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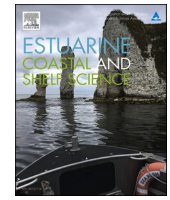

Quantification and valuation of the potential of shellfish ecosystem services in mitigating coastal eutrophication

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ABSTRACT

Eutrophication of coastal waters due to anthropogenic nutrient enrichment is a significant problem in European seas and worldwide. Bivalve shellfish remove nutrients from the water through filtration of suspended particles. This 'bioextraction' capacity is a key regulating ecosystem service that contributes to eutrophication control, but the extent and value of this ecosystem service has not been well established. This study aims to assess the potential of shellfish ecosystem services in mitigating coastal eutrophication in European coastal waters. The estimation of the amount of nutrients removed by different bivalve species under specific culture practices and locations is relevant due to the variety, geographical distribution, and scale at which shellfish species are farmed in Europe.

The approach used here consisted of (a) estimation of nutrient loading to European regional seas, and source apportionment where possible; (b) evaluation of nutrient removal by key species of bivalve shellfish; (c) analysis of the role of shellfish in top-down control of eutrophication as a complement to the established bottom-up approach of emissions reduction.

The nitrogen removal capacity of bivalves was assessed using two complementary approaches: (i) elemental analysis of N concentration in soft tissues and shell; and (ii) modelling the physiology of shellfish at the typical farm. In both cases, the results were upscaled to the shellfish aquaculture production in Western Europe.

Our results show that European shellfish aquaculture can help reduce negative water quality impacts of excess nutrients (nitrogen and phosphorus) in coastal communities. Different shellfish species have different removal rates, which can also be influenced by environmental conditions at the distinct locations and culture practices. The species responsible for the largest N removal (3356–3491 tonnes y^{-1}) was the Mediterranean mussel (accounting for 53–70% of the total) which is also the farmed species with highest production in Europe. Our study shows that the EU annual current production of over half a million metric tonnes of bivalves removes between 4.8 and 6.5 kilotonnes of N per year. The annual cost of removing the same amount of nutrients using other measures would be between 15.9 and 21.6 billion €. This ecosystem service has not been used in Europe as part of a nutrient management framework. The results of this study aim to provide the basis for strategic guidelines to include shellfish aquaculture in watershed-scale nutrient management policies in the EU, following the same principles as the on-going programmes in the United States.

1. Introduction

Excessive nutrient loading to the coastal zone has long been recognized as a causative factor of water quality impairment (Bricker et al., 2003; de Jonge and Elliott, 2001; Diaz and Rosenberg, 2008; OECD, 1982; OSPAR, 1999; Jesper et al., 2006; Elliott et al., 1999; Smith et al., 2006) and it is expected to increase during the 21st century as a result of precipitation changes, increasing population, and climate change (Paerl et al., 2014; Rabalais et al., 2010; Seitzinger et al., 2010; Sinha et al., 2017). This impairment falls into the broad category of eutrophication,

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characterized by a well-established set of symptoms or indicators including high concentrations of phytoplankton and/or macroalgae, harmful algal blooms, loss of submerged aquatic vegetation (SAV), and hypoxia (Bricker et al., 2007; Camargo and Alonso, 2006; OSPAR, 2017a; HELCOM, 2018b; Smith et al., 2006; Xiao et al., 2007). Eutrophication effects interfere with water-based recreational and economic activities such as tourism, aquaculture and fisheries, leading to substantial financial losses (Gobler, 2020; Heiskanen et al., 2019; Kemp et al., 2005).

The excess nutrients may be taken up by extractive species, either directly by inorganic extractive aquaculture, e.g. seaweeds uptake of dissolved forms, but also indirectly through uptake of organic particulates, e.g. shellfish. The *indirect* re-use of dissolved nitrogen and phosphorus, after conversion into particulate organic forms through primary production, is a key step in the removal of these compounds from coastal ecosystems; this is largely mediated by filter-feeding shellfish (Gerritsen et al., 1994; Higgins et al., 2011; Petersen et al., 2014, 2016; Ferreira and Bricker, 2016). When shellfish are harvested, the N and P contained within the tissue and shell are removed from the marine environment and returned to land.

In Europe and North America, there is an increasing awareness of the potential of shellfish culture to offset the symptoms of eutrophication. However, so far implementation of bivalve culture in this role in marine European waters is sparse (Taylor et al., 2019). In 2004, Sweden was the first country to use mussels to compensate for discharge of nitrogen from a sewage treatment plant during a 6-year trial period (Lindahl and Kollberg, 2009; Lindahl and Söderqvist, 2011). In Denmark, mitigation cultures have recently been accepted by the Danish Nature Agency as a potential measure for the removal of excess nutrients in Danish coastal waters (Eriksen et al., 2014; Petersen et al., 2016). In recent years, cultivation practices designed specifically for nutrient extraction, the so-called mitigation culture, have emerged (Petersen et al., 2016). In Denmark, nutrient extractive potential by bivalves has been tested and optimized to exhibit similar rates to highly efficient constructed terrestrial reduction mechanisms (Petersen et al., 2014; Taylor et al., 2019).

Eutrophication assessment frameworks (e.g. Bricker et al., 2003, 2008, 2018; Ferreira et al., 2007, 2011; Greenwood et al., 2019) have been developed to help inform successful development and implementation of water quality legislation in North America (e.g. United States Clean Water Act [Public Law 92-500]) and Europe (e.g. the Water Framework Directive WFD—2000/60/EC and the Marine Strategy Framework Directive MSFD—2008/56/EC). These indices have also been applied as part of the toolset for compliance with quality requirements such as the WFD Good Ecological Status (GEcS) and MSFD Good Environmental Status (GEnS) in Europe. The policy measures recommended for management of coastal eutrophication have largely focused on bottom-up control, achieved principally through reduction of land-based inputs.

In parallel, the role of filter-feeders in top-down control of phytoplankton in semi-enclosed coastal systems such as estuaries, fjords, and bays has been understood for many years (e.g. Cloern, 1982, Li et al., 2005; Lindahl et al., 2005; Shastri and Diwekar, 2006; Xiao et al., 2007), but only recently has been applied in a limited number of waterbodies (e.g. Reitsma et al., 2017; Cornwell et al., 2016) in Europe and North America as an integrated management tool. In support of this concept, US federal agencies have specifically recognized the environmental benefits of shellfish aquaculture (e.g. NOAA National Shellfish Initiative, 2011). In 2022, NOAA released a five-year strategic plan to promote aquaculture in the US that explicitly includes recognition of beneficial nutrient removal capabilities among other ecosystem services provided by bivalve aquaculture (NOAA Aquaculture Strategic Plan, 2022).

By contrast, in SE Asia and China bivalves are cultivated on a large scale in many bays and play a major role in maintaining water quality through removal of phytoplankton and detrital organic material from the water column (Sanggou Bay MOM, Pia Kupke-Hansen etc). This increases the underwater light penetration and short-circuits the organic decomposition cycle—both processes contribute to improved oxygenation by reducing bacterial oxygen uptake and promoting SAV growth. Although eutrophication-related legislation in Asia (GB 3097-1997; MEE, 2020; NBS, 2020) does not explicitly include top-down control, the intensity of both bivalve and seaweed aquaculture in coastal areas clearly acts as a significant management measure.

In recent years, various authors (e.g. Ferreira and Bricker, 2016; Kellogg et al., 2014; Newell et al., 2005; Ray and Fulweiler, 2021; Rose et al., 2021; Stephenson et al., 2010, 2017) have explored the possibility of incorporating bivalve shellfish aquaculture into nutrient trading frameworks that can internalize management measures and contribute to mitigation of eutrophication in both Europe and North America.

The combination of source abatement and nutrient re-use through top-down control of primary production by cultivated shellfish such as oysters, mussels, and clams fits into the concept of the circular economy, where waste materials can be effectively recycled to provide environmental and economic benefits ([ACR+], 2021; European Commission, 2017, 2020; FAO, 2022).

For a circular approach of this kind to be successful, four key criteria must be satisfied:

1. The loading of nutrients (nitrogen and/or phosphorus) reaching the coastal zone must be quantified, a regional breakdown must be made, source apportionment into point and diffuse emissions must be determined, and information on which nutrient (N or P) is limiting to primary production must be available[1];
2. An assessment of biomass production of filter-feeding bivalve shellfish, broken down by region, must be obtained. This will provide guidance on the relevance of the use of bivalves for top-down control and strategic planning information on where the activity should be promoted;
3. An evaluation of nutrient removal by bivalves on a unit basis must be carried out—various approaches are available for this and will be reviewed below;
4. A methodology for determination of the value of ecosystem services provided should be put in place, using for instance an approach based on avoided or replacement cost.

The contribution of bivalves to nutrient removal can be approached in two ways, both of which rely on a mass balance approach. The first, now in use in parts of the US, is based on the nutrient content of shellfish tissue and shell, that once harvested remove nutrients from the water. Because marine and estuarine coastal systems are generally nitrogen limited (Elser et al., 2007; Paerl, 2018; Paerl et al., 2008), emphasis is placed on N rather than P (e.g. in Reitsma et al., 2017; Grizzle et al., 2016), and the ecosystem service provided is predicated on harvest (landings). The concept is that N is physically removed from the waterbody when the organisms are removed (Reitsma et al., 2017). Tissue N content is determined through analysis of proximate composition (e.g. Grizzle et al., 2016; Higgins et al., 2011; Kellog et al., 2013), and landings are based on fisheries data (e.g. FAO, Eurostat). In a regional approach, a nutrient Best Management Practice (BMP) was approved for use in addressing legally required reductions in

[1] On a global scale, nutrient addition in estuarine and coastal environments is largely contributed by anthropogenic N and P, particularly from N. In the present study, we focused on nitrogen as currently there is a strong consensus among the scientific community that excess N loading is the primary cause of eutrophication in coastal ecosystems (Howarth and Marino, 2006). According to the EC Nitrates Directive, eutrophication is defined as 'the enrichment of water by nitrogen compounds causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of water concerned' (Anonymous, 1991).

Chesapeake Bay (Virginia and Maryland; Cornwell et al., 2016). The BMP is for harvested aquacultured oyster tissue only. An analysis in Mashpee River (Massachusetts; Town of Mashpee Sewer Commission, 2015) determined that culturing oysters for harvest, including both tissue and shell, could remove 50% of the legally required N removal needed to achieve water quality goals.

A complementary approach recognizes that N is a causative factor rather than a symptom of eutrophication and therefore focuses on removal of both phytoplankton and particulate detritus through shellfish feeding. This approach, which typically uses mathematical models, quantifies the net N removed from the water column through filtration of algae and detritus, and must therefore account for loss terms: N returned to the system through pseudofaeces, undigested food, excretion, and mortality (including dislodgments from the cultivation structures). Several models exist for different bivalve species, using both Net Energy Balance (NEB) (Brigolin et al., 2009; Saurel et al., 2014; Cubillo et al., 2018) and Dynamic Energy Budget (DEB) (Kooijman, 2010; Fuentes-Santos et al., 2019) formulations—the only constraint of such models is that the feeding behaviour and other physiological aspects are described in such a way as to allow quantification of net nitrogen uptake.

In the present work, we consider the whole of Western Europe and perform an assessment of nitrogen removal by farmed bivalves, taking into account the major species cultivated in European seas. This large-scale analysis is intended as a first estimate to provide information about the potential impact of bivalve nutrient removal in Europe, by region. Implementation of comprehensive nutrient management strategies that might include use of bivalves will be conducted at the relevant catchment – waterbody scale. The specific objectives of this study are:

1. To determine the mass of nitrogen removed by cultivated bivalves in the various Western European regions, and the role of particulate organic extractors in N loading abatement;
2. To assess the economic value of the regulating ecosystem service provided by bivalve aquaculture in eutrophication management.

2. Methodology

2.1. Nutrient loading to European Seas

A comprehensive literature review was performed to collate information on nitrogen and phosphorus inputs to European marine waters, in a similar way as in Ferreira and Bricker (2019). The results of this review are summarized in Table 1. The area covered by each of the regional European seas was used to determine the normalized nutrient loading per unit of surface area, which is shown in Table A1. Where possible, the nutrient sources have been categorized into: (i) point-sources, including industry, sewage, and finfish aquaculture effluents; (ii) diffuse sources, which mostly include riverine discharges; and (iii) atmospheric deposition, which although technically a diffuse source has been added as a separate category because it cannot easily be related to nutrient offset trading programs at the watershed level.

2.2. Bivalve production in Western Europe

Shellfish production data was extracted from Eurostat (2022) using bespoke REST web services software (Maritime and Environmental Thresholds for Aquaculture – META, Longline Environment Ltd., 2020) to identify the main shellfish species farmed in Europe and to upscale the impact of bivalve aquaculture to the country and EU level. The 2018 production data for five key commercial shellfish species was used: blue mussel (*Mytilus edulis*), Mediterranean mussel (*Mytilus galloprovincialis*), Pacific oyster (*Magallana gigas*), European flat oyster (*Ostrea edulis)*, and Manila clam (*Ruditapes philippinarum*).

Table 1
N and P loads to marine waters in 10^3 tonnes y^{-1} (% total in brackets) for European seas categorized according to the source of nutrients. The sub-total loading to each regional sea discriminate, when possible, the inputs from point-, diffuse, and atmospheric sources, and the % of the sub-total is shown in brackets. When available, sub-categories of each of these three sources of nutrients have been included, and the correspondent percentage with respect to each source is also shown within brackets.

Areas	Sources	Total N (10^3 tonnes y^{-1})	Total P (10^3 tonnes y^{-1})	Year
Norwegian Sea	**Point-sources** [a]	**35.9 (56)**	**5.86 (88)**	2012
	Industry[a]	*0.84 (1.3)*	*0.09 (1.4)*	2012
	Sewage effluents[a]	*3.53 (5)*	*0.43 (6)*	2012
	Finfish aquaculture (salmon & trout in Norway) [b]	*31.6 (49)*	*5.50 (81)*	2012
	Diffuse sources	**28.7 (44)**	**0.77 (12)**	
	Riverine loads[a]	*10.0 (16)*	*0.38 (6)*	2012
	Unmonitored areas[a]	*18.7 (29)*	*0.38 (6)*	2012
	Sub-total [a]	**64.7**	**6.63**	2012
Barents Sea	**Point-sources** [a]	**4.11 (37)**	**0.68 (82)**	2012
	Industry[a]	*0.05 (0.5)*	*0.003 (0.4)*	2012
	Sewage effluents[a]	*0.25 (2)*	*0.03 (4)*	2012
	Finfish aquaculture (Norway)[a]	*3.80 (35)*	*0.65 (78.3)*	2012
	Diffuse sources	**6.90 (63)**	**0.15 (17.5)**	
	Riverine loads[a]	*4.80 (44)*	*0.11 (13)*	2012
	Unmonitored areas[a]	*2.10 (19)*	*0.04 (4.5)*	2012
	Sub-total [a]	**11.0**	**0.83**	2012
Baltic Sea	**Point-sources**[d]	**28.9 (3.5)**	**1.6 (5.2)**	2014
	Finfish aquaculture [c]	*0.85 (0.1)*	*0.11 (0.3)*	2013
	Diffuse sources (riverine loads) [d]	**573 (69.4)**	**29.3 (94.8)**	2014
	Atmospheric deposition[d]	**224 (27.1)**		2014
	Sub-total [d]	**826**	**30.9**	2014
Greater North Sea	**Point-sources** [f]	**200 (13)**	**32 (82)**	2005
	Finfish aquaculture[e]	*15 (0.94)*		2014
	Diffuse sources (riverine loads) [f]	**800 (53)**	**5 (13)**	2005
	Atmospheric deposition[e]	**500 (33)**	**2 (5)**	2014
	Sub-total [e]	**1500**	**39**	2014
Celtic Seas	**Sub-total** [e]	**275**	**12.5**	2014
Bay of Biscay & Iberian Coast	**Sub-total** [e]	**450**	**12**	2014
Black Sea	**Point-sources** [f]	**160 (23)**	**20 (51)**	2005
	Diffuse sources (riverine loads) [f]	**330 (47)**	**11.5 (30)**	2005
	Other sources[f,i]	210 (30)	7.5 (19)	2005
	Sub-total [f]	**700**	**39**	2005
Mediterranean Sea - only Europe	**Point-sources** [f]	**300 (32)**	**40 (57)**	2005
	Finfish aquaculture[h]	46 (5)	8 (11)	modelled
	Diffuse sources (riverine loads) [f]	**470 (49)**	**22 (31.4)**	2005
	Other sources[f,i]	180 (19)	8 (11.4)	2005
	Sub-total [f]	**950**	**70**	2005
Mediterranean Sea- Europe and N. Africa	**Point-sources** [g]	**314 (17)**	**39 (36)**	2016
	Diffuse sources [g]	**1532 (83)**	**69 (64)**	2003–2007
	Sub-total [g]	**1845**	**109**	2003–2007
Total European seas	**Point-sources** [f]	**1000 (25)**	**150 (65)**	2005
	Diffuse sources (riverine loads) [f]	**2200 (55)**	**50 (22)**	2005
	Other sources[f,i]	800 (20)	30 (13)	2005
	Sub-total [f]	**4000**	**230**	2005
	Total (excluding N. Africa)	*4777*	*211*	

(*continued on next page*)

Table 1 (*continued*)

Areas	Sources	Total N (10^3 tonnes y^{-1})	Total P (10^3 tonnes y^{-1})	Year
	Total (including N. Africa)	*5672*	*249*	

[a] Skarbøvik et al. (2013).
[b] Stålnacke et al. (2009), Skarbøvik et al. (2011).
[c] HELCOM (2015).
[d] HELCOM (2018a).
[e] OSPAR (2017b).
[f] Bouraoui et al. (2011).
[g] Malagó et al. (2019).
[h] Due to the lack of available data, nutrient discharges from finfish aquaculture to the Mediterranean Sea were modelled for the two most important species, using the FARM model. We used the typical farming practice for sea bass and gilthead seabream in Turkey and Spain, obtained from the CERES EU H2020 project (https://ceresproject.eu/). The modelled finfish nutrient discharges were upscaled using the reported production for both species in 2019, obtained from the Federation of European Aquaculture Producers and FAO estimates (Apromar, 2019).
[i] Including atmospheric deposition, scattered dwellings and biological fixation.

2.3. Nutrient removal by shellfish aquaculture

Bivalve shellfish remove nutrients from the water through filtration of particles. Part of the digested material is used for growth of tissue and shell while the rest is expelled as faeces, pseudofaeces, or ammonia. Shellfish filtration rates are a function of several environmental factors, including temperature, salinity, food quantity and quality (Bayne, 1993, 2017; Barillé-Boyer et al., 1997; Cerco and Noel, 2005). In addition, different shellfish species exhibit significant variation in filtration rates, with the rates generally increasing with species size (Cranford and Grant, 1990; Prins et al., 1991; Powell et al., 1992; Smaal and Zurburg, 1997). In this study, the associated removal of nutrients was estimated in two ways: (i) using the nitrogen content per organism tissue and shell by means of elemental analysis; and (ii) through a mass balance of intake (filtration of algal and detrital POM) and loss of (i) nitrogen in organic material through pre-ingestive selection (pseudofaeces), undigested food (faeces), and natural mortality; and (ii) metabolic losses of inorganic nitrogen through excretion by the modelled population.

2.3.1. Elemental analysis estimation of shellfish nutrient removal

The carbon and nitrogen content of shellfish flesh and shells was determined by combustion elemental analysis on a Thermo Flash 2000 elemental analyser. In the case of mussels, the byssus threads were removed. Prior to analysis, shellfish flesh and shells were dried to a constant weight at 60 °C and ground to a fine powder. Subsamples of each were then weighed and encapsulated in pre-weighed tin cups for analysis.

The mass of carbon and nitrogen within each shell or flesh sample was then calculated as (Eq. (1)):

$$S_{C,N} = S_{dw} \times \left(\frac{P_{C,N}}{100}\right) \quad (1)$$

Where $S_{C,N}$ is the mass of carbon or nitrogen within the sample, S_{dw} is the dry weight of the sample and $P_{C,N}$ is the percentage carbon or percentage nitrogen in the sub-sample analysed by the elemental analyser.

2.3.2. Individual-based modelling of shellfish nutrient removal

The individual shellfish models used in this project are based on the generic AquaShell™ framework which has been developed and parameterized for several shellfish species, and validated for different locations across Europe (Ferreira et al., 2008a, 2008b, 2022; Sequeira et al., 2008; Nunes et al., 2011), the United States (Saurel et al., 2014; Bricker et al., 2015, 2018, 2020; Rose et al., 2015), and other parts of the world (Nobre et al., 2010; Santa Marta et al., 2020).

These individual growth models use a net energy balance (NEB) approach and have been published elsewhere for the shellfish species studied here: see Cubillo et al. (2017) for Mediterranean and blue mussels; Ferreira et al. (2023) for Pacific oyster; Ferreira et al. (2021) for European flat oyster; and Saurel et al. (2014) for Manila clam.

These bioenergetic models have been validated for the shellfish species and locations studied here and were used to predict shellfish growth, reproductive effort, and overall mass balance for the whole culture cycle at the individual level (Cubillo et al., 2021; Saurel et al., 2014).

The AquaShell models were updated to include a more realistic representation of the nitrogen content in detrital organic matter, which can constitute an important part of the diet of bivalve shellfish. Data from Enriquez et al. (1993) show a median C:N mass ratio of 20.8:1 in detrital POM (range 12.6–51.1, n = 25), substantially higher than the Redfield Ratio (C:N in mass ratio of 45:7, or 6.4:1).

This is consistent with data given in Valiela (1995), based on work by Valiela, Teal, Hobbie, Swain, Wilson, and Buchsbaum, and by Valiela et al. (1985), which show N to be 1–2% of dry weight for *Spartina alterniflora*. A conversion factor of POM to carbon of 0.38 (Grant and Bacher, 1998) results in a C:N mass ratio of 38:1 to 19:1.

The median value from Enriquez et al. (1993) was used to parameterise AquaShell, and individual shellfish models were then incorporated into the well-known Farm Aquaculture Resource Management (FARM) model to determine both production (i.e. harvestable biomass) and environmental effects.

The FARM model is applied at the local scale, *i.e.* for a typical farm. The culture practice is collated with stakeholders to define a farm area, including stocking density, seed and harvest size, and timing of the cultivation cycle. Measured water currents are used to simulate transport of water and associated properties through a farm area (typically divided into various segments or boxes), allowing the simulation of growth and environmental effects using an individual-based model (IBM). FARM simulates the production over a culture cycle and calculates net removal of algal and detrital carbon and nitrogen by bivalve shellfish.

The net nitrogen removal for a farm population over the culture cycle is simulated by means of a mass balance approach: FARM models (a) the sink terms, *i.e.* the feeding behaviour responsible for the gross intake (clearance rate) of phytoplankton and non-phytoplankton organics; and (b) the source terms, *i.e.* the return of nitrogen to the environment through pseudofaeces, faeces, excretion, mortality, and spawning.

The model also provides information on chlorophyll drawdown, oxygen consumption, loss of particulate matter to the sediment, and can be used to perform a marginal analysis on the profitability of bivalve culture relative to stocking density (Ferreira et al., 2007).

FARM has been used to model a broad range of shellfish species and geographical regions covering a wide range of aquaculture practices in both intertidal and subtidal environments (e.g. Cubillo et al., 2021; Ferreira et al., 2021).

The mass balance provides a value for net removal of nitrogen from the water column by farmed shellfish, which effectively equates to a drawdown of phytoplankton, one of the primary symptoms of eutrophication. The model then converts the net N removed by shellfish feeding and harvest to human population-equivalents (PEQ; 3 kg N y^{-1}) and calculates the potential economic value of the ecosystem service represented based on the costs of alternative nutrient management strategies.

In this work, the FARM individual-based population model (IBM) described in Ferreira et al. (2021) was applied. This IBM version has many advantages over the conventional population-dynamics equations, as it allows simulation of the typical variance within the cultivated population and eliminates artefacts that lead to excessively rapid population growth due to numerical diffusion. Individuals are assigned a probabilistic fitness at seeding (in the model) in terms of assimilation efficiency (AE, ±0–5% of the mean). This simulates genetic variation within the single cohort of organisms typically deployed for growout on a farm. In parallel, the model allows for size-dependent mortality, with smaller organisms having a higher death rate; in combination, the stochastic variation of AE and mortality generate a realistic Gaussian population distribution.

Population dynamics are determined by culture strategies and natural mortality. For each case study, FARM was parameterised with an accurate description of the culture practice used for pre-defined 'typical farms', considered to be representative for each species and region. Table A2 summarises the culture practice used for the model runs.

2.4. Potential contribution of shellfish N removal in Western Europe

N removal estimates, obtained using both methods (elemental analysis and modelling approach), were scaled up to European production level. For the elemental analysis approach, the mean values of the percentage of N in live weight of each species (Table 2) were scaled up to European production, (Eq. (2)):

$$N_{TE} = N_E \times P_{EU} \tag{2}$$

Where N_{TE} represents the total nitrogen removed by the estimated live weight production (tonnes); P_{EU} is the production for each species (tonnes); and N_E (%) is the content of nitrogen as a proportion of the total mass of the animal. This process was repeated to estimate the removal of N for each of the shellfish producing countries.

For the modelling approach, the scaling to EU production was estimated using data from Table 3 using (Eq. (3)):

$$N_{TF} = \frac{1}{1000}\left(\frac{P_{EU}}{P_F} N_F\right) \tag{3}$$

Where N_{TF} is the total net nitrogen removal determined for the EU using FARM (kg y^{-1}); N_F is the net nitrogen removal determined with FARM for a typical farm (kg y^{-1}); P_{EU} is the corresponding species production (tonnes); P_F is the modelled shellfish biomass obtained at the end of the culture cycle at the local (farm)-scale in FARM (tonnes). This process was repeated to estimate the removal of N on each of the shellfish producing countries.

2.5. Economic valuation of shellfish ecosystem services in mitigating coastal eutrophication

The cost avoided, or replacement cost method, recently used in evaluations of several US water bodies (Bricker et al., 2018, 2020; Parker and Bricker, 2020; Dvarskas et al., 2020) was applied here to estimate the economic value of the N removal by European bivalve aquaculture. This method provides the substitution or 'avoided' cost of land-based nutrient removal that would serve as additional revenue to the farmer in a nutrient credit trading program (Cornwell et al., 2016; Ferreira and Bricker, 2016; Bricker et al., 2018). Restoration of a part of the ecosystem (in this case, clean water) through N removal by wastewater treatment, and agricultural and urban BMPs are the most likely candidates to replace the N removal services by bivalve shellfish aquaculture (Barrett et al., 2022). The costs of those nutrient management strategies, taken from Rose et al. (2015), Ferreira and Bricker (2019), and Parker and Bricker (2020) were used here to provide a useful estimate of the potential economic value of N removal by oyster bioextraction.

Table 2
Percentage carbon and nitrogen contents in the shell and flesh of different shellfish species (% of live weight, in fresh mass).

		% of C in total wet weight			% of N in total wet weight		
Species	Origin	Min	Mean	Max	Min	Mean	Max
Blue mussel	Belfast Lough (NI)	10.1	11.1	14.8	0.76	0.88	1.12
Pacific oyster	Dundrum Bay (NI)	10.8	10.9	11.2	0.28	0.37	0.48
Flat oyster	Lough Foyle (NI)	6.01	8.80	9.97	0.19	0.29	0.32
Med mussel	Sagres (Portugal)	10.5	10.7	17.4	0.96	1.00	1.13
Manila clam	Venice Lagoon (Italy)	8.15	8.50	11.2	0.28	0.32	0.36

3. Results and discussion

3.1. Nutrient loading to European seas

The N and P discharges to the different European Seas are summarized in Table 1, and the normalized nutrient loading per ocean surface area is shown in Table A1. These data discriminate, when possible, the source-apportionment of nutrient loads, which is a key element for policy decisions.

According to the most recent estimates, the nutrients discharged to the whole European seas, per ocean surface area (including N. African discharges) is 731 and 32.2 kg km^{-2} y^{-1} for N and P, respectively. Without the N. African discharges, the loading reduces to 616 and 27.2 kg km^{-2} y^{-1} for N and P, respectively.

The European waters that receive highest N discharges are the Greater North Sea (1500 × 10^3 tonnes y^{-1}), Mediterranean Sea[2] (950 × 10^3 tonnes y^{-1}) and Baltic Sea (826 × 10^3 tonnes y^{-1}), while the Arctic region receives the lowest discharges: Norwegian Sea (65 × 10^3 tonnes y^{-1}) and Barents Sea (11 × 10^3 tonnes y^{-1}). The European waters that receive highest P discharges are the Mediterranean Sea[3] (70 × 10^3 tonnes y^{-1}), the Greater North Sea and the Black Sea (39 × 10^3 tonnes y^{-1} each), while the Arctic region receives, again, the lowest discharges: Norwegian Sea (6.6 × 10^3 tonnes y^{-1}) and Barents Sea (0.8 × 10^3 tonnes y^{-1}). The highest unit loading is to the Baltic Sea, with 2096 kg N km^{-2} y^{-1} and 78.4 kg P km^{-2} y^{-1}, followed by the Greater North Sea (with 1957 kg N km^{-2} y^{-1} and 50.9 kg P km^{-2} y^{-1}) and the Black Sea (with 1517 kg N km^{-2} y^{-1} and 84.5 kg P km^{-2} y^{-1}). The OSPAR IV region (Bay of Biscay and Iberian Coast) follows with 857 kt of N km^{-2} y^{-1}. With respect to the P, the highest inputs are those to the Black Sea (84.5 kg P km^{-2} y^{-1}), followed by the Baltic Sea, with 78.4 kg P km^{-2} y^{-1}, and the Greater North Sea (with 50.9 kg P km^{-2} y^{-1}). If North African discharges are included, the Mediterranean Sea has a similar N load (733 kg N km^{-2} y^{-1}) to the Celtic Sea (750 kg N km^{-2} y^{-1}), while if only European discharges to the Mediterranean are considered, there is an estimated loading of 377 kt N km^{-2} y^{-1}. The Arctic region has the lowest load when looking at the estimates per surface area, with 58.3 and 6.9 kg N km^{-2} y^{-1} and 6.0 and 0.5 kg P km^{-2} y^{-1} for the Norwegian and Barents Sea, respectively.

3.1.1. Sources of nutrient discharges

Nutrient loads to these coastal areas include (i) point sources, such as wastewater treatment plants (WWTPs) and septic tanks; (ii) diffuse sources, such as agricultural and urban run-off; and (iii) atmospheric deposition from industry, agriculture, fossil fuel combustion in power plants and vehicles. Important for successful nutrient management is knowledge of the sources in order to prioritize management strategies.

[2] 950 kt y^{-1} is the N loading from European countries to the Mediterranean Sea, excluding North African discharges. If we also consider the N discharges from N. Africa this value doubles to 1845 kt y^{-1} and thus the Mediterranean Sea would be the main contributor to N loading.

[3] 70 kt y^{-1} is the P loading from European countries to the Mediterranean Sea, excluding North African discharges. If we also consider the P discharges from N. Africa this value rises to 109 kt y^{-1}.

Table 3
Potential nitrogen removal through the culture cycle based on FARM model simulations, using the typical culture practice (based on typical farms) for the key bivalve shellfish species cultivated in Europe.

Species	Farm location	Harvestable biomass (ton ha^{-1} y^{-1})	APP (–)	N removal (kg ha^{-1} y^{-1})	N removed in harvest (%)	N removal (g ind^{-1})	Ecosystem service (PEQ ha^{-1} y^{-1})
Manila clam	Italy	106	65	752	0.71	0.07	228.0
Med. mussel	Portugal	20	10	206	1.04	0.10	62.5
Pacific oyster	Northern Ireland	33	78	281	0.85	0.59	85.3
Blue mussel	Netherlands	4	47	75	1.74	0.17	22.6
Flat oyster	Northern Ireland	21	286	154	0.74	0.66	153.7
	Average	*37*	*97*	*294*	*1.0*	*0.32*	*110.4*

This information is also useful in determining which sector might be most willing to buy nutrient credits within a fully developed nutrient credit trading program.

In most European seas the main load of N is diffuse (Fig. 1). An exception to this is the Norwegian Sea, due to the high N loadings from salmon and trout aquaculture in Norway that are considered as point-source inputs. The N inputs from Norwegian finfish aquaculture affect both the Norwegian and the Barents Seas, encompassing 49% and 35% of the N loading to those Seas, respectively. The majority of the P discharge to most European seas comes from point sources (Fig. 2). Only the Baltic Sea receives greater P inputs from diffuse sources (90%) than from point sources (5%), with atmospheric deposition[4] accounting for the remaining 5% (data from 2010 in HELCOM, 2015). The marked decrease in sewage effluents from 2005 to 2010 could explain the low P inputs from point sources (HELCOM, 2015).

When we consider the North African discharges to the Mediterranean Sea, most of the nutrient discharges come from diffuse sources (83% and 64% for N and P, respectively). The source apportionment for the whole Mediterranean region is mostly controlled by the situation of its main river basins including the Nile, Po, Ebro and Rhone River basins. The Nile accounts for 62% and 53% of the N and P diffuse loading, respectively (Malagó et al., 2019). If we consider only European loadings to the Mediterranean Sea, most of the P loading come from point sources (57% versus 31.4% from diffuse sources).

The relative contribution of finfish aquaculture to the total nutrient loading is much higher in the Norwegian and Barents Seas due to the large-scale Norwegian salmon and trout industry. In fact, finfish aquaculture represents 49% and 81% of the N and P discharges to the Norwegian Sea, respectively, and 35% and 78% of the N and P inputs to the Barents Sea.

3.2. Bivalve production in Western Europe

The five selected species in this study accounted for 95% of shellfish farmed production in Europe (Table A3) and between 40% and 45% of all farmed aquatic organisms (Eurostat, 2022; Longline Environment Ltd, 2020). Overall shellfish harvest in the EU was 585 kt in 2018, with the largest producing countries being Spain and France (Fig. 3). Spanish bivalve production accounted for 41% of 2018 harvest and was dominated by Mediterranean mussel farming. France accounted for 24% of the total, with Pacific oysters as the main farmed shellfish species, corresponding to 65% of French shellfish production in 2018. Italy is the third largest producer in the EU, with 15.4% of the total production, of which one third is Manila clam culture and two thirds Mediterranean mussels.

Mediterranean mussel is the most abundant farmed species, and its production in Europe was led by Spain, with 72% of the total volume harvested (EUMOFA, 2019), followed by Italy (18%) and Greece (6%). The higher production volume of mussels does not reflect on the overall value of these animals, when compared with oysters or clams. Despite contributing over 72% of volume to the total EU production, Spanish Mediterranean mussel farming accounts for 65% in value, due to the lower unit price. Blue mussel is the other important species, with the Netherlands and France accounting for 41 and 40% of all the EU blue mussel harvest in 2018, respectively. Pacific oysters were the most valuable shellfish of the five species analysed, being valued at more than both mussel species combined. European flat oyster harvest represents less than 3% of total oyster production in the EU, nonetheless this is an important species from an economic standpoint due to its high market value. Flat oyster production is mainly concentrated in four countries, which comprise over 90% of the farmed flat oyster in the EU (Table A3). In 2018 France was the largest producer, followed by Spain, the Netherlands, and Ireland. Much of the more recent development of this species is due to its perceived biodiversity value and contribution to ecosystem services (Bertolini and Pastres, 2022).

3.3. Nutrient removal by shellfish aquaculture

Shellfish filtration can enhance water clarity, allowing sufficient light penetration to support maintenance and expansion of seagrass habitat (Newell and Koch, 2004), and water quality by concentrated deposition of faeces and pseudofaeces (particles collected on the gills that the shellfish do not use as food; Newell, 2004; Newell and Koch, 2004; Newell et al., 2007). This benthic-pelagic coupling removes particulate organic matter (POM) from the water column, making it available to benthic detritivores, such as polychaetes and amphipods, which are a key element of the food chain for many estuarine and inshore coastal fish.

Increased biodeposition of organic matter in sediments can lead to increased bacterial denitrification that can help to remove N from estuarine systems (unless such deposition leads to hypoxic conditions which suppress nitrification; Childs et al., 2002). The associated bacteria in sediments of an oyster bed can remove 20% or more of the N in oyster wastes, using the same process that is used in modern wastewater treatment plants (Shumway et al., 2003).

Filter-feeding shellfish not only remove N from the water column to the benthos; they also incorporate N into their tissues. For example, (dry) Eastern oyster tissue has a mean N content ranging from 5.64% to 9.72%, and (dry) shell content is 0.12–0.3% (Kellogg et al., 2014 and citations therein; Sebastiano, 2013; Grizzle et al., 2016). These results compare well with data from the present study, for Pacific oysters and European flat oysters. Pacific oyster (dry) tissue had a mean nitrogen content ranging from 5.69% to 6.85%, and (dry) shell content was 0.08–0.27%. European native oyster (dry) tissue had a mean N content ranging from 4.38% to 8.84%, and (dry) shell content was 0.11–0.37%. When the shellfish are harvested, the N is removed from the system,

[4] The atmospheric deposition has not been included as a diffuse source in our calculations because the idea of nutrient offsets and trading is based on integrated watershed management.

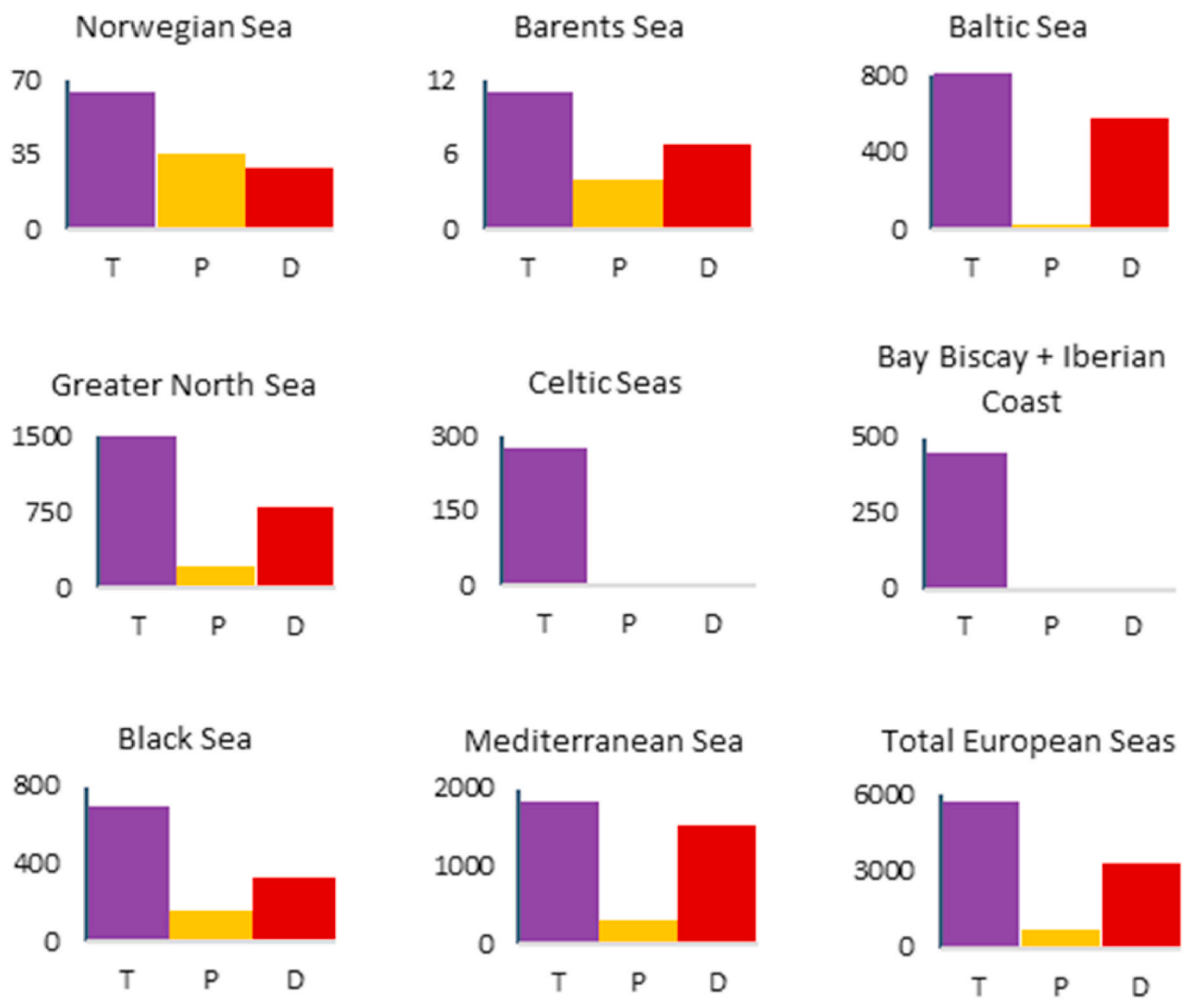

Fig. 1. Proportion of N inputs to the different European seas from point sources (P), diffuse sources (D) and total loading (T). The discharges to the Mediterranean Sea and the total European seas include loading from North Africa.

thereby recycling nutrients from sea to land (Shumway et al., 2003; Lindahl et al., 2005). In the oysters, about half of this N is in the relatively heavy shell (Kellogg et al., 2013; STAC, 2013). In contrast, when species with lighter shells, such as blue mussels, are harvested, less N is removed (Newell and Koch, 2004).

By interacting with water column phytoplankton dynamics and benthic denitrification, shellfish are likely to reduce nutrients that stimulate primary production in coastal waters, which often leads to low dissolved oxygen levels (hypoxia) as a result of organic decomposition, a serious environmental problem in many aquatic ecosystems worldwide (Atlantic States Marine Fisheries Commission, 2007; Diaz and Rosenberg, 2008).

3.3.1. Elemental analysis estimation of shellfish nutrient removal

Soft tissue and shell from the five main shellfish aquaculture species in the EU were analysed for C and N. Table 2 shows the percent carbon and nitrogen content of Manila clams from Italy; Mediterranean mussels from Portugal; and blue mussels, Pacific oysters and flat oysters from Northern Ireland. The mean C content was similar for all the species and ranged from 8.5% in Manila clams to 11.1% in Mediterranean mussels. The mean N content presents greater variability and ranged from 0.3% in flat oysters and Manila clams to 1.0% in Mediterranean mussels.

Elemental analysis estimates, coupled with the detailed shellfish production volumes for each country, allowed a baseline calculation of the N removed in 2018 by the main shellfish producing countries (Table 4). The bulk of N removal is due to mussel production, 69.6% from Mediterranean mussel and 20.0% from blue mussel. Spain removes 50.5% of the N, due to its high production volume of Mediterranean mussels. Nitrogen removal by blue mussels is relatively well distributed across all producing countries, with the Netherlands and France accounting for the highest proportion. Of the countries analysed, France has the second highest value of N removal (16.2%), due mainly to its production of blue mussels and Pacific oysters. Italy is by far the main Manila clam producer, and has the third highest N removal, 14.8%, due mainly to the >61000 tonnes of Mediterranean mussel harvest.

3.3.2. Individual-based modelling of shellfish nutrient removal

A combination of physiological and local-scale production models was employed to assess the capacity of some of the main EU cultivated shellfish species to reduce the primary symptoms of eutrophication. These models enable us to simulate the volume of water cleared by each species in the 'typical farm', the changes in concentration of key indicators such as chlorophyll and dissolved oxygen, and the mass balance of eutrophication-related nutrients such as N.

The simulation results of the FARM IBM local-scale model were annualised and normalized to the size of each 'typical farm' to allow for comparison (Table 3). The FARM model simulates biomass production at the end of the culture cycle and economic indicators such as the return on investment or APP (Average Physical Product). FARM estimates the N removed by the whole farmed population (harvestable and undersized individuals) after a typical cultivation cycle.

Table 3 shows highly variable yields, from 4 to 106 tonnes ha^{-1} y^{-1}

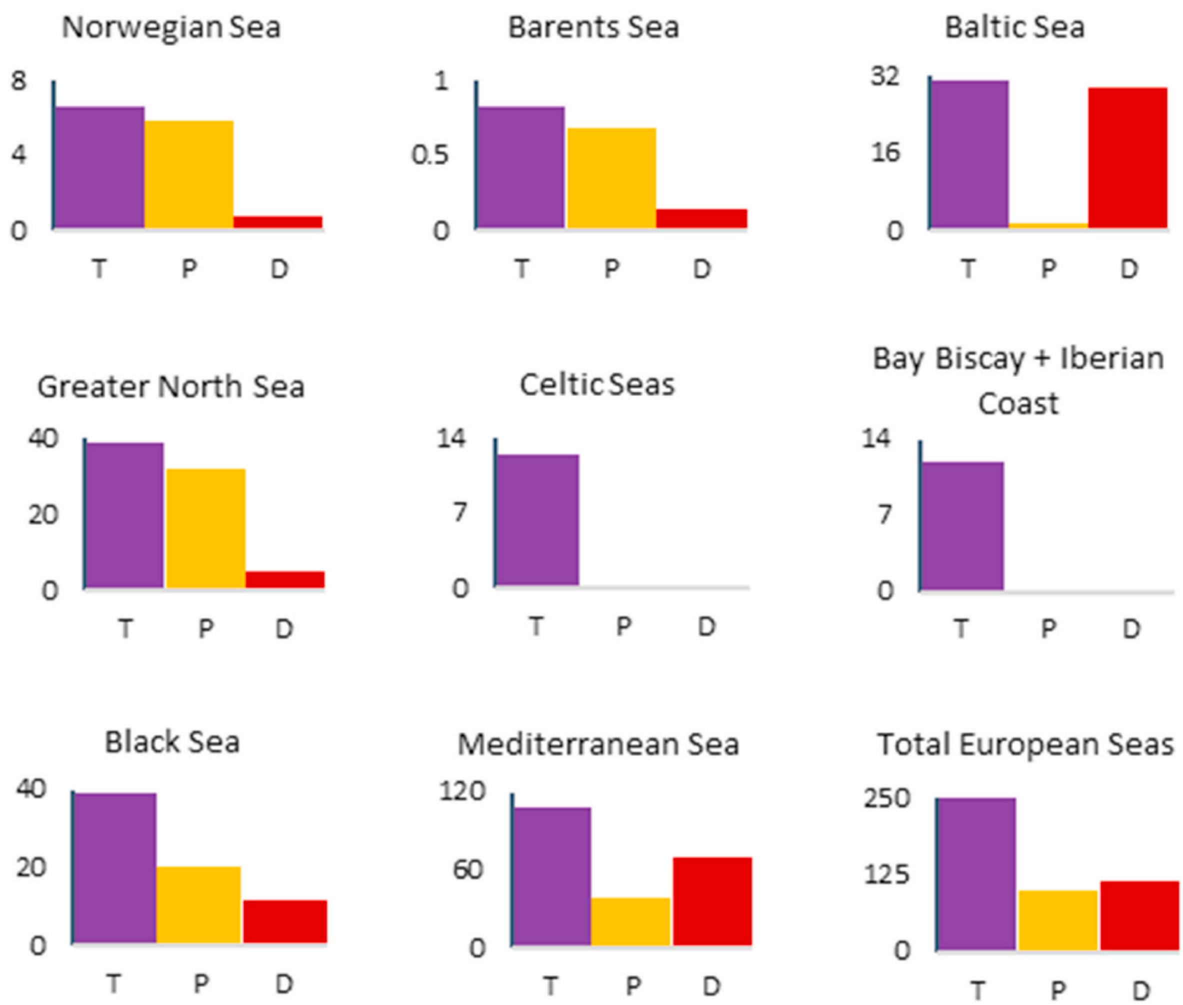

Fig. 2. Proportion of P inputs to the different European seas from point sources (P), diffuse sources (D) and total loading (T). The discharges to the Mediterranean Sea and the total European seas include loading from North Africa.

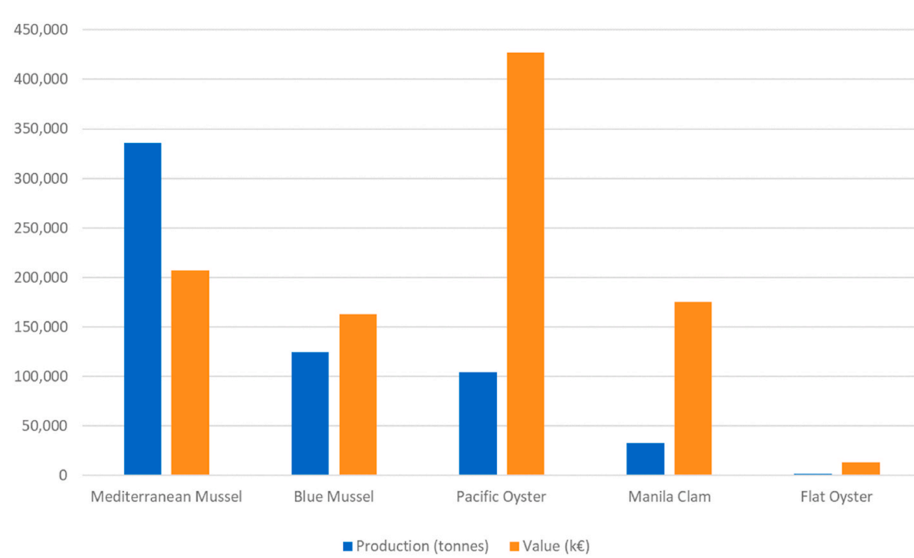

Fig. 3. Annual production (tonnes) and production value (k€) in the EU in 2018 for five bivalve species (from left to right): *Mytilus galloprovincialis, Mytilus edulis, Magallana gigas, Ruditapes philippinarum* and *Ostrea edulis*.

Table 4
N removed for each species in the key EU producer countries, estimated using the elemental analysis approach. Volumes are presented in tonnes for 2018.

	Blue mussel (tonnes y^{-1})	Mediterranean mussel (tonnes y^{-1})	Pacific oyster (tonnes y^{-1})	Flat oyster (tonnes y^{-1})	Manila clam (tonnes y^{-1})	Total (tonnes y^{-1})
UK	125		8.2	0.05		133
Netherlands	398			0.94		399
France	387	46	342	2.12	3.34	780
Ireland	122		31.1	0.73		153
Spain	0	2426	2.9	0.98	2.77	2433
Italy	0	614	0.3		99.1	713
Others	58	269	4.1	0.21		331
Total EU	965	3356	388	5.03	105	4819

depending on the species, which reflect the very different cultivation strategies (e.g. stocking densities) implemented at each site (see Table A2 for details on the farming practice). Similar to other nutrient mitigation methods, response to variability in conditions over the year, site-specific attributes, and cultivation practices can affect nutrient extraction efficiency (Rose et al., 2015; Taylor et al., 2019). The blue mussel culture in the Netherlands has the lowest yields, caused by low stocking densities, while Manila clam has the highest yields and the highest stocking densities (1000 ind. m^{-2}).

Among bivalves, oyster species removed more N in their tissues (around 0.6–0.7 g per harvestable-sized oyster), while clams remove the least amount (0.07 g of N per harvestable-sized clam). The average N removal was close to 0.32 g of N per individual.

The average annual yield was 37 tonnes ha^{-1} y^{-1}, with mean N removal of 294 kg ha^{-1} y^{-1}, and average N in harvestable biomass of 1%, which leads to an ecosystem service of 110 PEQ ha^{-1} y^{-1}. The culture of Manila clams in Italy and Native oysters in Northern Ireland provide the highest ecosystem services (Table 3).

The N removal estimated by the FARM model was scaled up to Western Europe production to provide a whole sector overview of the European nitrogen removal capacity (Table 5). As for the elemental analysis approach, the FARM model also places Mediterranean mussels as the main shellfish species for nitrogen removal, responsible for 53% of the total N removed by the five species. Blue mussels and Pacific oysters, with much less harvest volumes, are the second and third largest N removal crops, accounting for 29 and 14%, respectively. This result is similar to the elemental analysis approach, which also assigns more than double N removal capacity to blue mussels (20%) than Pacific oysters (8.1%).

The 53% N removed by Mediterranean mussel is mostly concentrated in Spain, which accounts for 39% of the European N removal. France is the second contributor, with 25% of the N removal, mostly from Pacific oyster and blue mussel culture. Italy and the Netherlands are in third place, responsible for 13 and 12% of the total N removal, respectively. Two thirds of Italian removal come from Mediterranean mussels and one third from Manila clams. In the Netherlands most of it comes from blue mussel aquaculture.

When comparing the capacity for N removal of each species using both approaches, the FARM IBM model leads to higher values for all species, except Med mussels (Fig. 4). The reason is the analytical approach estimates the N removed by the average harvestable-sized individuals, and only considers the fraction of shellfish physically taken out of the water. On the other hand, the modelling approach considers the N removal by the whole population, which also includes the undersized individuals that are left in the water but still contributed to N removal, since once assimilated into tissue and shell nitrogen is no longer available to support phytoplankton growth (Rose et al., 2015).

Although part of the farmed shellfish is not harvested and remain in the water, the eutrophication symptoms are reduced (Ferreira and Bricker, 2016). This reduction promotes increased water clarity, and the effect of shellfish, while in the water, is to short-circuit the eutrophication process, reducing secondary symptoms of eutrophication such as hypoxia (through reduction of suspended organic matter) and loss of submerged aquatic vegetation (through improved underwater light climate). As these authors pointed out, the modelling approach is useful to estimate the N removal in a shellfish natural reef with little or no human intervention, where there is little or no gathering of shellfish.

Both approaches show that blue and Mediterranean mussels have the highest capacity for N removal, reaching over 8 tonnes of N removed per 1000 tonnes of mussel harvested. Oysters and clams have a lower potential for N removal, ranging from 2.9 to 3.7 (analytical) and from 5.6 to 8.1 (modelling) tonnes of N removed for each 1000 tonnes harvested (Fig. 4).

3.4. Potential contribution of shellfish N removal in Western Europe

The N removed by each shellfish species (Tables 4 and 5) was grouped by country and by regional sea, and was compared to the N loading to each European sea (Table 1). The percentage of N from the total N loadings removed by shellfish production was estimated for each sea, using both approaches (Table 6).

Depending on the approach considered, shellfish aquaculture in Europe was able to remove between 0.12% and 0.17% of total nitrogen inputs in 2018. This amount can vary considerably, depending on which area is being analysed. For the Baltic and Norwegian Seas, the estimated N removal by farmed shellfish is very low (0.01–0.02% of N inputs), while in the Bay of Biscay and Iberian coast this percentage can go as high as 0.63–0.75%, due to the high-volume production of Mediterranean mussels in Spain.

Other figures can be estimated from these data, such as the N waste

Table 5
N removed for each species in each of the main EU producer countries, estimated using the FARM model. Volumes are presented in tonnes for 2018.

	Blue mussel (tonnes y^{-1})	Mediterranean mussel (tonnes y^{-1})	Pacific oyster (tonnes y^{-1})	Flat oyster (tonnes y^{-1})	Manila clam (tonnes y^{-1})	Total (tonnes y^{-1})
UK	248		19	0.1		267
Netherlands	791			2.4		794
France	769	48	784	5.3	7	1614
Ireland	242		71	1.9		315
Spain		2524	7	2.5	6	2540
Italy		639	1		220	859
Other	115	280	9	0.5		405
Total EU	1917	3491	891	12.7	234	6545

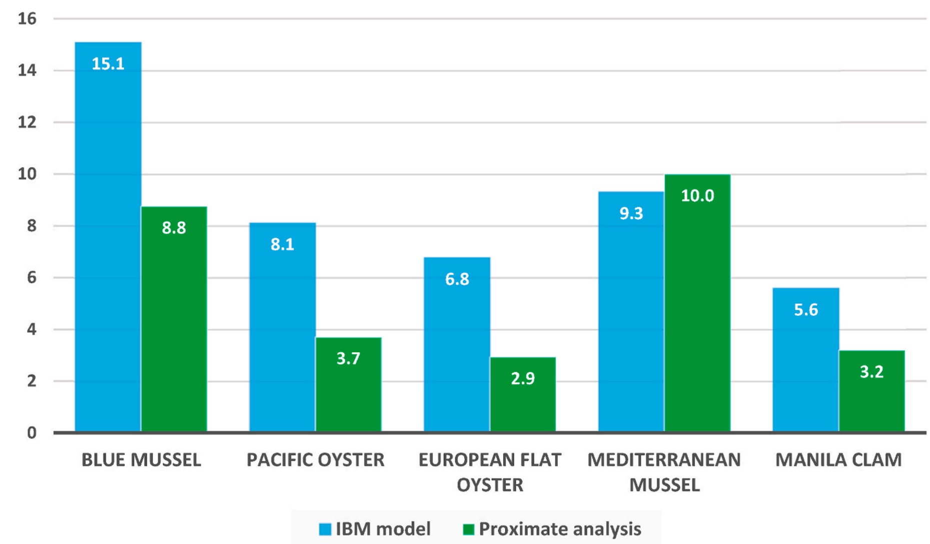

Fig. 4. Comparison between N removal estimates from the FARM IBM model and the elemental analysis, for the 5 top commercial shellfish species in the EU, in kg of N per ton of shellfish harvested. The IBM model estimates the N removed by the of the whole population in the farm, which includes the harvestable sized and the undersized individuals, while the proximate analysis estimates the N removed by the average harvestable-sized individuals.

Table 6
Nitrogen loading to European seas (adapted from Table 1), and nitrogen removed by shellfish aquaculture on each area as a percentage of total loads. Nitrogen removal percentages calculated based on the country of production, according to the FARM and elemental analysis approaches. N removal estimates and loads do not account for N atmospheric deposition in this table, because atmospheric deposition values were only available for the Baltic and Greater North Sea. The regional seas without N removal results is where we could not find reliable shellfish production estimates from Eurostat.

European sea	Nitrogen loading	Nitrogen removal (%)	
	Tonnes	FARM	Elemental analysis
Norwegian Sea	64,600	0.02	0.01
Barents Sea	11,010	–	–
Baltic Sea	601,900	0.01	0.004
Black Sea	700,000	–	–
Greater North Sea	1,000,000	0.10	0.05
Celtic Seas	275,000	0.17	0.08
Bay of Biscay and Iberian Coast	450,000	0.75	0.63
Mediterranean Sea - only Europe	950,000	0.20	0.14
Total	4,052,610	0.17	0.12

from finfish aquaculture that could be removed by shellfish farming, which according to our N loading values, would vary between 5.0% according to the elemental analysis and 6.7% with the FARM model.

From our estimations, current European bivalve aquaculture could remove between 4.8 and 6.5 kt of N per year, depending on the assessment method used (Table 7). This removal estimate represents an ecosystem service equivalent to water treatment for 1.5–2.0 million people, considering a per-person annual load of 3.3 kg N year^{-1} (Ferreira et al., 2007).

3.5. *Economic valuation of shellfish ecosystem services in mitigating coastal eutrophication*

A compelling aspect of the bioextraction discussion is the potential economic value of the water filtering ecosystem service provided by bivalve shellfish, and whether growers should be paid for the N removal capacity within a nutrient credit trading program (Cornwell et al., 2016; Ferreira and Bricker, 2016; Bricker et al., 2018; Parker and Bricker, 2020). Converting the benefits of the N removal ecosystem services to a common comparable unit (Dollars or Euros) represents a major challenge to economists (Peterson and Lipcius, 2003). Here, the cost of alternative nutrient management strategies was used to provide an estimated range of economic values for the N removal by European bivalve aquaculture, in the same manner as Barrett et al. (2022). The

Table 7
Financial benefits of an European nutrient credit trading framework to include shellfish farmers, in euros and USD, considering an exchange rate of 1USD = 1.02€ (November 23rd' 2022). The reported costs for the different categories of nitrogen removal strategies were taken from Rose et al. (2015). For detailed information about these costs (location, practice, reference, etc.), please see Supplemental Table S2 in Rose et al. (2015). The average remediation cost for each strategy was used to estimate the credit value.

Nitrogen removal		Minimum (Analytical)				Maximum (Model)
Nitrogen removed by shellfish (tonnes per year)		4819				6545
Population-Equivalents (PEQ @ 3.3 kg N per ind.)		1,460,240				1,983,458
	Remediation cost (€ kg^{-1} N)	*Credit valuation (Million € y^{-1})*		*Remediation cost (USD lb^{-1} N)*	*Credit valuation (Million USD y^{-1})*	
Value of eco-intensification		*Analytical*	*Model*		*Analytical*	*Model*
Approved agricultural BMP	0.2–1057	2547	3459	0.1–470	2497	3392
Stormwater control measures	67.5–8161	19825	26926	30–3629	19437	26398
Wastewater treatment upgrades	1.1–17113	41236	56005	0.5–7610	40427	54907
Wetlands	1.3–481	1163	1579	0.60–214	1140	1548
Average credit valuation		**16193**	**21993**		**15875**	**21561**

range of reported costs for the categories of N removal strategies considered in this study vary widely (Table 7).

As limits of technology are approached, nitrogen reductions at point sources become increasingly more expensive with upgrades sometimes costing hundreds of millions of dollars (Dvarskas et al., 2020). As efforts to upgrade wastewater treatment plants and reduce other point sources of N succeed, an increasing portion of the total N load to coastal and estuarine waters comes from non-point sources (HELCOM, 2015). The diffuse nature of non-point source N makes it challenging to address within the context of an ecosystem-scale N management program (Heiskanen et al., 2019). A wide variety of best management practices (BMPs) have been developed to deal with agricultural and stormwater runoff, two major categories of non-point source N. These BMPs can be expensive, may incur regular maintenance effort and cost, and typically need to be implemented throughout the watershed (Houle et al., 2013; Meals et al., 2010; Stephenson et al., 2010).

There are several assumptions required to apply the avoided costs or replacement costs approach: if bivalve shellfish are no longer harvested the N removal services they provide would need to be replaced by alternate nutrient management strategies, there is equivalency of N removal services among bivalve nutrient removal and other nutrient reduction strategies, the avoided cost good is the least cost for N removal, there is willingness to pay because there is recognized nutrient impairment of waterbodies and a need for N load reductions to meet water quality goals (Freeman et al., 2014). In the present study, the estimated benefits of incorporating cultivated shellfish into catchment-scale nutrient management programs could potentially be significant, ranging from 16 to 22 billion euros per year, depending on the assessment method used (Table 7). Promising results of bivalve bioextraction were also observed by Barrett et al. (2022). They found that an average of 275–581 kg N ha^{-1} y^{-1} could be removed via bioextraction at oyster, mussel and seaweed farms, and this additional N removal could be worth 84–505 USD t^{-1} in locations where nutrients are a management priority, based on N offset values in the United States and Europe.

None of these N removal strategies will solve the problem of coastal eutrophication alone. A successful watershed-scale N management program will likely incorporate aspects of all these categories. Quantification and valuation of N removed through shellfish harvest can be used by policy-makers and resource managers to assess the available set of nutrient management tools to design a comprehensive nutrient management plan that will most efficiently and cost effectively achieve water quality goals. Resource managers will need to balance efficacy of N removal strategies, cost, and available space to implement comprehensive N control plans. Recognizing all dimensions of the benefits and providing economic incentives to shellfish farmers for the nutrient reduction services could foster industry expansion (Rose et al., 2015). Inclusion of this ecosystem service in nutrient credit trading programs would be a logical mechanism for funding these payments, so that shellfish aquaculture could become part of the comprehensive, ecosystem-based nutrient management in European estuaries.

4. Conclusions

Eutrophication due to anthropogenic nutrient enrichment of coastal waters is a major issue in European seas. European water bodies are susceptible to the direct and indirect effects of excessive nutrient loading, and although point-source emissions are currently more controlled, diffuse emissions are a more complicated and expensive issue to solve (European Commission and Fabbri, 2020; Heiskanen et al., 2019; OSPAR et al., 2010). Due to the origin of such emissions, substantial changes might well be required to agriculture and livestock management, leading to high social and financial burdens to rural communities.

The bioextraction capacity of bivalve shellfish is an important regulating ecosystem service that contributes to eutrophication control, but to date it has not been used in Europe as part of a management framework. In other parts of the world such as the USA, there are examples of working nutrient credit trading schemes where bivalves form part of the overall N budget in integrated catchment management programmes (e.g. Cornwell et al., 2016; Town of Mashpee Sewer Commission, 2015). The present study examined the potential for including shellfish aquaculture in watershed-scale nutrient management policies in the EU and is a first estimate of the potential for use of bivalve shellfish within a management program. Actual management will be implemented on a waterbody – catchment level but this analysis suggests that bioextraction can be an important addition to nutrient management in some European regions.

The major challenge to the use of bivalves for nutrient management is the determination of the quantity of nutrients that are removed by these filter-feeding species. The range of environments and the diversity of farming practices is also a challenge in estimating nutrient removal, although it is also evidence of the extent to which the European coastal ecosystem is used to farming bivalves, across a large number of countries, often with significant production volumes.

The approach used in this work consisted in (a) estimation of nitrogen loading to European regional seas, and whenever possible source apportionment; (b) evaluation of nutrient removal by five key species of bivalve shellfish, using both a laboratory analysis of shellfish composition and a mathematical model of growth, the well-established FARM model; (c) evaluation of the role of shellfish in top-down control of eutrophication based on production figures, as a complement to the well-established bottom-up approach of emissions reduction.

Results show that different shellfish species have different nutrient (nitrogen) removal rates. Location and farm culture practices were also shown to affect shellfish removal rates. The cultivation of filter-feeders or seaweeds close to other fed-cultivated species such as finfish in integrated multitrophic aquaculture systems may play a role in nutrient recycling at the local (farm)- and bay-scale, that can be quantified using a similar methodology to the one proposed in this study.

Ninety five percent of all shellfish farmed in the EU is composed of the five species analysed in this article: blue mussel, Mediterranean mussel, Pacific oyster, flat oyster, and Manila clam. Together, these species make up 45% of all aquaculture production in the EU. Shellfish production was greater than 580,000 tonnes in 2018, and the N removed from European seas was estimated to be between 4819 tonnes and 6545 tonnes, depending on the assessment method used. Expressed as population equivalents, these numbers correspond to offsetting the emissions of 1.5–2.0 million Europeans.

In financial terms, the potential benefits of incorporating cultivated shellfish into a catchment-scale nutrient management scheme are significant. The estimated costs saved by shellfish aquaculture in Europe ranged between 16 and 22 billion €, depending on how N removal was estimated. These are average values estimated from the average remediation cost for each category and the maximum estimates are considerably higher. It is recognized that this potential valuation is associated mostly with diffuse loads which are challenging to reduce in many rural areas in Europe without severe economic and social consequences.

Nutrient management at the catchment scale is in line with other policy instruments such as the WFD, which aim to manage watersheds in an integrated manner across the various types of waterbodies. This type of study reflects the importance of having protocols to asses water quality parameters, concentration of nutrients, oxygen levels, harmful algal blooms, or potential effects on ecosystems through the aquaculture sector. Top-down control of eutrophication via shellfish aquaculture is recognized in qualitative terms but there has been no associated policy development at a European or national level.

The draft of policies to control eutrophication is most likely to succeed when using an ecosystem approach to manage human activities that impact the marine environment. This can be accomplished through the promotion of a sustainable use of ecosystem goods and services and by a stronger coupling of policies at the land-water interface.

Services provided by shellfish are not limited to nutrient removal in tandem with restoration of other nature-based solutions there are major societal benefits including greater food security, local employment, shoreline protection and cleaner waters, beneficial for local populations and for tourism. Shellfish farming, with its reduced ecological footprint, net removal of organic material, and low food-web nutritional requirements, is perhaps the best example of eco-intensification for blue growth.

CRediT authorship contribution statement

Alhambra Martínez Cubillo: Writing – review & editing, Writing – original draft, Investigation, Formal analysis, Data curation. **Andre Sobral Lopes:** Writing – original draft, Visualization, Investigation, Formal analysis, Data curation. **João G. Ferreira:** Writing – review & editing, Supervision, Software, Resources, Project administration, Methodology, Funding acquisition, Conceptualization. **Heather Moore:** Supervision, Project administration, Methodology, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Matthew Service:** Writing – review & editing, Supervision, Resources, Project administration, Funding acquisition, Conceptualization. **Suzanne B. Bricker:** Writing – review & editing, Writing – original draft, Supervision, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

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

Data availability

Data will be made available on request.

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Appendix

Table A.1
Nitrogen and phosphorus loading to marine waters (10^3 tonnes km^{-2} y^{-1}) for European seas.

Region	Source	Total nitrogen (kg km^{-2} y^{-1})	Total phosphorus (kg km^{-2} y^{-1})	Year of data
Norwegian Sea - I				
	Point-sources	32.4	5.3	2012
	Industry	0.8	0.1	2012
	Sewage effluents	3.2	0.4	2012
	Finfish aquaculture (salmon and trout in Norway)	28.5	5.0	2012
	Diffuse sources (riverine loads)	9.0	0.3	2012
	Unmonitored areas	16.9	0.3	2012
	Sub-total	**58.3**	**6.0**	2012
Barents Sea - I				
	Point-sources	2.6	0.4	2012
	Industry	3.3E-02	1.9E-03	2012
	Sewage effluents	0.2	2.1E-02	2012
	Finfish aquaculture (Norway)	2.4	0.4	2012
	Diffuse sources (riverine loads)	3.0	0.1	2012
	Unmonitored areas	1.3	2.3E-02	2012
	Sub-total	**6.9**	**0.5**	2012
Baltic Sea				
	Atmospheric deposition	555	5.3	2013
	Waterborne (riverine + point-sources)	1925	91.9	2010
	Diffuse sources (riverine loads)	1846	87.6	2013
	Point-sources	77.4	4.3	2013
	Finfish aquaculture	2.2	0.3	2013
	Sub-total	2480	97	2013
	Sub-total (normalized)	**2036**	**81.7**	2013
Baltic Sea				
	Atmospheric deposition	568		2014
	Point-sources	73.4	4.1	2014
	Diffuse sources (riverine loads)	1455	74.3	2014
	Sub-total (most up-to-date)	**2096**	**78.4**	2014
Greater North Sea - II				
	Atmospheric deposition	652	2.6	2014
	Waterborne	1304		2014
	Diffuse sources (riverine loads)	1044	6.5	2005
	Point-sources	261	41.7	2005
	Finfish aquaculture	20	3.4	2014
	Sub-total	**1957**	**50.9**	2014
Celtic Seas - III				
	Sub-total	**750**	**34.1**	2014
Bay of Biscay and Iberian Coast - IV				
	Sub-total	**835**	**22.3**	2014
Black Sea				
	Point-sources	347	43.3	2005

(*continued on next page*)

Table A.1 (*continued*)

Region	Source	Total nitrogen (kg km^{-2} y^{-1})	Total phosphorus (kg km^{-2} y^{-1})	Year of data
	Diffuse sources (riverine loads)	715	24.9	2005
	Other sources*	455	16.3	2005
	Sub-total	**1517**	**84.5**	2005
Mediterranean Sea - Europe				
	Point-sources	119	15.9	2005
	Diffuse sources (riverine loads)	187	8.7	2005
	Other sources*	71.5	3.2	2005
	Sub-total	**377**	**27.8**	2005
Mediterranean Sea- Europe and N. Africa				
	Point-sources	125	15.5	2003–2007
	Diffuse sources	609	28	2003–2007
	Sub-total	**733**	**43.1**	2003–2007
Total European seas				
	Point-sources	129	19.3	2005
	Diffuse sources (riverine loads)	284	6.4	2005
	Other sources*	103	3.9	2005
	Total (excluding N. Africa)	**516**	**29.7**	2005
	Total (excluding N. Africa)	**616**	**27.2**	This study
	Total (including N. Africa)	**731**	**32.2**	This study

Table A.2
Culture practice for the typical shellfish farms in the different European regions. These data were used to parameterise the FARM model.

	Blue mussel Oosterschelde Netherlands	Med. mussel Sagres Portugal	Pacific oyster Carlingford Lough Northern Ireland	Flat oyster Lough Foyle Ireland	Manila clam Sacca di Goro Italy
Farmed area (ha)	*8.0*	*10.1*	*18*	*60*	*0.25*
Culture structures	*Subtidal bottom*	*Suspended longlines*	*Intertidal trestles*	*Subtidal bottom*	*Subtidal bottom*
Seed cost (€ kg^{-1})	*1.0*	*1.0*	*5–10*	*25*	*1.0*
Sale price (€ kg^{-1})	*0.8*	*0.65*	*4.4*	*5.0*	*10*
Stocking density (ind m^{-2})	*100*	*312*	*160*	*100*	*1000*
Mortality (% cycle^{-1})	*10*	*10*	*25*	*44*	*30*
Seed weight (live weight)	*0.2*	*1.0*	*0.75*	*0.2*	*0.12*
First seeding day	*120*	*150*	*150*	*150*	*90*
Culture period (days)	*794*	*550*	*1035*	*1000*	*270*
Harvest weight (g live weight)	*10–20*	*10–20*	*>70*	*>90*	*>10*

Table A.3
Shellfish production (tonnes) in the main European producing countries.

	Blue mussel	Mediterranean mussel	Pacific oyster	Flat oyster	Manila clam	Total by country
UK	14,247		2220	19		16,486
Netherlands	45,482			319		45,801
France	44,192	4652	92,225	721	1045	142,836
Ireland	13,889		8385	250		22,524
Spain		242,725	785	334	868	244,712
Italy		61,415	80		30,991	92,486
Other	6608	26,918	1096	72		34,695
Total EU-28 production	110,172	335,710	104,792	1715	32,904	585,293

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