04033-1_8
- Bresnan, E. (2016). Scottish coastal observatory data. *Marine Scotland Science*. <https://doi.org/10.7489/1761-1>
- Bricker, S. B., Ferreira, J. G., Zhu, C., Rose, J. M., Galimany, E., Wikfors, G., Saurel, C., Miller, R. L., Wands, J., Trowbridge, P., Grizzle, R., Wellman, K., Rheault, R., Steinberg, J., Jacob, A., Davenport, E. D., Ayvazian, S., Chintala, M., & Tedesco, M. A. (2018). The role of shellfish aquaculture in reduction of eutrophication in an urban estuary. *Environmental Science & Technology*, 52(1), 173–183. <https://doi.org/10.1021/acs.est.7b03970>
- Bricker, S. B., Grizzle, R. E., Trowbridge, P., Rose, J. M., Ferreira, J. G., Wellman, K., Zhu, C., Galimany, E., Wikfors, G. H., Saurel, C., Landeck Miller, R., Wands, J., Rheault, R., Steinberg, J., Jacob, A. P., Davenport, E. D., Ayvazian, S., Chintala, M., & Tedesco, M. A. (2020). Bioextractive removal of nitrogen by oysters in Great Bay Piscataqua River Estuary, New Hampshire, USA. *Estuaries and Coasts*, 43(1), 23–38. <https://doi.org/10.1007/s12237-019-00661-8>
- Brink, C., Van Grinsven, H., Jacobsen, B. H., Rabl, A., Gren, I. M., Holland, M., Klimont, Z., Hicks, K., Brouwer, R., Dickens, R., Willems, J., Termansen, M., Velthof, G., Alkemade, R., Van Oorschot, M., & Webb, J. (2011). Costs and benefits of nitrogen in the environment. In M. A. Sutton, C. M. Howard, J. W. Erisman, G. Billen, A. Bleeker, P. Grennfelt, Van H. Grinsven, & B. Grizzetti (Eds.), *The European Nitrogen Assessment* (1st ed., pp. 513–540). Cambridge University Press. <https://doi.org/10.1017/CBO9780511976988.025>
- Burkholder, J. M., & Shumway, S. E. (2011). Bivalve Shellfish Aquaculture and Eutrophication. *Shellfish Aquaculture and the Environment* (pp. 155–215). John Wiley & Sons, Ltd. <https://doi.org/10.1002/9780470960967.ch7>
- Carmichael, R. H., Walton, W., & Clark, H. (2012). Bivalve-enhanced nitrogen removal from coastal estuaries. *Canadian Journal of Fisheries and Aquatic Sciences*, 69(7), 1131–1149. <https://doi.org/10.1139/f2012-057>
- Causse, J., Baurès, E., Mery, Y., Jung, A.-V., & Thomas, O. (2015). Variability of N export in water: A review. *Critical Reviews in Environmental Science and Technology*, 45(20), 2245–2281. <https://doi.org/10.1080/10643389.2015.1010432>
- Chen, X., Zhang, Z., Pan, M., Liu, Y., Li, C., Zhou, Y., Li, L., Dong, X., Dong, Y., Li, F., Liu, S., Wang, X., Tian, S., Liu, Y., Zhang, J., Qiu, Y., Wang, X., Cai, W., Tian, X., ... Dong, S. (2025). Oyster farming acts as a marine carbon dioxide removal (mCDR) hotspot for climate change mitigation. *Proceedings of the National Academy of Sciences*, 122(36), Article e2504004122. <https://doi.org/10.1073/pnas.2504004122>
- Clements, J. C., & Comeau, L. A. (2019). Nitrogen removal potential of shellfish aquaculture harvests in eastern Canada: A comparison of culture methods. *Aquaculture Reports*, 13, Article 100183. <https://doi.org/10.1016/j.aqrep.2019.100183>
- CMEMS (2023). *Copernicus Marine Data Store*. <https://data.marine.copernicus.eu/products>
- Cornwell, J., Rose, J., Kellogg, L., Luckenbach, M., Bricker, S., Paynter, K., Moore, C., Parker, M., Sanford, L., Wolinski, B., Lacatell, A., Fegley, L., & Hudson, K. (2016). *Panel Recommendations on the Oyster BMP Nutrient and Suspended Sediment Reduction Effectiveness Determination Decision Framework and Nitrogen and Phosphorus Assimilation in Oyster Tissue Reduction Effectiveness for Oyster Aquaculture Practices*. 197.
- Cornwell, J. C., Bricker, S. B., Lacatell, A., Luckenbach, M. W., Marenghi, F., Moore, C., Parker, M., Paynter, K., Rose, J., Sanford, L., Wolinski, W., Caretti, O. N., Reichert-Nguyen, J., & Slacum, H. W. (2023). *Nitrogen and phosphorus reduction associated with harvest of hatchery-produced oysters and reef restoration: Assimilation and enhanced denitrification: Panel recommendations*. Chesapeake Bay Program Partnership. <https://www.chesapeakebay.net/what/publications/oyster-bmp-expert-panel-second-incremental-report-assimilation-and-enhanced-denitrification> Water Quality Goal Implementation Team.
- Crown Estate Scotland, & Maritek (2019). *Shellfish Critical Mass Development Plan Pilot—Clyde; WP 3: Development Planning Process Template*.
- Cubillo, A. M., Lopes, A. S., Ferreira, J. G., Moore, H., Service, M., & Bricker, S. B. (2023). Quantification and valuation of the potential of shellfish ecosystem services in mitigating coastal eutrophication. *Estuarine, Coastal and Shelf Science*, 293, Article 108469. <https://doi.org/10.1016/j.ecss.2023.108469>
- Cunningham, C., & Hunt, C. (2023). *Scottish Blue Carbon—A literature review of the current evidence for Scotland's blue carbon habitats*. (No. 1326; NatureScot Research Report). NatureScot. <https://www.nature.scot/doc/naturescot-research-report-1326-scottish-blue-carbon-literature-review-current-evidence-scotlands>
- DAERA, D, of A. Environment and Rural Affairs. (2023). *DAERA Annual Chlorophyll a Survey (2008–2017)*. <https://www.data.gov.uk/dataset/693a1987-df2b-4532-84c5-47df63498714/daera-annual-chlorophyll-a-survey-2008-2017>
- Dai, M., Zhao, Y., Chai, F., Chen, M., Chen, N., Chen, Y., Cheng, D., Gan, J., Guan, D., Hong, Y., Huang, J., Lee, Y., Leung, K. M. Y., Lim, P. E., Lin, S., Lin, X., Liu, X., Liu, Z., Luo, Y.-W., ... Zhang, Z. (2023). Persistent eutrophication and hypoxia in the coastal ocean. *Cambridge Prisms: Coastal Futures*, 1, Article e19. <https://doi.org/10.1017/cft.2023.7>
- DEFRA (2012). *Marine Strategy Part One: UK Initial Assessment and Good Environmental Status*.
- Defra (2019). *Marine strategy part one: UK updated assessment and Good Environmental Status*. Department for Environment, Food & Rural Affairs. <https://www.gov.uk/government/publications/marine-strategy-part-one-uk-updated-assessment-and-good-environmental-status>
- Dvorskas, A., Bricker, S. B., Wikfors, G. H., Bohorquez, J. J., Dixon, M. S., & Rose, J. M. (2020). Quantification and valuation of nitrogen removal services provided by commercial shellfish aquaculture at the subwatershed scale. *Environmental Science & Technology*, 54(24), 16156–16165. <https://doi.org/10.1021/acs.est.0c03066>
- Environment Agency (2021). *Nitrates: Challenges for the water environment*. Environment Agency. <https://www.gov.uk/government/publications/nitrates-challenges-for-the-water-environment>

- Fal Fishery Cooperative CIC. FalFishery, & FalFishery (2025). <https://falfisherycooperativecic.org/>
- FAO (2023). *Global Aquaculture Production. Fisheries and Aquaculture Division [online]* [Dataset]. <https://www.fao.org/fishery/en/collection/aquaculture?lang=en>
- Ferreira, J. G. (2007). *SMILE: Sustainable mariculture in Northern Irish lough ecosystems*. SMILE.
- Ferreira, J. G., & Bricker, S. B. (2016). Goods and services of extensive aquaculture: Shellfish culture and nutrient trading. *Aquaculture International*, 24(3), 803–825. <https://doi.org/10.1007/s10499-015-9949-9>
- Ferreira, J. G., & Bricker, S. B. (2019). Assessment of Nutrient Trading Services from Bivalve Farming. In A. C. Smaal, J. G. Ferreira, J. Grant, J. K. Petersen, & Ø. Strand (Eds.), *Goods and Services of Marine Bivalves* (pp. 551–584). Springer International Publishing. https://doi.org/10.1007/978-3-319-96776-9_27
- Ferreira, J. G., Hawkins, A. J. S., & Bricker, S. B. (2007). Management of productivity, environmental effects and profitability of shellfish aquaculture—The farm aquaculture resource management (FARM) model. *Aquaculture*, 264(1), 160–174. <https://doi.org/10.1016/j.aquaculture.2006.12.017>
- Ferreira, J. G., Sequeira, A., Hawkins, A. J. S., Newton, A., Nickell, T. D., Pastres, R., Forte, J., Boday, A., & Bricker, S. B. (2009). Analysis of coastal and offshore aquaculture: Application of the FARM model to multiple systems and shellfish species. *Aquaculture*, 292(1), 129–138. <https://doi.org/10.1016/j.aquaculture.2009.03.039>
- Ferreira, J. G., Cubillo, A. M., Lopes, A. S., Marteleira, R., Service, M., Moore, H., Cromie, H., & Bricker, S. B. (2020). *GAIN D2.9 – Report & white paper on framework for a nutrient credit trading policy for Europe, integrating shellfish producers. Deliverable 2.9. GAIN - Green Aquaculture INTensification in Europe. EU Horizon 2020 project grant no. 773330*. (p. 58).
- Ferreira, J. G., Taylor, N. G. H., Cubillo, A., Lencart-Silva, J., Pastres, R., Bergh, O., & Guilder, J. (2021). An integrated model for aquaculture production, pathogen interaction, and environmental effects. *Aquaculture*. <https://doi.org/10.1016/j.aquaculture.2021.736438>
- Filgueira, R., Byron, C. J., Comeau, L. A., Costa-Pierce, B., Cranford, P. J., Ferreira, J. G., Grant, J., Guyonnet, T., Jansen, H. M., Landry, T., McKindsey, C. W., Petersen, J. K., Reid, G. K., Robinson, S. M. C., Smaal, A., Sonier, R., Strand, Ø., & Strohmeier, T. (2015). An integrated ecosystem approach for assessing the potential role of cultivated bivalve shells as part of the carbon trading system. *Marine Ecology Progress Series*, 518, 281–287. <https://doi.org/10.3354/meps11048>
- Filgueira, R., Strohmeier, T., & Strand, Ø. (2019). Regulating Services of Bivalve Molluscs in the Context of the Carbon Cycle and Implications for Ecosystem Valuation. In A. C. Smaal, J. G. Ferreira, J. Grant, J. K. Petersen, & Ø. Strand (Eds.), *Goods and Services of Marine Bivalves* (pp. 231–251). Springer International Publishing. https://doi.org/10.1007/978-3-319-96776-9_12
- Filippelli, R., Termansen, M., Hasler, B., Timmermann, K., & Petersen, J. K. (2020). Cost-effectiveness of mussel farming as a water quality improvement measure: Agricultural, environmental and market drivers. *Water Resources and Economics*. <https://doi.org/10.1016/j.wre.2020.100168>
- Foden, J., Devlin, M. J., Mills, D. K., & Malcolm, S. J. (2011). Searching for undesirable disturbance: An application of the OSPAR eutrophication assessment method to marine waters of England and Wales. *Biogeochemistry*, 106(2), 157–175. <https://doi.org/10.1007/s10533-010-9475-9>
- Freeman, A. M., Herriges, J. A., & Kling, C. L. (2014). *The measurement of environmental and resource values: Theory and methods*. Routledge. <https://www.taylorfrancis.com/books/mono/10.4324/9781315780917/measurement-environmental-resource-values-merrick-freeman-iii-joseph-herriges-catherine-kling>
- Gallardi, D. (2014). Effects of Bivalve Aquaculture on the Environment and Their Possible Mitigation: A Review. *Fisheries and Aquaculture Journal*, 05(03). <https://doi.org/10.4172/2150-3508.1000105>
- García-García, L. M., Sivyer, D., Devlin, M., Painting, S., Collingridge, K., & Van Der Molen, J. (2019). Optimizing monitoring programs: A case study based on the OSPAR eutrophication assessment for UK waters. *Frontiers in Marine Science*, 5, Article 503. <https://doi.org/10.3389/fmars.2018.00503>
- Golen, R. F. (2007). Incorporating Shellfish Bed Restoration into a Nitrogen TMDL Implementation Plan. *Proceedings of the Water Environment Federation*, 2007(5), 1056–1068. <https://doi.org/10.2175/193864707786619305>
- Gonzalez-Pola, C., Larsen, K. M. H., Fratantoni, P., & Beszczynska-Möller, A. (2023). *ICES Report on Ocean Climate 2021* (p. 20109588 Bytes). ICES Cooperative Research Reports (CRR). <https://doi.org/10.17895/ICES.PUB.24755574>
- Gren, I.-M. (2019). The economic value of mussel farming for uncertain nutrient removal in the Baltic Sea. *PLoS One*. <https://doi.org/10.1371/journal.pone.0218023>
- Jansen, H., & Van Den Bogaart, L. (2020). Blue carbon by marine bivalves: Perspective of Carbon sequestration by cultured and wild bivalve stocks in the Dutch coastal areas. *Wageningen Marine Research*. <https://doi.org/10.18174/537188>
- Jennings, S., Stentiford, G. D., Leocadio, A. M., Jeffery, K. R., Metcalfe, J. D., Katsiadaki, I., Auchterlonie, N. A., Mangi, S. C., Pinnegar, J. K., Ellis, T., Peeler, E. J., Luisetti, T., Baker-Austin, C., Brown, M., Catchpole, T. L., Clyne, F. J., Dye, S. R., Edmonds, N. J., Hyder, K., & Verner-Jeffreys, D. W. (2016). Aquatic food security: Insights into challenges and solutions from an analysis of interactions between fisheries, aquaculture, food safety, human health, fish and human welfare, economy and environment. *Fish and Fisheries*, 17(4), 893–938. <https://doi.org/10.1111/faf.12152>
- Jones, C., Branosky, E., Selman, M., & Perez, M. (2010). *How Nutrient Trading Could Help Restore the Chesapeake Bay*. <https://policycommons.net/artifacts/1360605/how-nutrient-trading-could-help-restore-the-chesapeake-bay/1974296/>
- Kaspar, H. (2014). *A multi-species shellfish hatchery as a key asset for the development of Scotland's shellfish aquaculture industry and the restoration of native oyster reefs*. Retrieved from MASTS: https://www.masts.ac.uk/media/35906/henry-kaspar_#8230
- Kotta, J., Futter, M., Kaasik, A., Liversage, K., Ratsep, M., Barboza, F. R., Bergstrom, L., Bergstrom, P., Bobsien, I., Diaz, E., Herkul, K., Jonsson, P. R., Korpinen, S., Kraufvelin, P., Krost, P., Lindahl, O., Lindegarth, M., Lyngsgaard, M. M., Muhl, M., ... Virtanen, E. (2020). Cleaning up seas using blue growth initiatives: Mussel farming for eutrophication control in the Baltic Sea. *Science of the Total Environment*. <https://doi.org/10.1016/j.scitotenv.2019.136144>
- Lillebø, A. I., Pita, C., Garcia Rodrigues, J., Ramos, S., & Villasante, S. (2017). How can marine ecosystem services support the Blue Growth agenda? *Marine Policy*, 81, 132–142. <https://doi.org/10.1016/j.marpol.2017.03.008>
- Lindahl, O. (2011). Mussel farming as a tool for re-eutrophication of coastal waters: Experienced from Sweden. In S. E. Shumway (Ed.), *Shellfish aquaculture and the environment* (pp. 217–237). Wiley. <http://onlinelibrary.wiley.com/doi/https://doi.org/10.1002/9780470960967.ch8/summary>
- Lindahl, O., & Kollberg, S. (2008). How mussels can improve coastal water quality. *Bioscience Explained*, 5(1), 1–14.
- Lindahl, O., & Kollberg, S. (2009). Can the EU agri-environmental aid program be extended into the coastal zone to combat eutrophication? *Hydrobiologia*, 629(1), 59–64. <https://doi.org/10.1007/s10750-009-9771-3>
- Maier, G., Nimmo-Smith, R. J., Glegg, G. A., Tappin, A. D., & Worsfold, P. J. (2009). Estuarine eutrophication in the UK: Current

- incidence and future trends. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 19(1), 43–56. <https://doi.org/10.1002/aqc.982>
- Morris, D. (2016). *Seawater temperature records for the UK Shelf—All Cefas Seawater Temperature Data 1880–2014* (Version 1) [Dataset]. Cefas Laboratory, UK. <https://doi.org/10.14466/CEFA SDATAHUB.4>
- Mykoniatis, N., & Ready, R. (2020). The potential contribution of oyster management to water quality goals in the Chesapeake Bay. *Water Resources and Economics*. <https://doi.org/10.1016/j.wre.2020.100167>
- NatureScot (2023, June 27). *Native oyster*. NatureScot. <https://www.nature.scot/plants-animals-and-fungi/invertebrates/marine-invertebrates/native-oyster>
- Newell, R., & Mann, R. (2012). *Shellfish aquaculture: Ecosystem effects, benthic-pelagic coupling and potential for nutrient trading, Report to the Secretary of Natural Resources, Commonwealth of Virginia, June 21*.
- Olivier, A., Jones, L., Le Vay, L., Christie, M., Wilson, J., & Malham, S. K. (2020). A global review of the ecosystem services provided by bivalve aquaculture. *Reviews in Aquaculture*, 12(1), 3–25. <https://doi.org/10.1111/raq.12301>
- Olivier, A., van der Vay, S., Malham, L., Christie, S. K., Wilson, M., Allender, J., Schmidlin, S., Brewin, S., J. M., & Jones, L. (2021). Geographical variation in the carbon, nitrogen, and phosphorus content of blue mussels, *Mytilus edulis*. *MARINE POLLUTION BULLETIN*, 167. <https://doi.org/10.1016/j.marpolbul.2021.112291>
- OSPAR (2023). *Riverine Inputs and Direct Discharges (RID)*. OSPAR Commission. <https://www.ospar.org/work-areas/hasec/hazardous-substances/rid>
- Painting, S. J., Collingridge, K., Garcia, L., Barry, J., Leaf, S., Best, M., Miles, A., McAliskey, M., Charlesworth, M., Haines, L., Fryer, R., Walsham, P., Webster, L., Bresnan, E., Roberts, A., Scanlan, C., & Engelke, C. (2018). *Nutrient inputs in water and air: UK Marine Online Assessment Tool*. <https://moat.cefas.co.uk/pressures-from-human-activities/eutrophication/nutrient-inputs/>
- Parker, M., & Bricker, S. (2020). Sustainable oyster aquaculture, water quality improvement, and ecosystem service value potential in Maryland Chesapeake Bay. *Journal of Shellfish Research*, 39(2), 269–281. <https://doi.org/10.2983/035.039.0208>
- Pernet, F., Dupont, S., Gattuso, J.-P., Metian, M., & Gazeau, F. (2025a). Bivalve farming is not a CO2 sink: From myth to legend, where is science? *Reviews in Aquaculture*, 17(4), Article e70067. <https://doi.org/10.1111/raq.70067>
- Pernet, F., Dupont, S., Gattuso, J.-P., Metian, M., & Gazeau, F. (2025b). Cracking the myth: Bivalve farming is not a CO2 sink. *Reviews in Aquaculture*, 17(1), Article e12954. <https://doi.org/10.1111/raq.12954>
- Petersen, J. K., Timmermann, K., Carlsson, M., Holmer, M., Maar, M., & Lindahl, O. (2012). Mussel farming can be used as a mitigation tool—A reply. *Marine Pollution Bulletin*, 64(2), 452–454. <https://doi.org/10.1016/j.marpolbul.2011.11.027>
- Petersen, J. K., Hasler, B., Timmermann, K., Nielsen, P., Tørring, D. B., Larsen, M. M., & Holmer, M. (2014). Mussels as a tool for mitigation of nutrients in the marine environment. *Marine Pollution Bulletin*, 82(1–2), 137–143. <https://doi.org/10.1016/j.marpolbul.2014.03.006>
- Petersen, J. K., Taylor, D., Bergström, P., Buer, A. L., Darecki, M., Filippelli, R., Gren, I. M., Hasler, B., Holbach, A. M., Nielsen, P., Lindegarth, M., Lund, I., Maar, M., Ritzenhofen, L., Sagan, S., Saurel, C., Schernewski, G., Stybel, N., & Timmermann, K. (2020). *Policy guidelines for implementation of mussel cultivation as a mitigation measure for coastal eutrophication in the Western Baltic Sea* (Nos. 362–2020; DTU Aqua Report). Technical University of Denmark, National Institute of Aquatic Resources, 2800 Kongens Lyngby. <https://orbit.dtu.dk/en/publications/policy-guidelines-for-implementation-of-mussel-cultivation-as-a-mitigation-measure-for-coastal-eutrophication-in-the-western-baltic-sea>
- Pinn, E. (2021). *Ecosystem Services, Goods and Benefits Derived From UK Commercially Important Shellfish* (p. 75). Seafish. <http://www.seafish.org/document/?id=35500>
- Pradeepkiran, J. A. (2019). Aquaculture role in global food security with nutritional value: A review. *Translational Animal Science*, 3(2), 903–910. <https://doi.org/10.1093/tas/txz012>
- Preston, J., Gamble, C., Debney, A., Helmer, L., Hancock, B., & Ermgassen, Z., P. S (2020). *European Native Oyster Habitat Restoration Handbook. UK & Ireland*. The Zoological Society of London.
- Pretty, J. N., Mason, C. F., Nedwell, D. B., Hine, R. E., Leaf, S., & Dils, R. (2003). Environmental costs of freshwater eutrophication in England and Wales. *Environmental Science & Technology*, 37(2), 201–208. <https://doi.org/10.1021/es020793k>
- Rabalais, N. N., Turner, R. E., Díaz, R. J., & Justić, D. (2009). Global change and eutrophication of coastal waters. *ICES Journal of Marine Science*, 66(7), 1528–1537. <https://doi.org/10.1093/icesjms/fsp047>
- Ray, N. E., & Fulweiler, R. W. (2021). Meta-analysis of oyster impacts on coastal biogeochemistry. *Nature Sustainability*, 4(3), 261–269. <https://doi.org/10.1038/s41893-020-00644-9>
- Reitsma, J., Murphy, D. C., Archer, A. F., & York, R. H. (2017). Nitrogen extraction potential of wild and cultured bivalves harvested from nearshore waters of Cape Cod, USA. *Marine Pollution Bulletin*, 116(1–2), 175–181. <https://doi.org/10.1016/j.marpolbul.2016.12.072>
- Rose, J. M., Bricker, S. B., & Ferreira, J. G. (2015). Comparative analysis of modeled nitrogen removal by shellfish farms. *Marine Pollution Bulletin*, 91(1), 185–190. <https://doi.org/10.1016/j.marpolbul.2014.12.006>
- Rose, J. M., Gosnell, J. S., Bricker, S., Brush, M. J., Colden, A., Harris, L., Karplus, E., Laferriere, A., Merrill, N. H., Murphy, T. B., Reitsma, J., Shockley, J., Stephenson, K., Theuerkauf, S., Ward, D., & Fulweiler, R. W. (2021). Opportunities and challenges for including oyster-mediated denitrification in nitrogen management plans. *Estuaries and Coasts*, 44(8), 2041–2055. <https://doi.org/10.1007/s12237-021-00936-z>
- Scottish Government, M. D. (2023). *Scottish Shellfish Farm Production Survey Data*. <https://doi.org/10.7489/1917-2>
- Seafish (2023a). *Aquatic profiles—Mussels*. Seafish. <https://www.seafish.org/responsible-sourcing/aquaculture-farming-seafood/species-farmed-in-aquaculture/aquaculture-profiles/mussels/sources-quantities-and-cultivation-methods/>
- Seafish (2023b). *Aquatic profiles—Oysters*. Seafish. <https://www.seafish.org/responsible-sourcing/aquaculture-farming-seafood/species-farmed-in-aquaculture/aquaculture-profiles/oysters/escapes-and-introductions/>
- Sea, M.A., Thrush, S.F., Hillman, J.R. (2021) Environmental predictors of sediment denitrification rates within restored green-lipped mussel *Perna canaliculus* beds. *Marine Ecology Progress Series*, 667, 1–13. <https://doi.org/10.3354/meps13727>
- Silva, C., Yanez, E., Martin-Diaz, M. L., & DelValls, T. A. (2012). Assessing a bioremediation strategy in a shallow coastal system affected by a fish farm culture—Application of GIS and shellfish dynamic models in the Rio San Pedro, SW Spain. *Marine Pollution Bulletin*, 64(4), 751–765. <https://doi.org/10.1016/j.marpolbul.2012.01.019>
- Sinha, E., Michalak, A. M., & Balaji, V. (2017). Eutrophication will increase during the 21st century as a result of precipitation changes. *Science*, 357(6349), 405–408. <https://doi.org/10.1126/science.aan2409>
- Smaal, A. C., Ferreira, J. G., Grant, J., Petersen, J. K., & Strand, Ø. (Eds.). (2019). *Goods and Services of Marine Bivalves*. Springer International Publishing. <https://doi.org/10.1007/978-3-319-96776-9>

- Stephenson, K., Aultman, S., Metcalfe, T., & Miller, A. (2010). An evaluation of nutrient nonpoint offset trading in Virginia: A role for agricultural nonpoint sources? Evaluation of nutrient nonpoint offset trading. *Water Resources Research*, 46(4). <https://doi.org/10.1029/2009WR008228>
- Taylor, D., Saurel, C., Nielsen, P., & Petersen, J. K. (2019). Production characteristics and optimization of mitigation mussel culture. *Frontiers in Marine Science*, 6, Article 698. <https://doi.org/10.3389/fmars.2019.00698>
- Ware, J. R., Smith, S. V., & Reaka-Kudla, M. L. (1992). Coral reefs: Sources or sinks of atmospheric CO₂? *Coral Reefs*, 11(3), 127–130. <https://doi.org/10.1007/BF00255465>
- Watson, S. C. L., Watson, G. J., Mellan, J., Sykes, T., Lines, C., & Preston, J. (2020a). *Valuing the Solent Marine Sites Habitats and Species: A Natural Capital Study of Benthic Ecosystem Services and how they Contribute to Water Quality Regulation* (Environment Agency R&D Technical Report No. ENV6003066R).
- Watson, S. C., Preston, J., Beaumont, N. J., & Watson, G. J. (2020). Assessing the natural capital value of water quality and climate regulation in temperate marine systems using a EUNIS biotope classification approach. *Science of the Total Environment*. <https://doi.org/10.1016/j.scitotenv.2020.140688>
- Watson, S., Watson, G., Mellan, J., Sykes, T., Lines, C., & Preston, J. (2020c). *Valuing the Solent Marine Sites Habitats and Species: A Natural Capital Study of Benthic Ecosystem Services and how they Contribute to Water Quality Regulation: Environment Agency R&D Technical Report ENV6003066R* (No. ENV6003066R). Environment Agency.
- Weber, M. A., Wainger, L. A., Parker, M., & Hollady, T. (2018). *The potential for nutrient credit trading or economic incentives to expand Maryland oyster aquaculture*. https://repository.library.noaa.gov/view/noaa/37930/noaa_37930_DS1.pdf
- Wessex Water (2023). Wareham Water Recycling Centre Nitrogen Removal Scheme. Unpublished report. Onorevole, K.M., Thompson, S.P., Piehler, M.F. (2018). Living shorelines enhance nitrogen removal capacity over time. *Ecological Engineering*, 120, 238–248. <https://doi.org/10.1016/j.ecoleng.2018.05.017>
- Wheeler, T. B. (2020, May 11). *Oyster growers hope polluters will shell out for nutrient credits*. Bay Journal. https://www.bayjournal.com/news/fisheries/oyster-growers-hope-polluters-will-shell-out-for-nutrient-credits/article_d5d4abac-8e1e-11ea-be85-8f3b710e121b.html
- Wijsman, J. W. M., Troost, K., Fang, J., & Roncarati, A. (2019). Global Production of Marine Bivalves. Trends and Challenges. In A. C. Smaal, J. G. Ferreira, J. Grant, J. K. Petersen, & Ø. Strand (Eds.), *Goods and Services of Marine Bivalves* (pp. 7–26). Springer International Publishing. https://doi.org/10.1007/978-3-319-96776-9_2
- Zan, X., Xu, B., Zhang, C., & Ren, Y. (2014). Annual variations of biogenic element contents of manila clam (*Ruditapes philippinarum*) bottom-cultivated in Jiaozhou Bay, China. *Journal of Ocean University of China*, 13(4), 637–646. <https://doi.org/10.1007/s11802-014-2140-5>
- Zu Ermgassen, P. S., Bos, O. G., Debney, A., Gamble, C., Glover, A., Pogoda, B., Pouvreau, S., Sanderson, W. G., Smyth, D., & Preston, J. (2020). *European native oyster habitat restoration monitoring handbook*.

Publisher's Note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.