This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement N° 773330

Deliverable report for

GAIN

Green Aquaculture Intensification in Europe

Grant Agreement Number 773330

Deliverable 2.9

Report & white paper on framework for a nutrient credit trading policy for Europe, integrating shellfish producers

Due date of deliverable: 30/04/2021
Actual submission date: 06/12/2021

Lead beneficiary: LLE

Authors: Ferreira, J.G.; Cubillo A.M.; Lopes, A.S.; Marteleira, R.; Service, M.; Moore, H.; Hunter, B.; Cromie, H.; Bricker, S.B.

WP 2 – Enhancement of Secondary Outputs

Task 2.4 – Valorisation of shellfish regulatory services

Dissemination Level:		
PU	**Public**	**Y**

Document log

Version	Date	Comments	Author(s)
V 1	15/12/2020	TOC draft	All
V 1.1	01/03/2021	Nutrient loading data	Cubillo, A.M.
V 1.2	02/03/2021	IBM	Cubillo, A.M.
V 1.3	19/03/2021	Shellfish Production	Lopes, A.S.
V 1.4	25/03/2021	TOC update	All
V 1.5	22/04/2021	Case studies, other	Bricker, S; Cubillo, A.M.; Lopes, A.S.
V 1.6	26/04/2021	Case study, other	Ferreira, J.G.
V 1.7	28/04/2021	Formatting, other	Lopes, A.S.
V 1.8	04/05/2021	Policy recommendations	All
V 1.9	05/05/2021	Executive summary	Lopes, A.S
V 1.10	10/05/2021	Final review	Ferreira, J.G.
V 1.11	15/10/2021	Update	All

The project received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under grant agreement nº 773330

Recommended Citation

Ferreira, J.G.; Cubillo A.M.; Lopes, A.S.; Marteleira, R.; Service, M.; Moore, H.; Hunter, B.; Cromie, H.; Bricker, S.; 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. 58 pp.

The project received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under grant agreement nº 773330

GLOSSARY OF ACRONYMS

Acronym	Definition
AE	Absorption Efficiency
CERES	Climate change and European aquatic RESources
EC	European Commission
EU	European Union[1]
FAO	Food and Agriculture Organization
FARM	Farm Aquaculture Resource Management
FEAP	Federation of European Aquaculture Producers
HELCOM	Helsinki Commission (Baltic marine protection)
IBM	Individual-Based Model
META	Marine and Environmental Thresholds for Aquaculture
NEB	Net Energy Balance
OSPAR	Oslo/Paris convention
PEQ	Population equivalent
SWAT	Soil and Water Assessment Tool
TFW	Total Fresh Weight

[1] In this report, all UK-related references are pre-Brexit, thus including the United Kingdom as a member state. All EU data include the UK for estimates of production, trade or other values, as applicable.

The project received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under grant agreement nº 773330

Press Release

EU Shellfish Aquaculture Improves Water Quality, Study Shows

European shellfish aquaculture can help reduce negative water quality impacts of excess nutrients (nitrogen and phosphorus) in coastal communities, according to a recent study funded by the European Union's Horizon 2020 programme. The study, conducted by a multi-national team during the GAIN project (Green Aquaculture Intensification in Europe http://www.unive.it/gainh2020_eu) examined the potential for including aquaculture of mussels, oysters, and clams in watershed-scale nutrient management policies.

Nutrient discharges to coastal waterbodies can stimulate excessive growth of algae leading to water quality degradation with consequences such as low oxygen, dead fish, and/or harmful algal blooms. These nutrient-related impacts have been reported for many EU estuaries.

Nutrient pollution, or eutrophication, is typically controlled by preventing nutrient discharges to coastal waters through management measures including wastewater treatment and careful use of agriculture fertilizer. Wastewater treatment has been very successful in reducing direct loads, but reductions that require major changes in agriculture, livestock, and community management are economically costly and may have severe social consequences.

Growing bivalve shellfish provides direct economic benefits to a community by supporting jobs and making local seafood available to consumers. It also provides ecosystem services—benefits that nature provides to people—including reductions of algae, which are eaten by the clams, oysters and mussels. The shellfish absorb nutrients into their tissue and shell and remove algae and nutrients from the waterbody, contributing to the environmental sustainability of estuaries, bays, and coastal zones.

The removal of algae by filter-feeding bivalve shellfish is an important and economically valuable ecosystem service—in the USA, compensation to shellfish farmers for the water clearance service they provide is at an advanced stage of debate; in the Chesapeake Bay, growers have been paid for services provided by oyster aquaculture.

Results of this study will provide the basis for strategic guidelines to develop a nutrient credit trading programme in Europe. Our study shows that EU annual production of over half a million metric tons of bivalves removes between 5 and 13 thousand tons of nitrogen per year. The annual cost of removing the same amount of nutrients using other measures would be between 18 and 48 billion €.

"Our hope is that our approach will be useful throughout the EU, and that our positive results will help inform discussions about the value of shellfish aquaculture to water quality, in addition to seafood

The project received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under grant agreement nº 773330

provision," said Prof. Roberto Pastres, coordinator of the GAIN project.

Prof. Pastres added “We recommend the inclusion of bivalves within comprehensive nutrient management plans. Shellfish farming, with its reduced ecological footprint, net removal of organic material, and low food-web nutritional requirements, is perhaps the best example of nature-based intensification for blue growth.”

The project received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under grant agreement nº 773330

Table of Contents

The project received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under grant agreement nº 773330

1. Executive summary

Eutrophication due to anthropogenic nutrient enrichment of marine waters is a major issue in European seas. These 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 which end up in the coastal zones are a more complicated and expensive issue to solve. 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 communities.

The bioextraction capacity of bivalve shellfish is a key regulatory 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 a part of the overall N budget as a part of integrated catchment management.

The estimation of the amount of nutrients removed by different bivalve species under specific culture practices and in different bays or estuaries is relevant in the EU due to the variety, geographical spread and scale at which shellfish species are farmed in Europe.

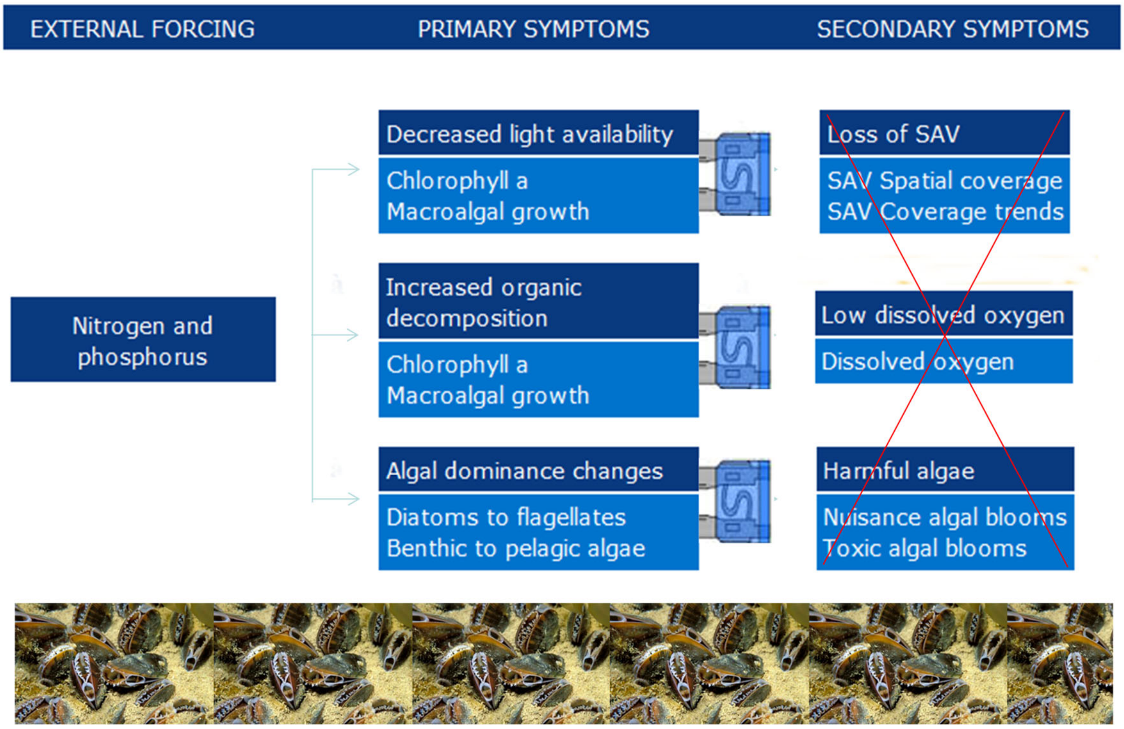


Fig. 1. General conceptual scheme of eutrophication, including top-down control by filter-feeding shellfish. The boxes for primary and secondary symptoms (identical to direct and indirect effects), show the symptom name (e.g. Decreased light availability), and below it the indicators for assessment. Shellfish act as a circuit-breaker (labelled S), interrupting the organic decomposition cycle (secondary symptoms), which are therefore (as a group) marked with an X.

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

This report aims to establish a framework for policy development by assessing the potential for

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

nitrogen removal of the most important cultured shellfish species in the EU and analysing how this can be incorporated in catchment-scale nutrient credit trading programmes. This assessment evaluates N removal by bivalves using harvest data for five key species and two complementary approaches for determining how much nitrogen is removed: (i) proximate analysis of N concentration in tissues and shell; and (ii) mathematical modelling of shellfish farms. In both cases, the results are upscaled to the total production in Europe.

Results for the blue mussel *Mytilus edulis*, the Mediterranean mussel *Mytilus galloprovincialis*, the Pacific oyster *Magallana gigas*, the European oyster *Ostrea edulis*, and the Manila clam *Ruditapes (Venerupis) philippinarum* show that different shellfish species have different removal rates, which can also be conditioned by the distinct locations and culture practices. The species responsible for the largest amount of N removed was the Mediterranean mussel, which is also the farmed species with highest production in the EU.

Farm-scale models, such as the well-known Farm Aquaculture Resource Management (FARM) model used in this work, provide an assessment of the drawdown of phytoplankton through bivalve filtration—this is represented by chlorophyll, a key indicator in EU Water Framework Directive (WFD, 60/2000/EC). This is important because the N load per se is not a symptom of eutrophication, but a causative factor; the key issues of eutrophication are the consequences of nutrient enrichment and cultivated bivalve filter-feeders act directly on these, thereby short-circuiting the process of eutrophication (Fig. 1) before more extreme effects (indirect effects or secondary symptoms) are felt.

Although farm-scale models are an important tool to assess nutrient removal, and are relatively cheap and straightforward to use, they do not provide a system-scale evaluation of changes to key WFD ecosystem indicators such as waterbody-wide reductions in chlorophyll.

A different class of system-scale mathematical models can be used for that purpose, and when combined with catchment models, can be used to examine detailed nutrient management options at the ecosystem scale. To illustrate the use of such models, which are both more complex and more costly than the local-scale approach, case studies from Europe and the United States are provided.

The objective of these studies is to illustrate to policy-makers how a more detailed catchment-scale approach can contribute to (a) increased environmental sustainability; (b) provision of an extra source of income for shellfish farmers while helping to fix land farmers, i.e. agriculture and livestock operators, in particular geographic areas; and (c) compliance with water-related legislation such as the WFD.

In financial terms, the benefits of incorporating cultivated shellfish into a catchment-scale nutrient management scheme are significant (Table 1). The remediation costs of different measures such as stormwater control or agricultural best practices are taken from Rose et al (2015) and Ferreira & Bricker (2019).

Table 1. Financial benefits of an EU-wide nutrient credit trading framework to include shellfish farmers.

Nitrogen removal		Minimum (analytical)	Maximum (FARM)
Nitrogen removed by shellfish (tonnes per year)		4900	13425
Population-Equivalents (PEQ @ 3.3 kg N per ind.)		1484848	4068076
Value of eco-intensification	Remediation cost (€ kg-1 N)	Credit valuation (Millions of €)	Credit valuation (Millions of €)
Stormwater control measures	3388	16601	45483
Approved agricultural BMP	435	2132	5840
Wastewater treatment upgrades	7047	34530	94604
Average credit valuation (millions of €)		17754	48642

If we consider only the lower estimate of nitrogen removal by shellfish, the average overall value of nutrient removal totals almost eighteen billion €. The maximum estimates are considerably higher, and it is recognised that the potential valuation is associated to nutrient loading that is for the most

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

part diffuse and is therefore challenging to reduce in many rural areas in Europe without severe social consequences.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

2. Introduction

The Green Aquaculture Intensification in Europe (GAIN) project (www.unive.it/gainh2020_eu) is funded by the European Union's Horizon 2020 (H2020) framework. The project is run by a consortium of 20 partners from 11 different countries, including Canada and China. The global scope of the consortium is enhanced by the involvement of the US National Oceanic and Atmospheric Administration (NOAA), which together with the University of Dalhousie (Canada) form the Galway[2] component of GAIN.

The primary aim of GAIN is to support the ecological intensification of aquaculture in the European Union (EU) and the European Economic Area (EEA), with the dual objectives of increasing production and competitiveness of the industry, while ensuring sustainability and compliance with EU regulations on food safety and environment.

The objectives of this work are to:

1. Determine how much nitrogen (N) is removed from different European waters through bioextraction by farmed shellfish;
2. Partition overall N removal to the national scale based on European bivalve production;
3. Assess the relevance of removal by farmed shellfish in terms of the total N loading to European seas;
4. Evaluate the potential role of European shellfish growers in watershed-level nutrient credit trading programs for eutrophication control.

3. State of the art

3.1. Nutrient management and eutrophication

Nutrient discharge to coastal waters is a major driver in the development of eutrophication symptoms (Bricker et al., 2003; Borja et al., 2008; Diaz & Rosenberg, 2008). The conceptual relationship for these primary and secondary symptoms, also called direct and indirect effects (OSPAR, 2010), is illustrated in Fig. 1.

In this work, we use the definition of eutrophication set out in the European Union (EU) Marine Strategy Framework Directive (MSFD, 2008/56/EC), since our emphasis is on the trading potential of nutrient abatement services. The MSFD defines eutrophication (Ferreira et al., 2011) as '*a process driven by enrichment of water by nutrients, especially compounds of nitrogen and/or phosphorus, leading to: increased growth, primary production and biomass of algae; changes in the balance of organisms, and water quality degradation. The consequences of eutrophication are undesirable if they appreciably degrade ecosystem health and/or the sustainable provision of goods and services.*'

Despite reductions of loads globally, eutrophication is still an issue in many waterbodies as illustrated in Fig. 2, which shows the > 400 waterbodies (total area > 245 000 km^2) that are known to have hypoxic areas, one of the main indicators of nutrient related impairment (Diaz & Rosenberg, 2008). In the US, 65% of estuaries and coastal waterbodies have moderate to high eutrophication impacts according to a 2007 assessment (Bricker et al., 2007, 2008).

Assessments of EU waterbodies also show that eutrophication is still a concern. For example, despite

[2] The Galway Statement is a tripartite agreement between the EU, US, and Canada for promotion of ocean science. See https://research.noaa.gov/article/ArtMID/587/ArticleID/2360/Transatlantic-Research-Cooperation-to-Treasure-and-Protect-the-Atlantic-Ocean-Event

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

reductions of land-based loads and signs of improvement (e.g. decreases in chlorophyll concentrations), 97% of the Baltic is still considered eutrophic, with 12% considered to be in the worst eutrophication status category (HELCOM, 2018). Application of the OSPAR Common Procedure indicated that eutrophication is still a problem in 7% of the North-East Atlantic, mainly in coastal areas.

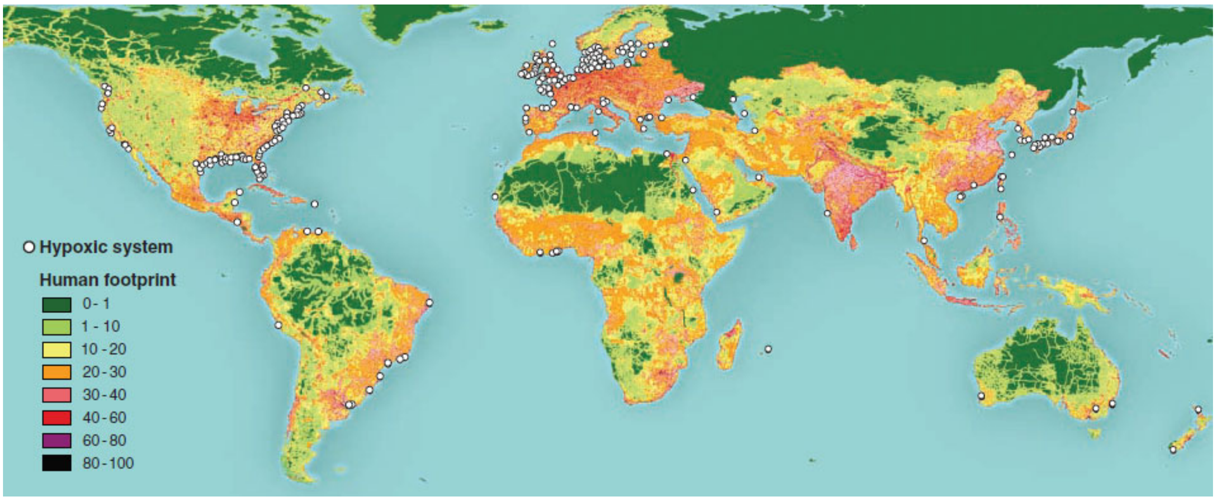


Fig. 2. Global distribution of >400 systems with reported eutrophication-associated hypoxic zones. The distribution matches the global human footprint (human footprint is expressed as a percent; from: Diaz & Rosenberg, 2008).

The Greater North Sea had the largest problem area (approximately 98 000 km^2), extending along the coast from Belgium to Danish and Swedish waters. Small problem areas (5 to 400 km^2) were found along the coast of France, Norway and the United Kingdom (OSPAR, 2017). In the Celtic Seas, many small inshore and coastal areas were classified as problem areas (approximately 500 km^2) and in the Bay of Biscay two problem areas (approximately 800 km^2) were indicated. The OSPAR assessment of Norwegian and Barents Seas showed that 96% of the region was classified as non-problem area, but inner coastal areas of the Skagerrak were classified as problem areas (OSPAR, 2017).

Finally, the first comprehensive Marine Strategy Framework Directive (MSFD) assessment of eutrophication status within the English Channel, southern bight of the North Sea, the Celtic Seas, the Bay of Biscay and the Western Mediterranean Sea indicated that, despite efforts in recent decades to reduce nutrient inputs, the pressure on coastal marine ecosystems was still high (Lefebvre & Devreker, 2020).

Nutrients may be taken up directly in inorganic extractive aquaculture, e.g. for seaweeds such as Nori (*Porphyra yezoensis*), and other plants such as water spinach (*Ipomoea aquatica*), but also indirectly through organic extraction. 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; Ferreira & Bricker, 2016).

Land-originated nitrogen and phosphorus discharge is a major causative factor of freshwater and marine coastal eutrophication worldwide. The excessive load of nutrients triggers overgrowth of algae which afterwards decompose, leading to a decrease of the oxygen concentration in water. Low oxygen levels are a threat to biodiversity and can cause fish and shellfish mortality. Additionally, nutrient-related algal blooms can cause high turbidity leading to loss or degradation of seagrass beds and may lead to other issues such as toxic algal blooms that interfere with tourism and water-based activities, interdiction of shellfish aquaculture, and fish kills (Xiao, et al., 2007).

The estimation of non-point sources of nutrients, originating in fertilizer loss from crop fields and pastureland, livestock waste products, and rural domestic wastewater, is often based on model

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

simulations which can be divided into mechanistic models and empirical tools. Mechanistic models allow the simulation of hydrological processes, groundwater flows, nutrient transport and in-stream transformations, but often require significant parameterization and input/validation data. For example, the continuous-time hydrological model SWAT enables users to address gradual environmental problems but requires both maps and tabular data as inputs (Marteleira, et al., 2018).

On the other hand, simpler empirical approaches such as the export coefficient model (ECM) help to quantify the diffuse loads of N and P based on the concept that the total non-source pollution load can be derived from the sum of loads from the various land uses (Cheng, et al., 2018). Point sources from human settlements connected to sewers, wastewater treatment plants, industries, and paved areas, are typically estimated based on population data and the emission rates per capita if the measured loads are not accessible (Malagó, et al., 2019).

In this work, an extensive literature review was completed to collate information on nitrogen and phosphorus inputs to European marine waters. This updates the loading data detailed in Ferreira & Bricker (2016, 2019).

3.2. Bivalve production in Europe

In order to structure robust policy measures such as a nutrient credit trading program, solid datasets are required to estimate, model, and analyse potential policy impacts. For this work, farmed shellfish production data was needed for scaling of impacts to country and EU level.

Public policies that aim to promote aquaculture production can also benefit from reliable datasets. Consumer demand for more and better seafood traceability is also a driver for improved seafood datasets, with innovative methodologies for production, trade, and consumption estimates (Pieniak et al., 2013; Lopes & Ferreira, 2020; Krause et al, 2020).

Large aquatic production datasets, such as those from Eurostat or FAO, are aggregated as commodities due to the long time periods and high number of countries involved. These normalized datasets, despite potential discrepancies in live or net weight volumes and loss of taxonomic detail (Girard et al., 1998; Pauly et al., 2002; FAO, 2007; Rodgers et al., 2008; Fabinyi et al., 2016; Lopes et al., 2017), are an important starting point for establishing sound public policies.

In this work, shellfish aquaculture production data from Eurostat, extracted and transformed through the META web platform (Longline Environment Ltd, 2020) were used to identify the main species farmed in Europe. This allowed the characterization of the key producer countries, and the contribution of each to nutrient removal from aquatic ecosystems.

3.3. Simulation of bivalve ecosystem services

Filter feeding bivalves have complex interactions with their environment. The impact of these interactions depends on a number of variables, including farming practice (stocking density of the bivalve populations, etc.), physical conditions such as temperature and salinity, and water quality, particularly insofar as it determines the food supply. To model the interaction of bivalves with the ecosystem, we must first be able to simulate the physiological responses and growth at the individual level.

Over the last two decades, a substantial research effort has been placed on modelling individual growth of different shellfish species, with a particular emphasis on those that are cultivated commercially such as oysters (Ren & Ross, 2001; Gangnery et al., 2003), clams (Solidoro et al., 2000; Pastres et al., 2001), and mussels (Brigolin et al., 2009; Fuentes-Santos et al. 2019). The main objective of such models is to represent growth in individual biomass based on physical and biogeochemical determinants, although some also simulate environmental interactions.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

The most useful models for analysing response provide a deterministic simulation of physiology for the species of interest, e.g. Scope for Growth models (Brigolin et al., 2009), Dynamic Energy Budget models (Kooijman, 2010; Fuentes-Santos et al., 2019) or species-specific ecophysiological models (Scholten & Smaal, 1998, 1999); and scale individual processes to a typical farm (Cubillo et al., 2016; Føre et al., 2016). Several of these models have been incorporated into broader, population-dynamics simulations (Gangnery et al., 2004; Brigolin et al., 2009; Nobre et al., 2010; Nunes et al., 2011; Ferreira et al., 2014; Filgueira et al., 2014; Cubillo et al., 2016), where mortality is typically included as a forcing function, based on reported data from industry sources.

The simulation of aquaculture at specific test sites requires local-scale models—a wide variety of these is available, including models to simulate production (Gangnery et al., 2004; Ferreira et al., 2008; Larsen et al., 2014), environmental impacts (Sequeira et al., 2008; Brigolin et al., 2009; Fabi et al., 2009; Rose et al., 2015), and economic optimisation (Saurel et al., 2014; Cubillo et al., 2018) of bivalve aquaculture.

One of the best-known examples is the Farm Aquaculture Resource Management (FARM) model, which has been extensively used to assess nutrient mass balance at the individual and local (farm) scale. The model determines the net food removal resulting from filter-feeding, biodeposition of organic matter through faeces and pseudofaeces, and excretion of dissolved substances such as ammonia. Also evaluated are changes in eutrophication indicators, chlorophyll *a* and dissolved oxygen, resulting from the filtration by oysters, clams, or mussels during the culture period, using components of the Assessment of Estuarine Trophic Status (ASSETS) model (Bricker et al., 2003; Ferreira et al., 2010). ASSETS has been historically applied to over a hundred estuaries across the US (https://ian.umces.edu/neea/) (Bricker et al., 2007; Borja et al., 2008).

Determination of the nutrient (carbon, nitrogen and phosphorus) and suspended sediment reduction, effectiveness of farming practices, and nutrient-credit trading revenue for farmers can be calculated using the FARM model (Rose et al., 2014, 2015). The model converts estimated N removed by shellfish feeding and harvest to human population-equivalents (PEQ) and calculates the potential value of the ecosystem service represented, providing a 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 & Bricker, 2016; Bricker et al., 2018).

In this work we further developed the FARM model to apply an individual-based approach at the population scale, a methodology that has many advantages over the conventional population-dynamics equations.

4. Methodology

4.1. Nutrient loading estimates

The nutrient loading of some European waters was previously reviewed by Ferreira and Bricker (2016) and Ferreira and Bricker (2019). In those reviews, the source-apportionment of nutrient loads, a key element for policy decisions, was not fully available in many regions.

In the present work, we have expanded and improved the European dataset used by Ferreira and Bricker (2019) to include more detailed and up-to-date loading data for different European regional seas (Table 3 and Table 4). It was not possible to find loading data for the different regions from the same year and we took this into consideration when analysing the results. Where possible, we have discriminated nutrient sources by combining data from various authors, including: (i) Skarbøvik et al. (2013) for the Norwegian and Barents Seas; (ii) HELCOM (2015, 2018) for the Baltic Sea; (iii) OSPAR (2017) for the Greater North Sea, Celtic Seas and Bay of Biscay; and (iv) Bouraoui et al. (2011) for the

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

Greater North Sea, Black Sea, and Mediterranean Sea; and (v) Malagó et al. (2019) for the North Africa discharges to the Mediterranean Sea.

We used the area covered by each of the regional European seas to calculate the normalized nutrient loading per unit of surface area (kg km^{-2} y^{-1}), which is shown in Table 3 and Table A.1. Where possible, we have discriminated the nutrient sources into: (i) point-sources, including industry, sewage, and finfish aquaculture effluents; (ii) diffuse sources, which mostly includes 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.

Due to the lack of available data, the nutrient discharges of finfish aquaculture to the Mediterranean Sea have been modelled for the two most important species (sea bass and gilthead seabream) by means of the FARM model. We used the typical farming practice for sea bass and gilthead seabream in Turkey and Spain, respectively, obtained from the Climate change and European aquatic RESources (CERES) EU project (https://ceresproject.eu/). We then upscaled the modelled finfish nutrient discharges considering the reported production of sea bass and gilthead seabream in the Mediterranean Sea in 2019, which is 253 and 210 kt y^{-1}, respectively, according to FAO and FEAP estimates (Apromar, 2019).

4.2. Bivalve production

Bivalve production data was sourced from the META (Longline Environment Ltd, 2020) and Eurostat (Eurostat, 2020) websites. The volumes and values for the selected years and species were compared to determine production trends and variations in production between 2016 and 2018. A short economic analysis was also conducted for these bivalve products. This was done for each species and the main producing countries were identified.

4.3. Nutrient removal by shellfish

Bivalve shellfish remove nutrients from the water through filtration of particles. A 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, 1997; Barillé et al., 1997; Cerco & Noel, 2005;). In addition, different shellfish species exhibit significant variation in filtration rates, with the rates generally increasing with species size (Cranford & Grant, 1990; Prins et al., 1991; Powell et al., 1992; Smaal & Zurburg, 1997). For large species, such as the Eastern or American oyster, *Crassostrea virginica*, filtration rates have been estimated at 163 litres per gram of oyster tissue per day (NOAA Fisheries Office of Habitat Conservation, undated-a). The associated removal of nutrients was estimated in two ways: (i) by using the per organism content (in tissue and shell) of nutrients; and (ii) by a mass balance of intake and loss (i.e. faeces, pseudofaeces, excretion, mortality) by the modelled population.

4.3.1. Proximate analysis estimate 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. Combustion elemental analysis is the process of determining the elemental composition of a pre-weighed sample of a material by flash combustion and separation of the constituent elements in gaseous form. Prior to analysis, shellfish flesh and shells were dried to a constant weight at 60°C and ground to a fine powder.

Flesh samples were ground manually in a clean mortar and pestle; whole shells were ground using an industrial ball-mill. Subsamples of each were then weighed and encapsulated in pre-weighed tin cups for analysis. Sample analysis was then conducted on the elemental analyser setup for CN analysis. Briefly, each pre-weighed sample was held in the auto-sampler. The sample drops into an oxygen rich

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

steel combustion chamber, held at 950°C. Flash combustion occurs in a 10 sec stream of ultra-pure oxygen (flow rate 240 ml/min) and aided by the exothermic oxidation of the tin cup, which raises the combustion temperature to over 1000°C.

The gaseous carbon and nitrogen oxides are then carried by a constant stream of helium carrier gas (140 ml/min) through a column of cobaltous oxide catalyst to remove sulphur and halogenated compounds. The purified gas then passes through a column of reduced copper, held at 680°C, which converts the nitrous oxide gases present to elemental Nitrogen and retains excess oxygen. Any water vapour present is then removed from the gas mixture by passage through a Magnesium perchlorate packed adsorption filter, after which the Carbon and Nitrogen are separated on a chromatographic column and passed to a thermal conductivity detector.

The electrical signals are processed using Thermo Eager Xperience software to calculate the carbon and nitrogen percentages for each sample. The instrument was calibrated following manufacturer's instructions, using the K-factor calibration methodology with Acetanilide as an analytical standard. Aspartic acid (Thermo Fisher part number: 338 400 22) and a commercial soil reference material (Thermo Fisher part number 334 400 25) were used as quality control (QC) material to check instrument calibration, with two replicates of each QC material analysed within each batch of 30 samples.

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.

4.3.2. IBM 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,b; Sequeira et al., 2008; Nobre et al., 2010b; 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., 2009, 2010a; Ferreira et al., 2012, 2014; 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 (*Mytilus galloprovincialis*) and blue mussels (*M. edulis*); Ferreira et al. (2010 and 2011) for Pacific oyster (*Magallana gigas*); Bricker et al. (2015) for Eastern oyster (*C. virginica*); and Saurel et al. (2014) for Manila clam (*Ruditapes philippinarum*). Thus, only key processes of interest to this work will be described here.

Recent improvements to these individual shellfish growth models have led to greater accuracy in simulating both shellfish production and their environmental effects. This improved modelling approach has been successfully tested for mussels and oysters in various European systems in the framework of the CERES EU H2020 project (Ferreira et al., 2021; Cubillo et al., 2021), and in several Northern Ireland Loughs (Carlingford Lough, Belfast Lough, Dundrum Bay, and Lough Foyle).

The main disadvantage of previous individual modelling approaches is the lack of elasticity when dealing with food availability at limit conditions, i.e. when food supply is very low (oligotrophic) or very high (highly eutrophic). As a consequence, these models require modification, particularly for conditions of excessive food supply, which has typically been done either by adding or re-parameterizing equations. The main aim of the new AquaShell growth models is to improve model

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

generalization enabling them to simulate shellfish growth across a broad range of environments, from oligotrophic to highly eutrophic waters. To achieve this, the individual model now includes two modifications to simulate the feeding behaviour:

- Limitation of the bivalve's food intake by establishing a maximum ingestion rate based on the gut capacity/gut volume (related to the animal size by means of an allometric relationship) and the gut passage time (the time needed for food particles to pass through the gut), following Scholten and Smaal (1998a, 1999a);
- Limitation of the ingestion rate by production of pseudofaeces, which is estimated by means of a Michaelis-Menten equation based on the concentration of suspended particles in seawater, following Scholten and Smaal (1998b, 1999b).

These bioenergetic models have been validated for the shellfish species studied here and were used to predict shellfish growth, reproductive effort, and overall mass balance for the whole culture cycle at the individual level (Ferreira et al., 2010; Ferreira et al., 2012; Ferreira et al., 2014).

The improved parameterization of the environmental effects of shellfish culture on the environment, in particular the net removal of organics (phytoplankton and detritus), and a robust nutrient (C and N) mass balance, provide a better assessment of the provisioning and regulatory ecosystem services of shellfish. The individual shellfish models are then incorporated into the FARM population model to determine both production and environmental effects (removal of particulate suspended particles, particulate organic waste, excretion of dissolved nitrogen, and oxygen consumption) at the local-scale.

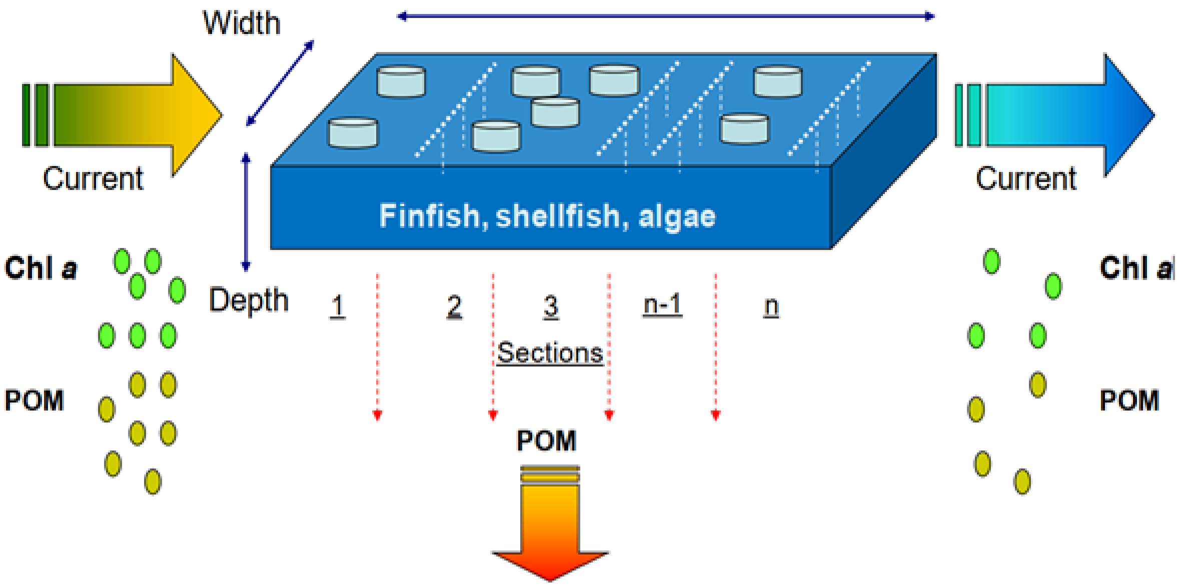


Fig. 3. Conceptual diagram of the FARM model (Ferreira et al., 2007).

The FARM model simulates processes at the local (farm) scale, considering advective water flow and transport of particulate and dissolved material as water moves across the shellfish culture structures (Fig. 3). FARM simulates the production over one culture cycle and calculates gross phytoplankton and detrital carbon and nitrogen removal by shellfish due to feeding (Fig. 3). The net removal is obtained by discounting losses due to pseudofaeces, excretion, mortality, and spawning.

The mass balance provides a value for net removal of nitrogen from the water column by the population of shellfish, which effectively equates to a drawdown of phytoplankton, of one of the primary symptoms of eutrophication. The FARM local-scale model has been applied to a broad range of finfish and shellfish species and geographical regions covering a wide range of aquaculture practices

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

(bottom culture, trestles, cages, and suspended culture) in both intertidal and subtidal environments.

For use in an individual-based population model, each individual in the population is ‘created’ following the object-oriented paradigm as applied to ecological modelling, originally formulated by Silvert (1993) and Ferreira (1995). The properties of these objects include a number of attributes related to their growth performance and environmental interactions (e.g. food eaten, particulate organic waste etc). Individuals may die during the culture cycle and therefore mortality status is an intrinsic property of each. The physiological models referred above are all deterministic, but since our objective is to simulate the typical variance of a cultivated population, the individuals that comprise it are stochastically assigned a fitness parameter in terms of assimilation efficiency AE (±0–5% of the mean AE). This simulates genetic variation within the single cohort of organisms typically deployed at grow-out stage. Fitness is generated at runtime, so the probability of two model runs being identical is extremely small.

Estimation of minimum population

The simulation of large populations is not time-efficient, and particularly for bivalves, where millions of organisms may be cultivated in a farm, a minimum sample size for accurate simulation must be determined. This sample size can then be scaled to a greater number of organisms, allowing large populations to be simulated realistically with an acceptable model run time. There is an underlying assumption that the population is normally distributed. The minimum population size was determined using a similar approach to Brigolin et al. (2009) and was adapted to consider a population size of 10,000 individuals that can then be scaled to any greater number, as detailed in Ferreira et al. (2021).

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 2 summarises the culture practice for the model runs.

There are two options for harvest in FARM: individuals may be removed as soon as they reach the harvest size (Harvest When Ready, or HWR) or removed only at the end of the culture period. For this work we opted for the former option, which most closely represents how shellfish farmers normally operate. Stochastic events such as disease may cause additional mortality but are not included in the version of FARM applied for this work.

Table 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 Dundrum Bay Northern Ireland	Flat oyster Lough Foyle Ireland	Manila clam Sacca di Goro Italy
Farmed area (ha)	8.0	10.1	6.0	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	11-15	5	10
Stocking density (ind m^{-2})	200	312	74	100	1000
Mortality (% $cycle^{-1}$)	10	10	7	44	30
Seed weight (live weight)	0.2	1.0	30	0.2	0.12
First seeding day	120	150	150 to 300	150	90
Culture period (days)	794	550	550-730	1000	270
Harvest weight (g live weight)	>15	10-20	>70	>90	>10

4.3.3. Nitrogen removal at the European scale

N removal estimates, obtained using both methods (proximate analysis and modelling approach), were scaled up to European production level. For the proximate analysis approach, the mean values of the

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

percentage of N in live weight (wet mass) of each species (Table 7), were scaled up to European production, as (Eq. 2):

$$Nremoved_{prox} = N_{shellfish} \times Production \qquad (2)$$

Where Nremoved$_{proxA}$ (tonnes) represents the total Nitrogen removed by the estimated live weight production; Production (tonnes) is the production for each species; and N$_{shellfish}$ (%) 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 on each of the shellfish producing countries.

For the modelling approach, the scaling to EU production was estimated using data from Table 9, as below (Eq. 3):

$$Nremoved_{model} = \frac{\left(\frac{Production}{TPP}\right) \times N\ removal}{1000} \qquad (3)$$

Where Nremoved$_{Model}$ (tonnes) represents the total N removed by the estimated live weight production; Production$_{Europe}$ (tonnes) is the corresponding species production; TPP (tonnes) is the FARM modelled production for each species at the local (farm)-scale, N removal (kg y^{-1}) is the N removed by each species at the local (farm)-scale. This process was repeated to estimate the removal of N on each of the shellfish producing countries.

5. Results and discussion

5.1. Nutrient loading to European seas

Nutrient loading estimates from the literature review performed here were compared to the 2016 previous estimates (Ferreira & Bricker, 2019) in Table 3. The data set for nutrient loading to the different European seas now includes more categories to show the source of the nutrients and additional geographic regions.

The total N discharges to European seas are 19.8% higher than those calculated by previous research, while the total P discharges decreased by 38.4% (Table 3). In the last few decades Europe has implemented different measures that have resulted in a reduction in the use of mineral fertiliser, and a progressively decrease in the nutrient surpluses of agricultural origin. According to the 2018 European water's assessment of status and pressures, between 2000 and 2013 agricultural nitrogen surplus decreased by 7%, while phosphorus surplus decreased by 50% (EC, 2017; Kristensen, 2018). This fact is reflected in the reduction of P discharges suggested by our research but not in the N discharges, which according to our estimates have increased by nearly 20%.

Estimates of nutrient discharges from the literature suggest some changes. For example, the estimates of nutrient discharges to the Norwegian Sea have decreased by 23.3% (N) and 58.6% (P). This is in agreement with the Norwegian Environmental Agency report of 2012 that shows decreasing trends of nutrient loads to the Norwegian Sea from 1990 to 2010.

The contribution of finfish aquaculture to the estimated N and P discharges to the Norwegian, Barents and Greater North Sea has increased progressively from 1990 to 2014 (Skarbøvik, 2012; Norderhaug et al., 2016) and accounted for 49%, 35% and 1% of the total N loading in 2012, respectively. The contribution of finfish aquaculture to the estimated P discharges to the Norwegian and Barents Sea was 81% and 78% of the total P loading in 2012. For Norway as a whole[3], the nutrient loadings from

[3] Considering the total inputs to coastal Norwegian waters.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

fish farming contributed to about 70% of the total P inputs and about 32% of the total N inputs in 2012 (NEA, 2012).

The estimated N and P discharges to the Baltic Sea have decreased by 1.4% and 26.3%, respectively. According to HELCOM (2015), the direct source inputs of N and P to the Baltic Sea have decreased markedly by 43% and 63%, respectively, from 1995 to 2010. This report also mentions a marked decrease in sewage effluents from 2005 to 2010 that explains the P reduction.

The estimated N loading to the Barents Sea has increased by 35.9% while the estimated P loading has decreased by 17.2%. Estimates for N and P discharges to the Mediterranean Sea, including the discharges from Europe and North Africa, have decreased by 8.4% and 46.8%, respectively.

We cannot compare the estimated nutrient discharges to the Celtic Seas, North Sea, and Iberian coast because we include discharges to the whole Iberian coast and the Greater North Sea, while Ferreira and Bricker (2019) only consider the discharges to the Bay of Biscay and the North Sea, i.e. a total loading value for the Bay of Biscay, the Celtic Seas, and the North Sea combined). According to OSPAR (2017), joint efforts by OSPAR Contracting Parties have resulted in significant reductions in nutrient inputs to individual coastal waters and to OSPAR Marine Regions as a whole. The Greater North Sea had a reduction in P inputs between 1990 and 2015 (approximately 50%) and to a lesser extent a reduction in N inputs (approximately 25%). Since 2003, total N and P inputs to the Greater North Sea have remained fairly constant at 1,500 and 40 kt y^{-1}, respectively. Statistically significant reductions have been achieved for nitrogen to the Celtic Seas, and for phosphorus to all three OSPAR Regions: Greater North Sea, Celtic Seas, Bay of Biscay and Iberian coast.

Table 3. Nitrogen and phosphorus loading to European seas (10^3 tonnes y^{-1}), percent difference with previous literature review (Ferreira & Bricker, 2019), and nutrient loading per unit area (kg km^{-2} y^{-1}).

	Total N (kt y^{-1})	Total P (kt y^{-1})	Δ N (%)	Δ P (%)	N per unit area (kg km^{-2} y^{-1})	P per unit area (kg km^{-2} y^{-1})
Norwegian Sea	64.7	6.60	-23.3	-58.6	58.3	6.00
Barents Sea	11	0.80	35.9	-17.2	6.90	0.50
Baltic Sea	826	31	-1.40	-26.3	2096	78.4
Greater North Sea	1500	39			1957	50.9
Celtic Sea	275	12.5	[4]		750	34.1
Bay of Biscay and Iberian Coast	450	12			835	22.3
Black Sea	700	39			1517	84.5
Mediterranean Sea – only Europe	950	70	-52.8	-65.7	377	27.8
Mediterranean Sea - Europe and N. Africa	1845	109	-8.40	-46.8	733	43.1
Total including North Africa	*5672*	*249*			*731*	*32.2*
Total incl. N. Africa & excluding Black Sea[5]	*4972*	*210*	*19.8*	*-38.4*	*641*	*27.1*

According to the most recent estimates, the European waters that the receive highest N discharges are the Greater North Sea (1500 × 10^3 tonnes y^{-1}), Mediterranean Sea[6] (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[7] (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

[4] Ferreira and Bricker (2019) show only the discharges to the Bay of Biscay, the North Sea, and the Celtic Seas as a whole.

[5] Included to allow comparison with the loading estimates from the previous research.

[6] 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 1,845 kt y^{-1} and thus the Mediterranean Sea would be the main contributor to N loading.

[7] 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}.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

the lowest discharges: Norwegian Sea (6.6 × 10^3 tonnes y^{-1}) and Barents Sea (0.8 × 10^3 tonnes y^{-1}).

Table 4 (and Table A.1 in more detail) show the normalized nutrient loading per unit of surface area (kg km^{-2} y^{-1}) based on the area covered by each European Sea. The loading estimate considering the whole European seas (including N. African discharges) is 731 and 32.2 kg km^{-2} y^{-1} for N and P, respectively. If we consider only European discharges it reduces to 616 and 27.2 kg km^{-2} y^{-1} for N and P, respectively.

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}.

Table 4. N and P sources and loads to marine waters in 10^3 tonnes y^{-1} (% total in brackets) for European seas.

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]*	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	46 (5)	8 (11)	modelled
	Diffuse sources (riverine loads) [f]	470 (49)	22 (31.4)	2005
	Other sources [f]*	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

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

Other sources [f*]	800 (20)	30 (13)	2005
Sub-total [f]	4000	230	2005
Total (excluding N. Africa)	*4777*	*211*	
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 (2018)
[e] OSPAR (2017)
[f] EC (2011)
[g] Malagó et al. (2019)
[*] Including atmospheric deposition, scattered dwellings and biological fixation.

If North African discharges are included, the Mediterranean Sea has a similar nitrogen load (776 kg N km^{-2} y^{-1}) to the Celtic Sea (785 kg N km^{-2} y^{-1}). If we consider only European discharges to the Mediterranean, there is an estimated a total loading of 405 kt of nitrogen km^{-2} y^{-1}. The Arctic region has the lowest load when we look 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.

Sources of nutrient discharges

Nutrient loads to these coastal areas include (i) point sources, such as wastewater treatment plants (WWTPs); (ii) diffuse sources, such as agricultural and urban run-off; and (iii) atmospheric deposition from fossil fuel combustion in power plants and vehicles.

In most European seas the main load of N is diffuse (Fig. 4). 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 (Fig. 6). The majority of the P discharge to most European seas comes from point sources (Fig. 5). Only the Baltic Sea receives greater P inputs from diffuse sources (90%) than from point sources (5%), with atmospheric deposition[8] accounting for the remaining 5% of the total phosphorus input to the Baltic Sea (data from 2010 in HELCOM 2015).

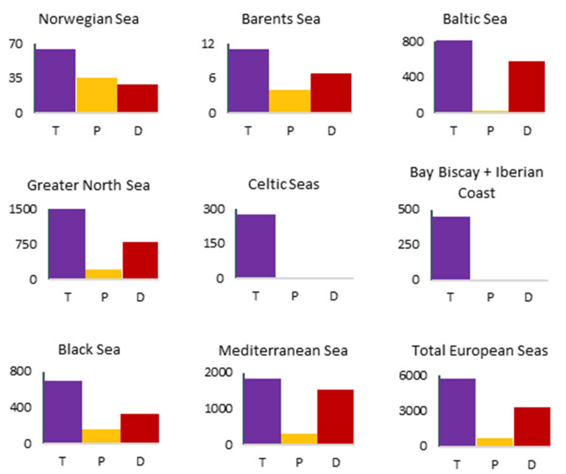


Fig. 4. Proportion of N inputs to the different European seas from point sources (P), diffuse sources (D) and total loading (T). The discharges to the Mediterranean Sea and the total European seas include loading from North Africa.

[8] 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.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

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 Europe loadings to the Mediterranean Sea, most of the P loading come from point sources (57% versus 31.4% from diffuse sources).

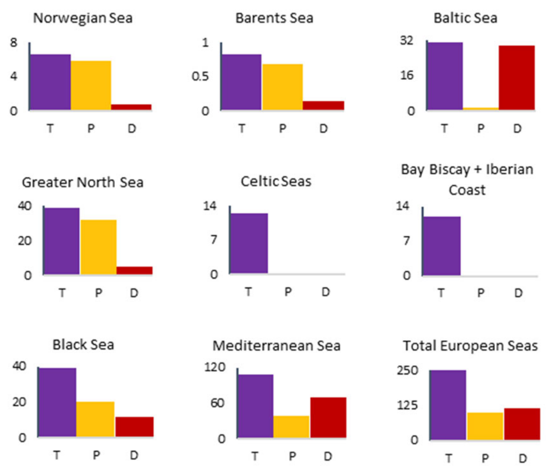


Fig. 5. Proportion of P inputs to the different European seas from point sources (P), diffuse sources (D) and total loading (T). The discharges to the Mediterranean Sea and the total European seas include loading from North Africa.

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 (Fig. 6).

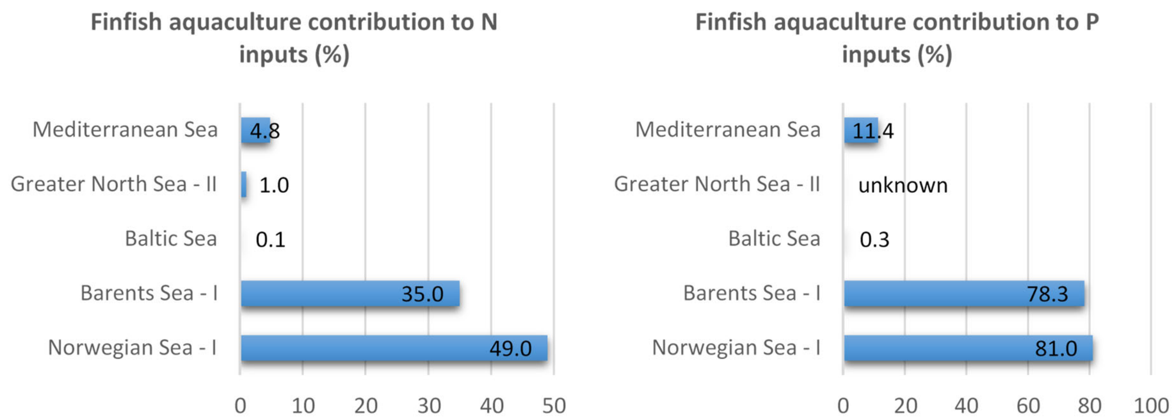


Fig. 6. Contribution of finfish aquaculture to N (left) and P (right) inputs to different European seas.

5.2. Bivalve production

Overall analysis

The five selected species in this report correspond to over 95% of shellfish farmed production in the EU, and between 40% and 45% of all farmed aquatic organisms (Eurostat, 2020; Longline Environment Ltd., 2020).

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

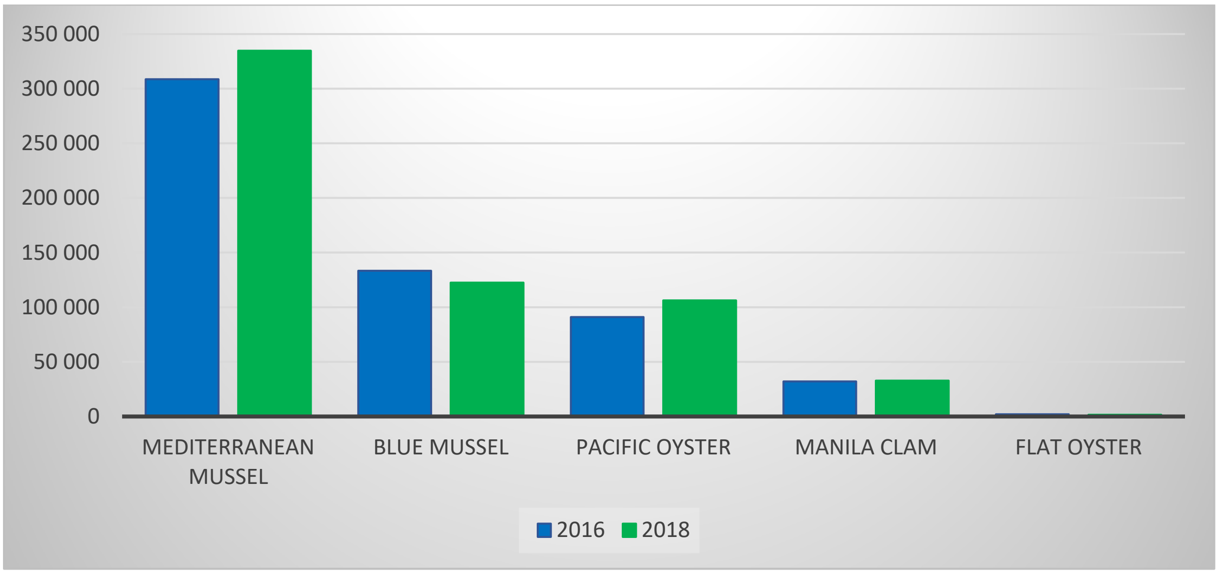


Fig. 7. Bivalve production (tonnes) in the EU in 2016 and 2018 for five species (from left to right): Mytilus galloprovincialis, Mytilus edulis, Magallana gigas, Ruditapes philippinarum and Ostrea edulis. Total production volumes of these five species increased by 5.4% from 2016 to 2018. Detailed volumes for each species can be seen in Table 5.

Overall production increased by 5.4% compared to 2016 data (Table 5), due mainly to the higher production volume of *Mytilus galloprovincialis* in Spain. These values contrast with the overall change in flat oyster production with a decrease of about 20%, which represents a loss of 421 tonnes. The largest contributor to this decrease was the drop in production in the Netherlands, which decreased by more than 50%, from 643 tonnes to 319 tonnes (Fig. 7).

The largest producing countries in the EU are Spain and France. Spanish bivalve production ranges from 39% (2016) to 42% (2018) of EU output and is dominated by Mediterranean mussel farming. France accounted for 23% and 24% of bivalve farming in the EU in 2016 and 2018 respectively. Pacific oysters are the main farmed shellfish in France, corresponding to more than 65% of farmed species production in the country in 2018. Blue mussel is the other important species, with France accounting for 31% of all the EU blue mussel harvest in 2018.

Table 5. Shellfish production (tonnes) in the EU in 2016 and 2018 for five key species (left to right): Mytilus galloprovincialis, Mytilus edulis, Magallana gigas, Ruditapes philippinarum and Ostrea edulis (Eurostat, 2020; Longline Environment Ltd, 2020).

EU Production totals (tonnes)	Mediterranean mussel	Blue mussel	Pacific oyster	Manila clam	Flat oyster	Total
2016	308 679	133 293	90 975	32 145	2 052	552 048
2018	334 685	122 552	106 254	32 743	1 631	581 179
Variation 2016 to 2018	26 007	-10 700	15 278	598	-421	22 793
Variation 2016 to 2018 (%)	8.4	-8.1	17.8	1.9	-20.5	5.4

The higher production volume of mussels does not reflect on the overall value of these animals, when compared with oysters or clams. Pacific oysters were the most valuable shellfish of the five species analysed, being valued at more than both mussel species combined (*Table 6*).

Table 6. Shellfish production value (k€) in the EU in 2016 and 2018 for five species (from left to right): Mytilus galloprovincialis, Mytilus edulis, Magallana gigas, Ruditapes philippinarum and Ostrea edulis (Eurostat, 2020; Longline Environment Ltd, 2020).

EU Production value (k€)	Mediterranean mussel	Blue mussel	Pacific oyster	Manila clam	Flat oyster	Total
2016	192 323	194 199	434 020	150 958	15 034	965 718
2018	205 957	177 180	429 471	174 538	12 536	973 818
Variation 2016 to 2018	13 634	- 17 019	- 4 549	23 580	- 2 498	15 885

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

Variation 2016 and 2018 (%)	7.1	-8.8	-1.0	15.6	-16.6	1.6

The differences in price of these products introduce some uncertainty in the final value achieved by the production. In the specific case of Pacific oysters, the 17.8% (Table 5) increase in farmed volume in 2018 did not translate in an increase of the total value, which decreased by 1% (Table 6). On the case of Manila clam, the increase in production volume (Table 5), was accompanied by the highest increase in value of the 5 species: over 15% (Table 6).

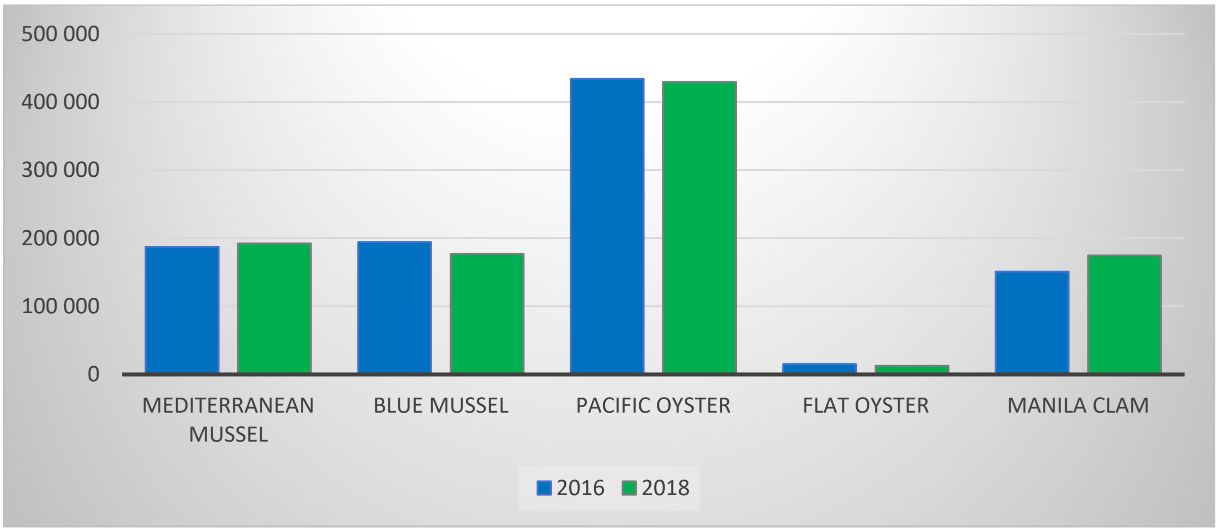


Fig. 8. Value of bivalve production (k€) in the EU in 2016 and 2018 for the five species analysed (from left to right): Mytilus galloprovincialis, Mytilus edulis, Magallana gigas, Ruditapes philippinarum and Ostrea edulis. Total value of these five species increased by 0.5% between 2016 and 2018.

The flat oyster, despite its low farmed volume when compared with the other bivalves (Fig. 8), attracted a much higher price than the other species, averaging over 7 € kg^{-1} in the period analysed (Fig. 9).

Although the higher value of oysters when compared to mussels have provided larger value outcomes, the predominance of restaurant consumption of flat and Pacific oysters has, during the COVID-19 pandemic, led to a series of issues that negatively impacted the farmed oyster market. In France, the drop in demand decreased the prices and increased the risk of pathogen spread in overstocked farms.

The farmed mussel sector was also negatively impacted with up to an 80% decrease in sales and drops in prices, leading to financial problems in several companies. In the Italian aquaculture sector, the lockdown in April 2020 led to an unprecedented 70% contraction in farmed fish and shellfish sales (EUMOFA, 2020 (1); EUMOFA, 2020 (2); EUMOFA, 2020 (3)).

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

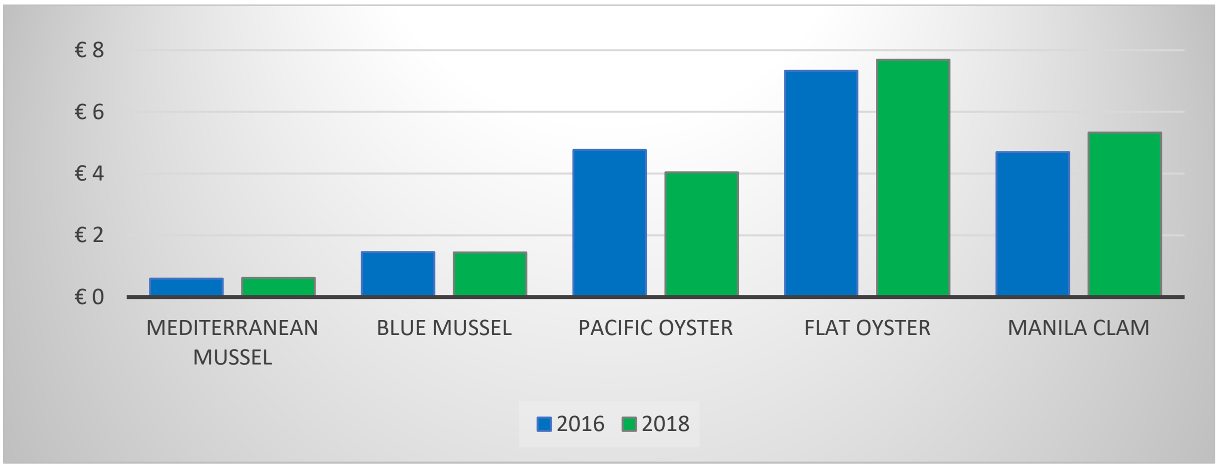


Fig. 9. Price (€ kg^{-1}) in 2016 and 2018 of the five bivalve species analysed. Oysters and clams are the highest-value species.

Detailed species analysis

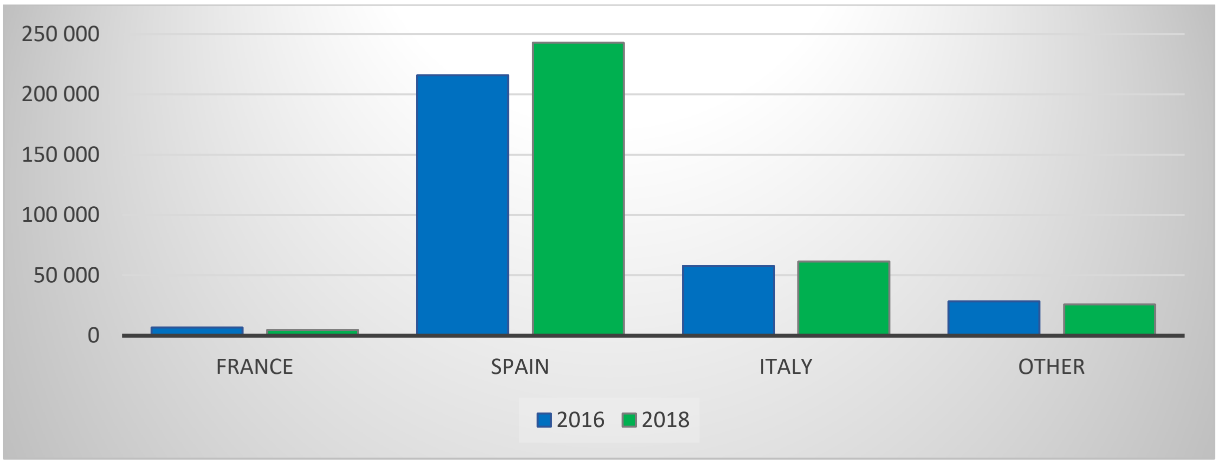


Fig. 10. EU Mediterranean mussel production (tonnes) for key producing countries in 2016 and 2018.

Mediterranean mussel was the species with the largest total tonnage increase in production (Fig. 10) in 2018 when compared to 2016, with 26 007 additional tonnes harvested (8.4% increase). Spain maintains its position in the EU with 73% of the total volume harvested, mostly in Galicia (EUMOFA, 2019). The other key players in EU farmed production of *Mytilus galloprovincialis* are Italy (18%) and Greece (6%). The production increases in both Spain and Italy have offset decreases in other less important countries for this species such as Bulgaria, Greece, and France (Table 5).

Value-wise, French mussels achieve the highest prices, reaching 1.68 € kg^{-1} in 2018, which represented a drop from the 1.91 € kg^{-1} average in 2016. Italian production prices have increased between 2016 and 2018, from 0.84 € $kg^{-1}$to 0.87 € kg^{-1}. Spanish production, by far the largest in total value and volume, has maintained the price at 0.55 € kg^{-1} since 2016.

Despite contributing over 70% of volume to the total EU production, Spanish Mediterranean mussel farming accounts for 60% in value, due to the lower unit price. Italian mussels have higher value and account for a total of 18% of EU mussel production value.

Mytilus edulis production (Fig. 11) has decreased by 8.9%, which represents a loss of 10 970 tonnes of blue mussel harvested when compared to 2016 data. France and the Netherlands accounted for 81% of the 110 062 tonnes production in 2018. This is in line with both the harvested volume and the trend from previous years, in which these two countries have had similar production volumes of around 40%

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

of total EU production each. The volume drop in 2018 was, for the most part, due to the decrease in production in the Netherlands (7 692 tonnes) and to a lesser extent in Ireland (2 267 tonnes), which is the third largest producer in the EU, with an average production of 14% of EU production since 2010 (Table 5).

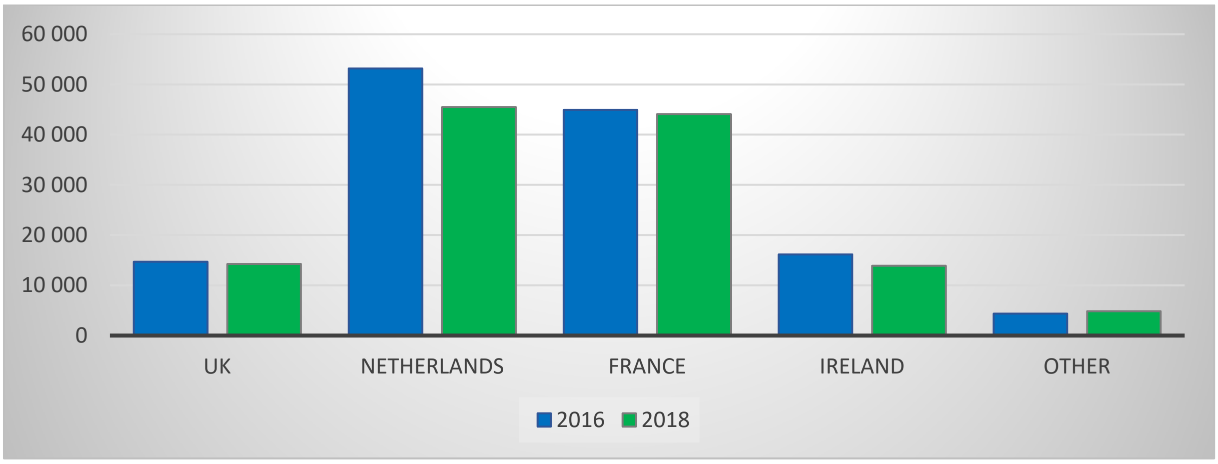


Fig. 11. Blue mussel production (tonnes) in 2016 and 2018 for the main EU producer countries.

Blue mussel total value, as opposed to volume, is higher in France than in the Netherlands. In the Netherlands, the highest price was in 2018 with 1.13€ kg^{-1}, an increase from the 0.84€ kg^{-1}in 2016. In France, *M. edulis* was sold at 2.53€ kg^{-1} in 2016 and 2.11 € kg^{-1} in 2018.

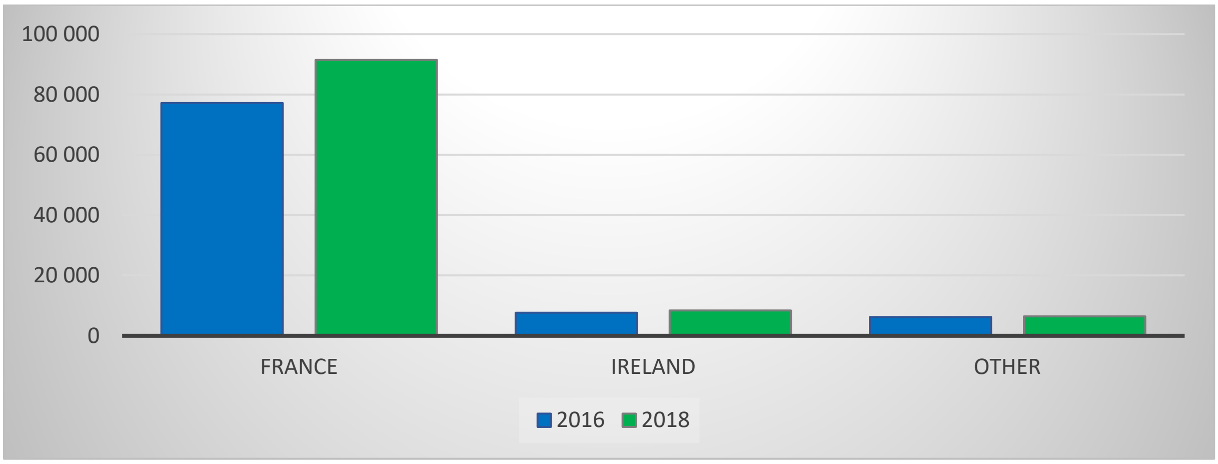


Fig. 12. Pacific oyster production (tonnes) in 2016 and 2018 for the main EU producer countries.

These differences in price have translated into a dominance of total value of French blue mussels, varying between 65% and 58% of total value in 2016 and 2018 respectively when compared with 26% and 32% of Dutch farmed blue mussels in 2016 and 2018 respectively.

The most farmed oyster in the EU, Pacific oyster, had a 17.8% increase in production, reaching 106 254 tonnes in 2018 (Fig. 12). This increase was mostly due to higher French production volumes that rose from 77 135 tonnes in 2016 to 91 455 tonnes in 2018; France remains the largest producer, accounting for 89% of EU *Magallana gigas* production. Ireland, which has seen an upward tendency since 2010, has also increased production by 9.8%, reaching 8 385 tonnes in 2018 (Table 5).

Pacific oyster farming in the EU generates the highest value, not only due to the high price point (Fig. 9), but also due to the considerable volume (Fig. 7). Production is highly concentrated in France, as is

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

the value generated accounting for 90% (2016) and 88% (2018) of total value in the EU.

The price of Pacific oyster in the largest producer in 2016 reached 5 € $kg^{-1}$and decreased to 4.12 € kg^{-1}in 2018. In the total of the EU these values differ slightly from 4.77 € kg^{-1}in 2016 and 4.04 € kg^{-1}in 2018.

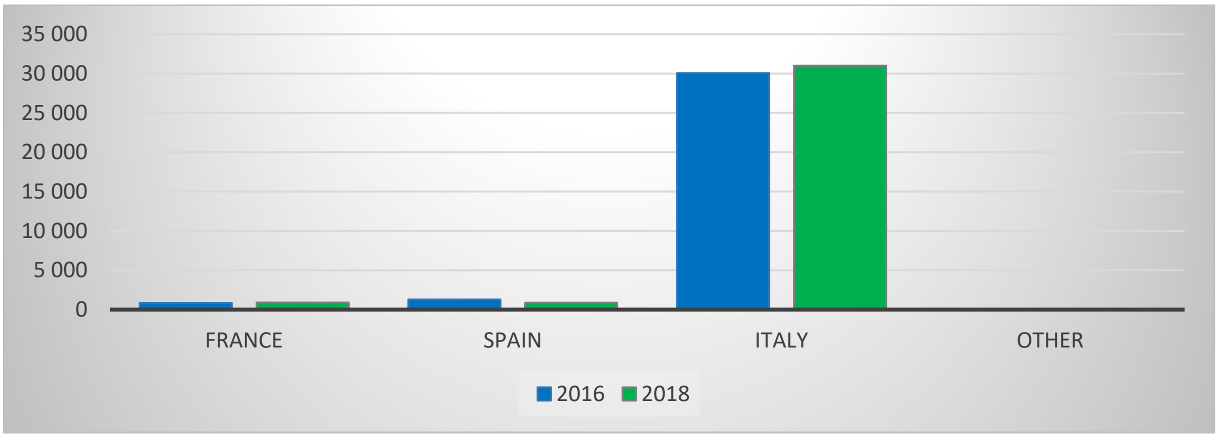


Fig. 13. Manila clam production (tonnes) in 2016 and 2018 for the main EU producer countries.

Manila clam production (Fig. 13) increased between 2016 and 2018 by 598 tonnes, a 1.9% increase. The country with the highest contribution to *Ruditapes philippinarum* production is Italy, with more than 90% of total (30 990 tonnes in 2018). A slight increase in production was also registered in Italy, which balanced the production decrease in Spanish production.

The average price of Manila clam in 2016 was 4.70 € kg^{-1}, rising to 5.23 € kg^{-1} in 2018. These values are slightly above the 4.41 € kg^{-1} (2016) and 5.16 € kg^{-1} (2018) in Italy, which is the largest value producer, responsible for over 88%.

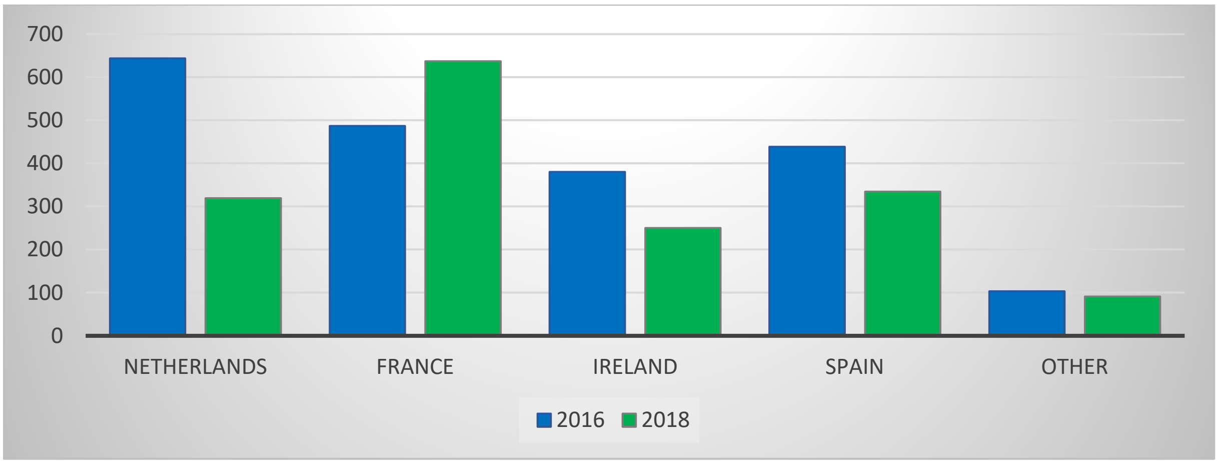


Fig. 14. Flat oyster production (tonnes) in 2016 and 2018 for the main EU producer countries.

Ostrea edulis output (Fig. 14) represents less than 3% of total oyster production in the EU, nonetheless this is an important species from an economic standpoint due to its higher market value: In France, the largest EU market for oysters, *Magallana gigas* was sold at 4 115 € t^{-1}, and *Ostrea edulis* was sold at 8 213 € t^{-1} in 2018 (Longline Environment Ltd, 2020). Since 2016, production dropped by 20.5% to 1 631 tonnes. Production is concentrated in four countries, which comprise over 90% of the farmed flat oyster in the EU. France, which is the largest producer in the EU, was the only country of the four that increased production (by 31%), with Ireland, Spain and the Netherlands losing 15%, 23%, and 50% respectively (Table 5).

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

Flat oysters are the shellfish species with greatest distribution in terms of production in the EU, as well as having the highest unit value. In terms of price, of the four main countries, the highest registered price (in 2016) was in France with 8.88 € kg^{-1}, dropping to 8.21 € kg^{-1} in 2018. In 2018, the unit price for Dutch farmed Pacific oysters overtook French oysters, reaching 10.69 € kg^{-1}, a rise of 25% from 8 € kg^{-1} in 2016.

Spanish and Irish oyster prices are lower when compared with those farmed in the abovementioned countries, reaching 5.16 € kg^{-1} in 2016 and 5.29 € kg^{-1} in 2018 in Spain and 5.94 € kg^{-1} and 5.52 € kg^{-1} in Ireland in 2016 and 2018 respectively.

5.3. Proximate analysis of bivalves

The proximate analysis of bivalves was completed with samples from the main farmed shellfish species in the EU: Manila clams from Venice lagoon, collected by GAIN partner UNIVE; Mediterranean mussels from Sagres, in the south of Portugal, sourced by GAIN partner SGM; blue mussels from Belfast Lough, Pacific oysters from Dundrum Bay and flat oysters from Lough Foyle, all from Northern Ireland and collected by GAIN partner AFBI (Table 7).

Table 7. Percentage carbon and nitrogen contents in the shell and flesh of different shellfish species (% of live weight, in fresh mass).

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

Table 7 shows a summary of the proximate composition analysis for five aquaculture shellfish species: two species of mussels (*M. edulis* and *M. galloprovincialis*), two species of oysters (*M. gigas* and *O. edulis*) and the Manila clams (*V. philippinarum*).

The mean nitrogen and carbon content is similar for all the species. The N content is very low and ranges from 0.2% in flat oysters and 0.3% in Manila clams to 1.13% in Mediterranean mussels, while the C content ranges from 6% in Manila clams to 17.4% in Mediterranean mussels.

Overall, mussel species contain a greater proportion of N and C than oysters and clams.

5.4. Role of shellfish in nutrient removal

Shellfish filtration can enhance water clarity, allowing sufficient light penetration to support maintenance and expansion of seagrass habitat (Newell & 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 & 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.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

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 a high proportion of N into their tissues. For example, (dry) oyster tissue has a mean nitrogen content ranging from 5.64% to 9.72%, and (dry) shell content is 0.19-0.3% (Kellogg et al., 2014 and citations therein; Sebastino et al., 2015; Grizzle et al., 2016). When the shellfish are harvested, the N is removed from the system, 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 large shell (Kellogg et al., 2013; STAC 2013a). In contrast, when species with lighter shells, such as blue mussels, are harvested, less N is removed (Newell & Koch, 2004).

By interacting with water column phytoplankton dynamics and denitrification, shellfish are likely to reduce nutrients that stimulate eutrophication 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 & Rosenberg, 2008).

5.4.1. Scaling proximate analysis data to European level

Proximate analysis estimates, coupled with the detailed shellfish production volumes for each country allowed a baseline calculation of the nitrogen removed in 2018 by the main shellfish producing countries (Table 8).

Table 8. N removed for each species in the key EU producer countries, estimated using the proximate analysis approach. Volumes are presented in tonnes for 2018.

Nitrogen removed in EU in 2018 (tonnes) – proximate analysis						
	Blue mussel	Mediterranean mussel	Pacific oyster	Flat oyster	Manila clam	Total
UK	125					125
Netherlands	398			0.94		399
France	386	47.3	339	1.87	2.83	777
Ireland	122		31.1	0.73		153
Spain		2 426		0.98	2.77	2 430
Italy		614			99.1	713
Others	42	258	23.8	0.27		325
Total EU-28 production	1 073	3 346	393	4.79	104.7	4 922

The bulk of nitrogen removal is due to mussel production, 20% from blue mussel and 64% from Mediterranean mussel. Nitrogen removal from blue mussels is relatively well distributed across all producing countries, with the Netherlands and France accounting for a high proportion: 37% and 36% respectively. Spain removes 46%, due to its high production volume of Mediterranean mussels. Italy, the main clam producer, has the third highest removal, 15%, due mainly to the >61 000 tonnes of Mediterranean mussel harvest.

Of the countries analysed, France has the second highest value of N removal, due mainly to its production being distributed across the five species, although more concentrated in blue mussels and Pacific oysters.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

5.4.2. Scaling modelling outputs to European level

We used a combination of physiological and local-scale production models to assess the capacity of some of the main cultivated shellfish species in Europe to reduce the primary symptoms of eutrophication. These models enable us to simulate the volume of water cleared by each species in the typical (test) 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.

Table 9 shows the simulation results of the FARM IBM local-scale model which have been annualised and normalised to the size of each 'typical farm' to allow for comparison. The FARM model simulates biomass production at the end of the culture cycle (harvestable and undersized) and economic indicators such as the return on investment or APP (average Physical Product).

FARM estimates the nitrogen removed as part of the harvestable biomass as well as the nitrogen removed by the whole population (harvestable and undersized individuals) after a typical cultivation cycle. The model also translates this N removal to PEQ and calculates the additional benefit that the farmer would obtain if the ecosystem services provided by farmed shellfish were accounted for.

Table 9. Potential nitrogen removal based on FARM model simulations through a culture cycle, using typical culture practice (based on typical farms) for the key bivalve shellfish species cultivated in Europe.

Species	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	105	65	2144	2.05	0.20	648
Med. mussel	Portugal	20	9.6	488	2.46	0.25	148
Pacific oyster	Northern Ireland	23	2.0	571	2.53	1.77	173
Blue mussel	Netherlands	21	71	269	2.06	0.31	82
Flat oyster	Northern Ireland	18	251	451	2.46	2.22	451
	Average	37	80	785	2.31	0.95	300

Table 9 shows highly variable harvest results (e.g. yields from 18 to 105 t ha^{-1} y^{-1}), depending on the species, which reflect the very different cultivation strategies (e.g. stocking densities) implemented at each farm (see Table 2 for details on the farming practice). The flat oyster culture in Northern Ireland has the lowest yields, caused by high mortality rates, while Manila clam has the highest yields and the highest stocking densities (1000 ind. m^{-2}).

We find the lowest and highest APP (or return on investment, i.e. the ratio of biomass produced to biomass seeded) in Northern Ireland. The lowest is found in Pacific oyster culture in Dundrum, due to the fact that the culture starts with 30 g individuals. The highest is found in flat oysters, due to the combination of small seed and high harvest size (120 g).

Apart from the APP, the rest of variables in Table 9 show homogeneous results for the culture of Pacific oyster in Northern Ireland, Mediterranean mussels in Portugal, and blue mussels in the Netherlands, with average annual yields of 37.4 tonnes ha^{-1} y^{-1}, mean N removal of 785 kg ha^{-1} y^{-1}, and average N in harvestable biomass of 2.31%, which leads to an ecosystem service of 300 PEQ ha^{-1} y^{-1} on average.

The N removal estimated by the FARM model (Table 9) was scaled up to EU production to provide a whole sector overview of the European N removal capacity, estimated by means of the modelling approach.

As for the proximate analysis approach, the FARM model also places Mediterranean mussels as the main shellfish species for nitrogen removal. Blue mussels are the second in production volume and the second largest N removal crop, and Pacific oysters the third-highest N removing organisms.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

Table 10. N removed for each species in each of the main EU countries, estimated using the FARM model. Volumes are presented in tonnes for 2018.

Nitrogen removed in EU in 2018 (tonnes) – FARM model						
	Blue mussel	Mediterranean mussel	Pacific oyster	Flat oyster	Manila clam	Total
UK	272					272
Netherlands	868			6		875
France	842	113	2 080	13	16.9	3 065
Ireland	265		191	5		461
Spain		5 809		7	16.6	5 833
Italy		1 470			592	2 062
Other	93	618	146	2		858
Total production	2 340	8 010	2 417	32	625	13 425

The 60% N removed by Mediterranean mussel is mostly concentrated in Spain, while the 18% removed by Pacific oysters are mainly from France. Blue mussel farming corresponds to 17% of N removal. The two main countries are the Netherlands and France, with over 400 tonnes each.

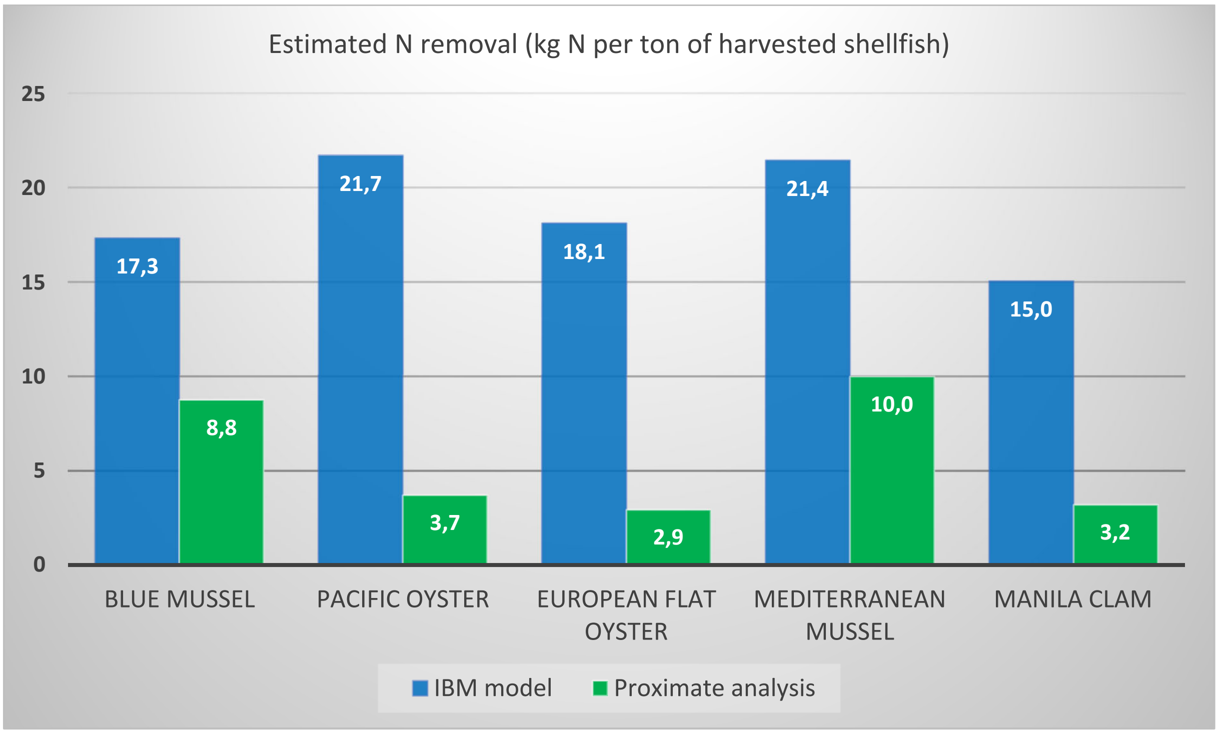


Fig. 15. Comparison between the FARM model and the proximate analysis estimates for N removal (in kg) for each of the 5 species, per ton of shellfish harvested.

When comparing the capacity for N removal of each species using both approaches, the IBM model leads to higher values for all species (Fig. 15). The reason is the analytical approach estimates the N removed by the average harvestable-sized individuals, while the modelling approach considers the N removal by the whole population, that is the harvestable-sized and undersized individuals, plus the individuals that didn't survive and thus aren't harvested, but contributed to N removal.

The proximate analysis approach shows that Mediterranean and blue mussels have the highest capacity for N removal, reaching over 8 tonnes of N removed per 1 000 tonnes of mussel harvested.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

Both oysters and the Manila clams range from 2.9 to 3.7 tonnes of N removed for each 1 000 tonnes harvested (Fig. 15).

The modelling approach shows that Pacific oyster and Med mussel culture has the greatest removal of N. Flat oyster culture and blue mussel culture follow with similar values. By contrast with the proximate analysis approach, the modelling approach estimates more homogeneous N removal by oysters, mussels, and clams.

5.5. Mass balance of bivalve production and nutrient loading

The N removed by each shellfish species (Table 8 and Table 10), was grouped by country and by European regional sea. The data were grouped with N loading for each European sea from Table 3 and Table 4. The percentage of nitrogen from the total N loadings removed by shellfish production was estimated for each area, using both approaches (Table 11).

Depending on the approach considered, shellfish production in Europe was able to remove between 0.12% and 0.31% of total nitrogen inputs in 2018. This value can vary considerably, depending on which area is being analysed. For the North Sea, the removal is below 0.11% of inputs, and in the Bay of Biscay and Iberian coast, it can go as high as 1.3%, when the modelling approach is considered. This higher value for the above area is due to the high-volume production of Mediterranean mussels in Spain.

Other values can be estimated from this data, such as the nitrogen from finfish aquaculture removed by shellfish farming, which would vary between 5.1% according to the proximate analysis and 13.93% with the FARM model.

Table 11. Nitrogen loading in European seas (adapted from Table 3 and Table 4), and the nitrogen removed on each area as a percentage of total loads. Nitrogen removal percentages calculated based on the country of production, according to the FARM and proximate analysis approaches.

	Nitrogen loading	Nitrogen removal (%)	
	Tonnes	FARM	Proximate analysis
Norwegian Sea	64 600	-	-
Barents Sea	11 010	-	-
Baltic Sea[9]	601 900	-	-
Greater North Sea[9]	1 000 000	0.11	0.05
Celtic Seas	275 000	0.17	0.06
Bay of Biscay and Iberian Coast	450 000	1.30	0.54
Black Sea	700 000	-	-
Mediterranean Sea - only Europe	950 000	0.54	0.16
Total	4 052 610	0.31	0.12

6. Case studies

The aim of the three case studies presented in this section is to complement the local-scale approach used for determining N removal by shellfish by means of a system-scale analysis. This is too costly and demanding to be applied as an assessment method on e.g. a national scale, but the case studies illustrate the type of information that may be obtained with this kind of in-depth study.

[9] Atmospheric deposition values were not used to estimate N removal, because only for the Baltic and Greater North Sea those values were available. N removal estimates and loads do not account for N atmospheric deposition.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

6.1. Dundrum Bay

General waterbody characteristics

Dundrum Bay (Fig. 16) is located on the southeast coast of County Down, Northern Ireland. Inner Dundrum Bay is a small, sheltered bay connected to the more exposed south facing Outer Dundrum Bay by an inlet channel. Inner Dundrum Bay has two parts which extend on a SSW-NNE axis. The larger northern part (Inner Dundrum North) is approximately 3 km long and up to 1 km wide. The smaller southern part (Inner Dundrum South) extends for approximately 2 km to the SW and is up to 500 m wide, both drain to the Outer Bay via the Dundrum Outer channel.

Inner Dundrum Bay is approximately 6 km in length (from SW to NE) and is 1.4km at its widest point. The Inner Bay is intertidal except for the inlet channel: it is a mesotidal embayment which is flushed through a single tidal inlet ~200 m wide and ~1 km long. This embayment exchanges most of its volume every tidal cycle due to the tidal oscillation at the mouth of the inlet linking the bay with the neighbouring shelf. Due to its very small storage volume, the mixing and advection processes in Outer Dundrum Bay will determine the fraction of Inner Bay water which gets reincorporated in the next flood tide. In addition to the tide, wind and coastal processes, the inner bay is influenced by 4 rivers that contribute to the modulation of salinity and to the residual flow across the inlet. These are the Ardilea and the Blackstaff Rivers, which flow into the NE part of the bay, and the Carrigs and Moneycarragh Rivers, which flow into the SW part of the Inner bay, west of the Downshire Bridge.

Outer Dundrum Bay is approximately 14.5 km wide from St. John's Point to Newcastle and extends approximately 6 km from the coast. The outer bay is relatively shallow, and is framed by sandy beaches, interrupted by only a few reefs and rocky foreshore.

Dundrum Bay (both Inner and Outer) is designated as a Special Area of Conservation (SAC)—Murlough SAC (Site Code: UK0016612) and a small part is a Nature Reserve, due to the presence of dune systems, sandbanks, mud and sand flats, and saltmarshes and due to the presence of the common seal (*Phoca vitulina*). Many species of waders, duck, and geese visit the estuary particularly during the spring and autumn migration periods.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

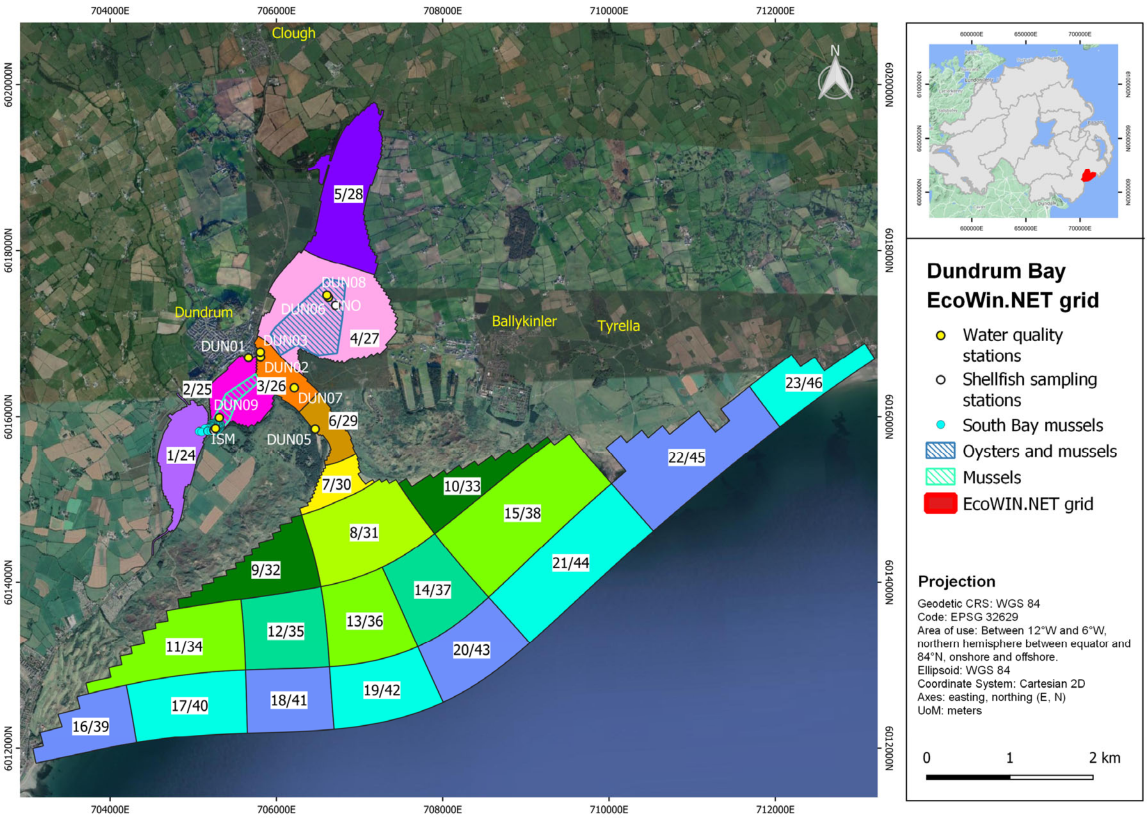


Fig. 16. Dundrum Bay, showing sampling stations, shellfish growing areas, and EcoWin.NET boxes (labelled 1-23 for Upper boxes and 24-46 for lower model boxes).

Catchment and loading

The catchment area is approximately 150 km^2 and contains the four main rivers draining into the bay.

Land use is approximately 80% agricultural, predominantly sheep followed by cattle, pigs and poultry. There are 7 urban wastewater networks in the catchment. Up to 65% of the overall catchment's major freshwater sources may impact the Inner South area and 35% potentially affect the Inner North (Grant et al., 2017).

The village of Dundrum is on the north-west shoreline and Dundrum WWTW is a Membrane Bio-Reactor type treatment plant which discharges directly into the Inner South Bay. within 70 m of the DB2 aquaculture area. This area is also impacted by WWTW discharges on the Carrigs River. There are two small Rotating Biological Contactors (RBC) WWTW (Maghera and Leitrim) and a larger Activated Sludge Plant (Annsborough ASP) that treats human waste from the Annsborough and Castlewellan areas. The Moneycarragh River also impacts on the Inner South area and has one small RBC plant (Drumaroad) which is a significant distance upstream from the bay. Inner North is impacted by two small RBC Plants, Clough and Loughinisland draining to the Ardilea and Blackstaff Rivers respectively.

There are a number of possible direct sources of untreated sewage into Inner Dundrum Bay from spills to the rivers, spills from the WWTW and CSOs. Run-off from agricultural land will also enter the bay directly or from the rivers.

Inner Dundrum Bay is located within a predominantly rural catchment. The Carrigs River catchment has high numbers of sheep in the upper reaches combined with significant numbers of cattle (Grant et al., 2017). In the lower reaches, the numbers of sheep and cattle (ruminants) significantly decrease as urbanisation increases in the lower reaches of the river. There is a high number of poultry in the lower Carrigs and its tributaries. The Spate River adjacent to the Carrigs also has large numbers of poultry

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

and cattle within its catchment. The Moneycarragh River catchment is dominated by large numbers of sheep in the upper reaches and cattle on the lower ground. The lower reaches of the Moneycarragh River is more suitable for arable farming and cattle grazing which is reinforced by the large numbers of sheep in the upperlands (NIW 2015, 2017).

The Ardilea and Blackstaff river catchments are heavily dominated by large numbers of sheep, cattle and poultry. The Ardilea and Blackstaff Rivers both have significant areas of arable farming. Daily *E. coli* loadings vary between species, sheep rank first (18.1 x 10^9), followed by pigs (8.9 x 10^9), cattle (5.4 x 10^9), gulls (2 x 10^9), humans (1.9 x 10^9) and lastly poultry (0.24 x 10^9) (Jones & White, 1984).

Land use within the catchment area is dominated by pastures, followed by natural grassland, moors and heathland, and complex cultivation patterns. Along the coastal zone, land use varies between pastures, discontinuous urban fabric, agricultural activities and recreational areas such as sport and leisure facilities and beaches.

Bivalve shellfish aquaculture

The designated shellfish waters in Dundrum cover an area of approximately 2.12km^2 and are located in the inner bay (DOENI, 2009a). Aquaculture in the Inner Dundrum Bay area occurs in two licensed areas, where there is a history of shellfish cultivation since 1980. The licensed shellfish area covers 51.6 ha in the north and 11.8 ha in the south (Fig. 16) but shellfish culture is in the proximity of the main charted river channels: approx. 12 ha in the north side (not in the river channel) and 6 ha in the south side (in the river channel).

The inner north area is licensed for both Pacific oysters and blue mussels and the inner south area for blue mussels. Inside the north area, trestle culture is allowed within the licensed site whilst within the south area only bottom culture is permitted and is restricted to the charted channels. The area covered by oyster trestles in the north is currently ~ 6 ha and the area covered by mussel beds in the south is ~12 ha.

At present, Inner Dundrum South mussels are not harvested due to water quality issues but there is a large stock of mussels present. The Inner Dundrum North site is mostly dedicated to Pacific oyster cultivation on intertidal trestles, although blue mussels are cultivated on intertidal trestles in a small area closer to the quays.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

Eutrophication modelling of the bay

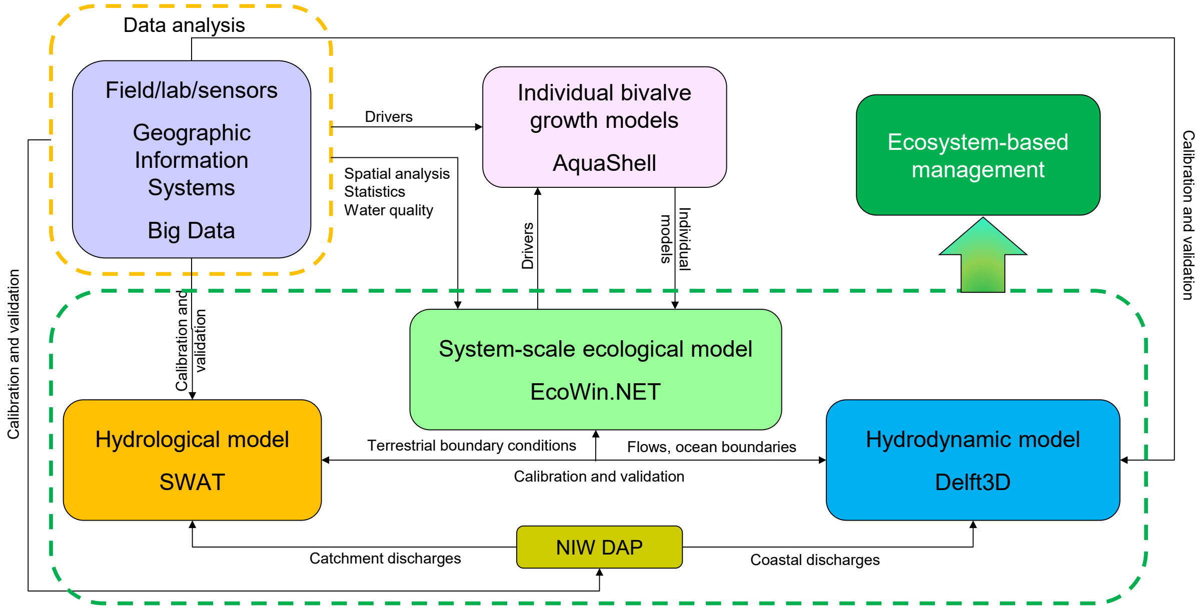


Fig. 17. SUCCESS ecological modelling framework applied to Dundrum Bay.

The SUCCESS (System for Understanding Carrying Capacity, Ecological, and Social Sustainability) modelling framework was applied to Dundrum Bay. The key elements of this framework are shown in Fig. 17. Each model has a number of uses *per se* and addresses different management challenges, and the linkages among models allow the whole set to be leveraged for improved decision-support.

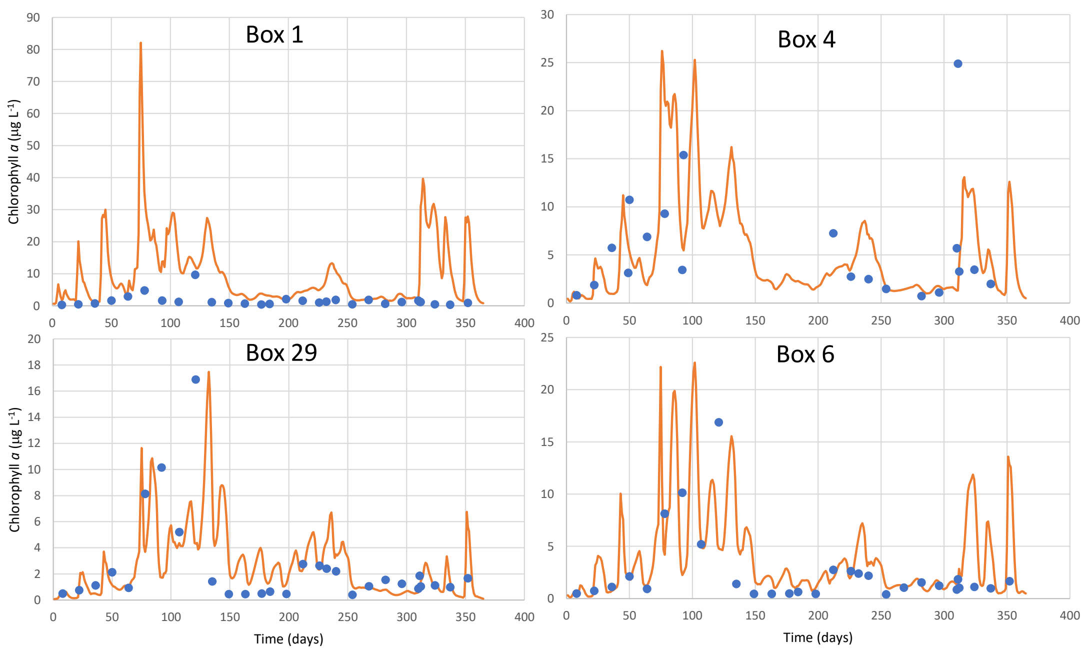


Fig. 18. Chlorophyll concentration in four model boxes (inner Dundrum Bay and inlet channel).

The well-tested EcoWin.NET (EWN) ecological model was used within the SUCCESS framework to analyse the role of bivalves in top-down control of eutrophication symptoms. Both chlorophyll and POM are key variables in EWN, since they constitute the food that bivalves need in order to grow. Fig.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

18 and Fig. 19 show the validation curves for these two state variables.

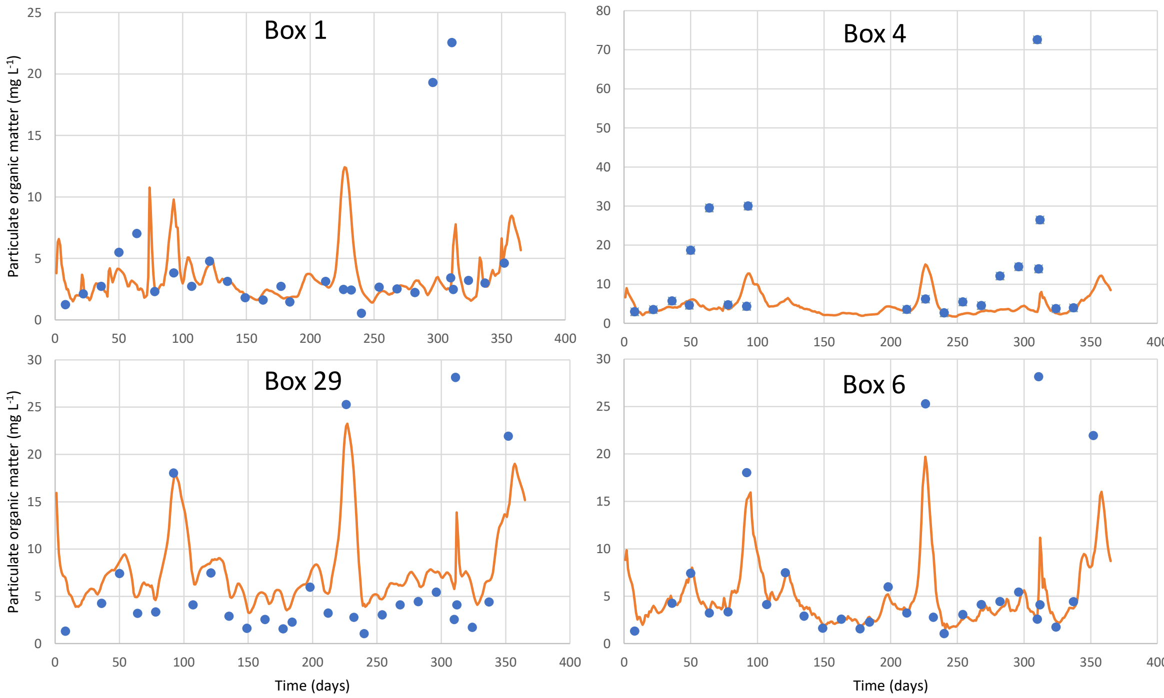


Fig. 19. Particulate organic matter (POM) concentration in four model boxes (inner Dundrum Bay and inlet channel).

The match between observed and simulated data is acceptable both with respect to numeric range and temporal variability. Chlorophyll in boxes 4, 6, and 29 is well matched, but this does not mean the winter peak shown in the model for Box 1 should be considered an error—it is associated with a peak discharge at day 70, although EWN appears to overestimate the chlorophyll concentration.

POM in boxes 1, 6, and 29 is well simulated, certainly well enough to drive the shellfish growth models, but in Box 4 there are peaks in the data which the model does not reproduce, possibly because sampling in this shallow intertidal area took place at low water and was in essence measuring water quality within the river channel.

Table 12 shows the effect of shellfish cultivation on the percentile 90 of chlorophyll in three EWN boxes: Box 2 and Box 4 are the surface boxes where shellfish are grown, and Box 6 is in the centre of the inlet channel.

Table 12. Top-down control of eutrophication in inner Dundrum Bay and the inlet channel.

	Box 2	Box 4	Box 6
Standard model with shellfish P_{90} (µg chl L^{-1})	17.8	11.8	10.2
No top-down control by shellfish P_{90} (µg chl L^{-1})	18.6	12.2	10.7
Difference (%)	4.5	3.8	5.1

The increase in the typical chlorophyll maximum ranges from 3.8 to 5.1%. For the surface boxes where cultivation takes place, the southern part of the inner bay shows a higher difference, but more interestingly, the inlet channel shows the highest difference—these results suggest that the effect of top-down control occurs in a broader area of the bay, since the benthic filter-feeders are removing food from the water passing through the cultivation sites. Changes to cultivation practice will thus be reflected in a more general way on bay-scale eutrophication.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

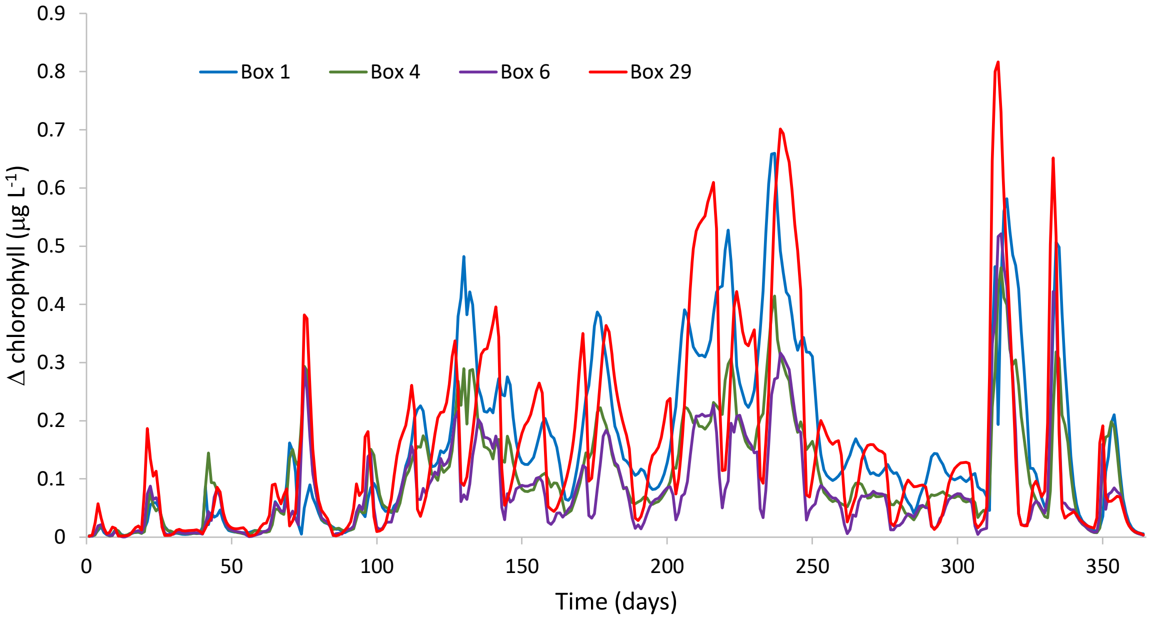


Fig. 20. Top-down control of eutrophication (phytoplankton with and without bivalves) in four model boxes (inner Dundrum Bay and inlet channel).

Fig. 20 shows the change in chlorophyll concentration for the inner part of Dundrum Bay. Box 1 in the inner bay and box 29, the lower depth layer of the inlet channel, show the greatest differences when shellfish are switched off in the EWN model. Although the difference never exceeds 1 $\mu g\ L^{-1}$, partly due to the short water residence time, it reaches a 23% reduction in box 1 and a 30% reduction in box 29. The latter is particularly interesting because it shows a pronounced vertical gradient—box 6 only has a maximum reduction of 15%, and neither box 6 nor box 29 are shellfish cultivation areas.

The bay-scale assessment of differences in chlorophyll concentration—one of the primary symptoms of eutrophication—due to top-down control by farmed shellfish, can only be done by means of an ecosystem model. Models such as FARM can determine food depletion within a farm, but they cannot predict what the resulting effect will be at the full bay scale. Since phytoplankton abundance, biomass, and composition is one of the biological quality elements in the WFD, and since both abundance and biomass are usually represented by chlorophyll as a proxy, system-scale models are a valuable management tool for considering different scenarios for eutrophication management.

6.2. Chesapeake Bay

General waterbody characteristics

Chesapeake Bay is a large estuary (11,600 km^2) located in the mid-Atlantic region of the US. The Bay is shared by Maryland and Virginia which receives point and non-point discharges from the 166 $x10^3\ km^2$ watershed that includes parts of six states—Delaware, Maryland, New York, Pennsylvania, Virginia and West Virginia—and the District of Columbia. The Chesapeake Bay's land-to-water ratio is 14:1: the largest of any coastal water body in the world. The Bay is almost 300 km long, with a relatively deep (20 to 30 m) and central channel confined by a sill at its seaward end. About 50% of freshwater enters Chesapeake Bay through the Susquehanna River at the head of the bay with an additional 20% accounted for by freshwater inflow from the Potomac River in Maryland and the James River in Virginia. Other major tributary river systems draining into the bay are the Patapsco, Patuxent, Chester, Choptank, and Nanticoke Rivers in Maryland and the Rappahannock and York Rivers in Virginia. Average depth of Chesapeake Bay is 7.3 m with depths in tributary rivers ranging from 3.0 – 5.1 m. Tidal range varies from 0.3 m at the head of the Bay to about 1.0 m near the mouth of the bay with a

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

range of 0.50-0.82 m in the tributaries (Bricker et al., 2007, 2008).

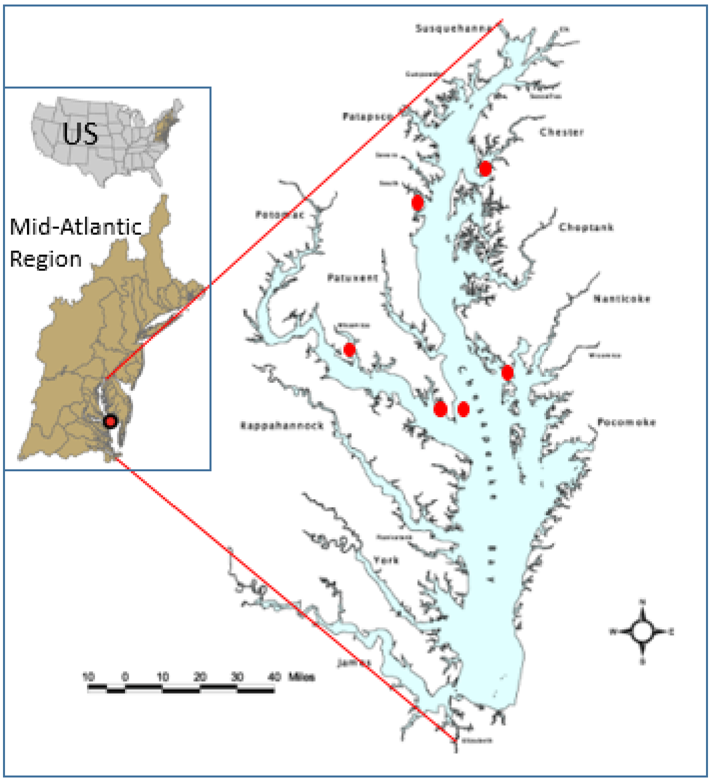


Fig. 1. Location of Chesapeake Bay and major tributaries within the Mid-Atlantic region of the United States (inset). MD oyster farms used for estimation of oyster N content also shown (red dots).

Parts of the Bay, especially the central channel, are salinity stratified which creates a relatively long resident time (90 – 180 d) that can lead to hypoxic conditions, which has historically been a problem in the Bay (Cooper & Brush, 1993; Bricker et al., 2007). The largest land use is forested (56%) with agriculture making up 33%, and urban areas making up 9% of the watershed area. The total N input to the bay in 2017 was 115 x 10^6 kg (G.Shenk, NCBO Chesapeake Bay Modeling Group, pers. comm.) or ~10 tons N km^{-2} yr^{-1}.

The Chesapeake Bay watershed is home to >18 million people with >50% of the population living along or near the Bay's shores. Historically, the Bay has been very productive providing habitat for thousands of species and supporting diverse human activities and economies. However, population growth, development, and changes in land and water use within the Bay's watershed have resulted in pollution, changes in sedimentation patterns, and habitat degradation. Once one of the world producers of oysters (Churchill, 1920), populations have been reduced to ~1% of historic populations and fish kills, harmful algal bloom events, and hypoxic water conditions are now regular occurrences (Kemp et al., 2005). Despite considerable efforts by Federal, State, and local agencies for >30 years, Chesapeake Bay water quality has not substantially improved system-wide and the "fishable and swimmable" goals of the Clean Water Act are yet to be attained. A 2009 White House Executive Order (13508) recognized the primary pollutants causing ecosystem degradation were nitrogen, phosphorus, and sediment and mandated reductions by improving existing water pollution control strategies and through development of innovative nutrient reduction tools and practices.

Aquaculture industry characteristics

Chesapeake Bay oyster growers cultivate the Eastern oyster, *Crassostrea virginica* (Gmelin, 1791) using two different farming practices, bottom spat-on-shell with no gear or container (cage) culture. Growers using bottom practices spread spat-on-shell oysters, produced using remote setting techniques (Meritt & Webster, 2013), on the bottom. The typical observed mortality is 75%–80% during the 36- month growing cycle (Congrove, 2008; Abbe et al., 2010; Kingsley-Smith et al., 2009; Parker et al., 2020). After planting, there are periodic checks to monitor growth but no regular handling of oysters during the growing cycle. Oysters grown on bottom generally take 36 months to reach market size (7.6 cm, 3 inches; Parker et al., 2020). Container (cage) operations (bottom or floating) typically use juvenile oysters that are raised in a nursery to prevent losses from predation, at about 15 mm they are placed into mesh bags inside cages. As the oysters grow, they are split into multiple bags or containers with larger mesh sizes to prevent overcrowding and allow for better water flow (i.e., food delivery) through the cages. Container-grown oysters are handled several times during the 18- to 24-mo growing cycle,

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

with an overall reported mortality of 30%–50%. Container grown oysters generally take 18 to 24 months to reach 3 inches (7.6 cm; Parker et al., 2020).

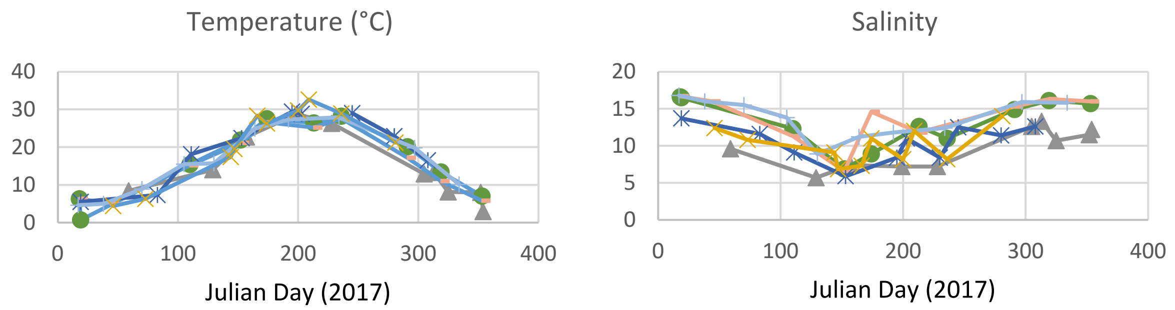
In MD Chesapeake Bay, lease sizes range from 0.10 to 100 acres per lease. Total lease area was 5,320 acres in 2017 and the industry is growing quickly (van Senten et al., 2019). In 2017, the mean lease size for bottom-culture operations was 18.5 acres, and mean lease size for container operations was 4.69 acres for all farms in MD Chesapeake Bay (MD DNR, 2019). In VA Chesapeake Bay, where the aquaculture industry is well developed, mean lease size is 24.7 acres, ranging from 0.10 to 1,927 acres per lease and in 2017 there were 139,000 acres leased (Beckensteiner et al., 2020).

The spat-on-shell practice makes up 60% of harvest in MD and 16% of harvest in VA (Table 13), while container culture makes up 40% of harvested oysters in MD and 84% in VA. In both states, triploid oysters are used mainly in container cultivation practices and diploid oysters are used in bottom spat-on-shell culture. Diploid oysters can reproduce, triploid oysters are used for aquaculture because they cannot reproduce and thus grow faster. In this study, we used six oyster farms in MD Chesapeake Bay to determine average N removal per oyster. The 6 farms, 4 container culture and 2 bottom culture, represent reasonably well the 70%:30% ratio of harvest from container and bottom culture farms (Fig. 1).

Table 13. Harvest of cultivated oysters, bottom spat on shell with no gear and container, for 2017. Percentages of total harvest represented by container and bottom harvest in each state and percent of total Chesapeake Bay harvest represented by Virginia also shown.

State	Harvest total	Cage/container harvest	Bottom spat-on-shell harvest	Source
Maryland	20,368,150	8,046,775	12,321,375	van Senten et al. 2019; MDFMP, 2019; MDACC, 2020
Virginia	46,150,000	38,700,000	7,450,000	Hudson, 2019
MD + VA	66,148,275	46,756,700 (71%)	19,769,200 (29%)	
	MD (30%) VA (70%)	MD (17%) VA (83%)	MD (62%) VA (38%)	

The annual average salinity and other water quality variables at the study farms that are used for model inputs are shown in Table 14 and Fig. 2.



The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

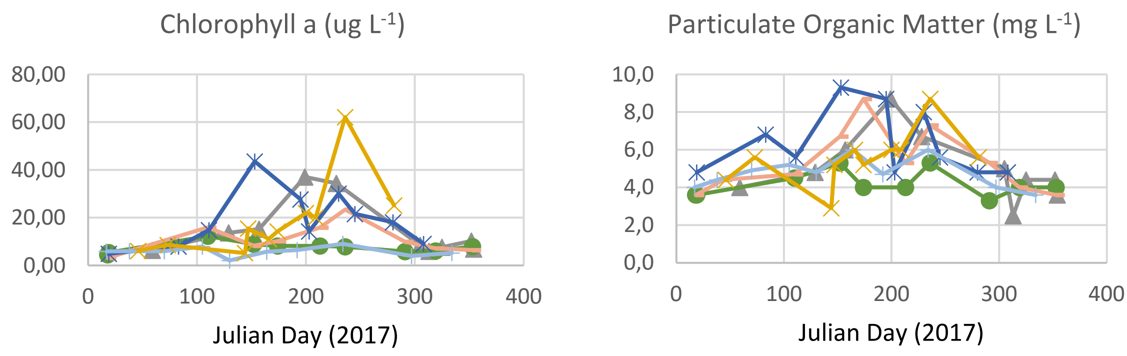


Fig. 2. Plots of monthly measures (2017) of FARM IBM data inputs at 6 MD oyster farms.

Water quality data and reported harvest for 2017 were used because it was a typical rainfall year while 2018 was an extremely wet year that caused reduced oyster growth and mortality throughout the Bay due to reduced salinities.

All of the MD sites (Fig. 1) have high-level impacts from nutrient loads including moderate to high levels of chlorophyll (CHL; range among all sites of highest annual CHL concentrations was 10–32 $\mu g\ L^{-1}$; Parker & Bricker, 2020). These concentrations suggest that additional nutrient management measures are needed, given the association of seagrass die off, changes in the diversity of phytoplankton community, and low bottom water dissolved oxygen (DO) concentrations observed at 15 μg CHL L^{-1} and higher (US EPA, 2001; Bricker et al., 2003).

Table 14. Average annual values for FARM IBM inputs for the 6 Maryland oyster farm locations.

Site	Temperature °C	Salinity -	Chlorophyll a (ug L^{-1})	POM (mg L^{-1})	TPM (mg L^{-1})
Chester River	14.5	9.7	14.6	5.0	16.9
Calvert Bay	16.7	13.1	7.5	4.2	13.0
Chesapeake Bay	16.1	13.7	10.7	5.4	19.4
Honga River	16.0	13.8	5.8	4.8	17.2
Wicomico River	20.0	10.5	19.2	6.3	15.3
West River	21.5	10.0	19.0	5.6	12.7

Method for bioextractive nutrient estimation

This study (Parker & Bricker, 2020) used a combination of models and field data to evaluate impacts of N inputs from the watershed on water quality, and the potential improvement of nutrient related water quality degradation via oyster related nutrient removal. The objective was to determine the feasibility of using oysters as an additional N management measure and to determine the potential value of the water cleaning ecosystem service that the oysters provide. The main model used was the local-scale Farm Aquaculture Resource Management (FARM) oyster production model, calibrated to Chesapeake Bay (Ferreira et al., 2007; Cubillo et al., 2018). The model simulates farm-scale oyster production and oyster associated N removal via assimilation into tissue and shell. Here, the focus is on N because it is, globally, most often the limiting nutrient in estuarine waters and has been the focus of coastal nutrient management (Malone et al., 1996; Howarth & Marino, 2006).

An average N removal per oyster (0.9 g per oyster) was determined from model results for the 6 farms and used in combination with the total oysters harvested (Table 13) to estimate potential N removal for all of Chesapeake Bay. The estimated N removal by oyster aquaculture in Chesapeake Bay was 60 tons per year, equivalent to 0.052% of total N inputs to the Bay. While this seems small, it is optimistic to note that areal removal rates compare favorably to reported N removal effectiveness of approved

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

agricultural best management practices (Rose et al., 2015). Additionally, this estimate does not include denitrification losses, which can be significant, and thus can be considered a conservative estimate (Kellogg et al., 2013; Bricker et al., 2020; Rose et al., 2021).

Some of these data were used in the Chesapeake Bay Program Oyster Best Management Practice Panel evaluation of the potential use of cultivated oysters as a nutrient management measure. Harvested oyster tissue was approved as a nutrient best management practice in December, 2016 (Cornwell et al., 2016). In this program, harvest of one million diploid 3 inch oysters gives a credit of 198 lbs (90 kg) of N removed and the amount increases to 287 lbs (130 kg) N credited if the oysters are triploid (Table 15). The next step was to determine the value of the bioextractive nitrogen removal service to evaluate potential economic benefits of this oyster aquaculture ecosystem service.

An avoided or replacement costs economic analysis was used to estimate the value of the removed N. This method assumes that the N removal service provided by oyster aquaculture would have to be replaced by an alternative management measure (King & Mazzotta, 2000).

Table 15. Chesapeake Bay Program Approved Harvested Oyster Tissue for Nutrient BMP Credits (Adapted from Cornwell et al., 2016).

BMP	Length (cm)	Kg N reduced/million oysters harvested	Kg P reduced/million oysters harvested
Diploid Oyster Aquaculture	5.7	50	10
	7.6	90	10
	10.2	150	20
	12.7	220	20
	≥ 14	310	30
Triploid Oyster Aquaculture	5.7	60	10
	7.6	130	10
	10.2	260	30
	12.7	440	50
	≥ 14	670	70

The costs to remove one kg of N by alternate reduction strategies were used to provide estimates of the economic value of the removed N, giving a range of potential values. Typically, the least cost option within the watershed (e.g. Dvarskas et al., 2021) will be used to determine the value of removed N. The alternative management costs specific to Chesapeake Bay that were used to assign value to oyster related N removal were; improvements to wastewater treatment plants at three different treatment levels (8, 5, and 3 mg L^{-1}, \$35 - \$104 kg^{-1} $year^{-1}$; Jones et al., 2010), implementation of agricultural BMPs (\$7 - \$1034 kg^{-1} $year^{-1}$; Stephenson et al., 2010; Jones et al., 2010), and urban non-point controls (\$66 - \$4873 kg^{-1} $year^{-1}$; Stephenson et al., 2010; Jones et al., 2010). We were able to compare these results to estimated value based on the 2017 value of N credits from the VA Nutrient Credit Exchange program (\$8.33 per credit; VNCEA, 2017; VA DEQ, 2017). The range of potential economic values for the oyster aquaculture related N removal is \$0.132 to \$292 million per year compared to the VCNEA based value of \$0.500 million per year. This illustrates that the avoided costs or 'costs saved' are typically greater than the costs paid to growers, which are determined by the market.

Conclusions and Policy implications

- Bioextractive removal of N through oyster does not contribute significantly to the removal of nitrogen compared to total inputs. The 60 tons N per year removed by 0.052% of the total annual N input.
- Based on a cost avoided method of estimation, the value of the N removed through oyster cultivation and harvest Chesapeake Bay ranges from \$0.132 to \$292 million depending on the alternative management measure used to estimate costs. The value based on the VCNEA is \$0.50 million per year. If shellfish growers were included in a nutrient trading program, the N removal

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

by their shellfish could be credited toward fulfilment of required N reductions and the growers could be compensated for the removal service provided by their shellfish. Oyster related removal (tissue only) can be credited within the MD Chesapeake Bay Nutrient Credit Trading Program and two growers have already received payment for the nutrient removal service provided by their oysters (Cornwell et al., 2016; Wheeler, 2020).

- Although not valued, there are additional ecosystem services provided by shellfish such as denitrification N losses (Kellogg et al., 2013; Bricker et al., 2018, 2020), and increased water transparency that can lead to valuable improvements to water quality such as increased seagrass and habitat, and improved bottom water dissolved oxygen. Thus, the economic value of all shellfish ecosystem services is likely much greater.
- Recent research has shown that costs and N removal efficiency of shellfish cultivation compare favourably with approved Best Management Practices (BMPs; Stephenson et al., 2010; Rose et al., 2015). The removal rates for Chesapeake Bay are consistent with removal rates of already approved agricultural BMP practice (Rose et al., 2015).
- In addition to adding to oyster grower revenue, this valuation of oyster associated nutrient removal ecosystem services will enhance public awareness of water quality issues and could help shift attitudes to allow increased opportunities for shellfish aquaculture and stimulate local economies.

6.3. Long Island Sound

General waterbody characteristics

Long Island Sound is a large estuary (3,259 km^2), located in the northeastern United States between the states of Connecticut to the north and New York to the south (Fig. 3). Three major tributaries, Connecticut, Thames and Housatonic Rivers, enter from the north. The Sound is connected to the Atlantic Ocean at its eastern end and to the East River and New York Harbor to the west. Tidal height is 1.9 m, average depth is 20 m and residence time is 2-3 months. The watershed area is 12,773 km^2 and includes parts of the states of Vermont, New Hampshire, Massachusetts, Connecticut (CT), Rhode Island, and New York (NY). The watershed is highly developed, particularly in New York City and New Haven, CT metropolitan areas. The watershed population in 2010 was ~8.93 million people with nearly half the population living near the coast. The average watershed population density is 121 persons km^{-2} but in the coastal areas it is much higher; 2,151 persons km^{-2} in the NY metropolitan area and 541 persons km^{-2} in the New Haven, CT metropolitan area. The largest land use is forested (70%) with urban areas making up 16%, and agriculture making up 11% of the watershed.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

Human activities in the watershed contribute to state and local economies but related point and non-point discharge (50,000 tons nitrogen (N) per year in the mid-2000s or 15 tons km^{-2} yr^{-1}) to the Sound has led to severe water quality degradation, particularly low dissolved oxygen. Annual patterns of higher springtime chlorophyll (a measure of phytoplankton, the primary food for shellfish) and lower summer dissolved oxygen concentrations are similar in the 4 regions of the Sound. However, higher chlorophyll and lower dissolved oxygen concentrations, both indicators of high level of nutrient related degradation (Bricker et al., 2007), in the Narrows and Western segments of the Sound show the influence of larger nutrient discharges from the regions of highest population density (Fig. 4). Hypoxia was used in a 2000 analysis designed to guide development of a plan for 58.5% N load reductions (by 2017) to fulfill water quality objectives (NYSDEC and CTDEP, 2000). Wastewater treatment plant upgrades to biological nutrient removal resulted in >40% reductions in N loads by 2012, but further analysis concluded that full attainment of desired water quality standards would require additional reductions or increased assimilative capacity. Nutrient removal by bivalve aquaculture was noted as a possible additional management measure.

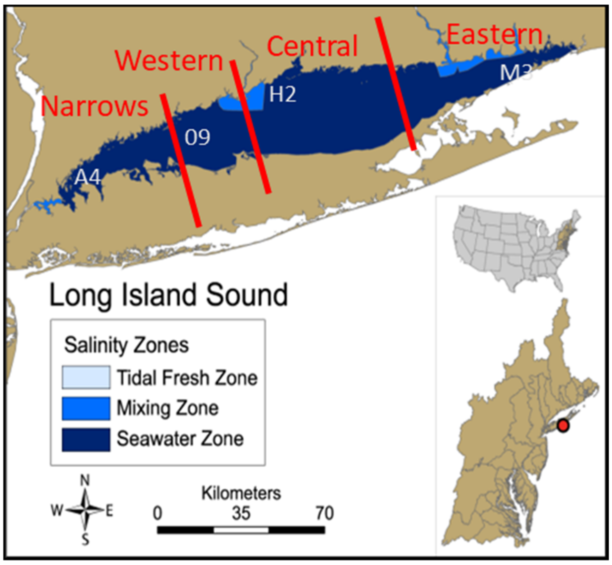


Fig. 3. Location of Long Island Sound in the north eastern US and within the northeast region (inset). The salinity zones represent the annual average, depth average salinity characteristics of the sound (tidal fresh <0.5, mixing zone is 0.5 – 25, seawater zone is > 25). Also shown are station locations of water quality parameters from Fig. 4.

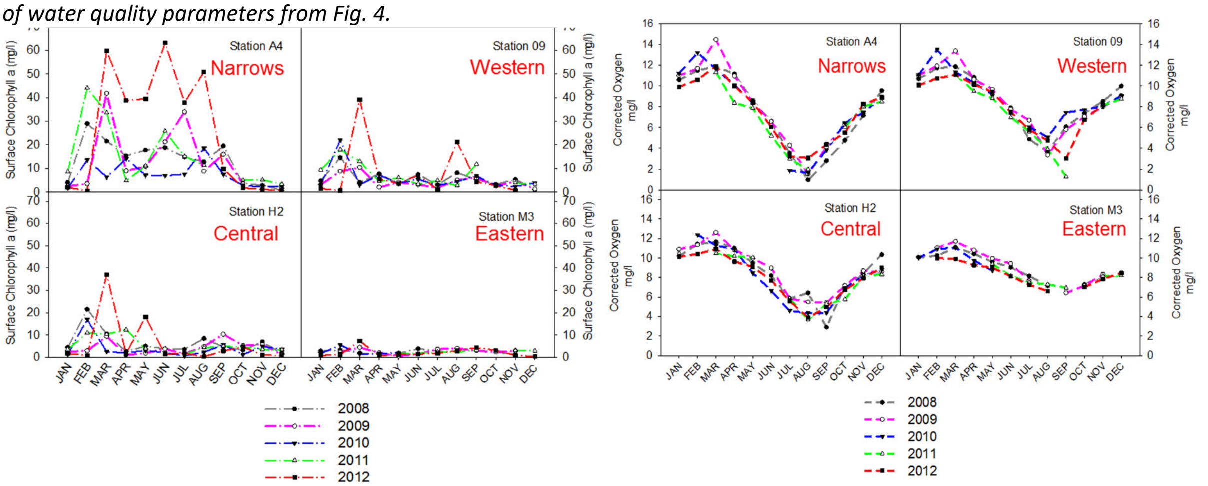


Fig. 4. Monthly chlorophyll concentrations (left) and monthly bottom water dissolved oxygen concentrations (right), January – December, 2008 – 2012 at 4 Long Island Sound stations (see Fig. 3 for locations).

Aquaculture industry characteristics

The shellfish industry in Long Island Sound is long-standing and well established. The industry includes harvest of both Eastern oyster (*Crassostrea virginica*) and hard clam (quahog; *Mercenaria mercenaria*) from shellfish grounds in the states of CT and NY, which share the Sound. Long Island's history, culture, and traditions are linked to notable clam and oyster harvests from the Sound; the NY 'Blue Point' oyster, first marketed as such in 1813, is still known as a high-quality product (Kurlansky, 2006; Churchill, 1920). Connecticut oyster production (4 x 10^6 bushels $year^{-1}$) was third only to Chesapeake Bay in the early 1900s (both Virginia and Maryland produced 5 x 10^6 bushels $year^{-1}$; Churchill, 1920).

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

The growers use diploid oysters, which are slower growing than triploid oysters (which cannot reproduce) because they reproduce and thus lose conditions during spawning periods. For many years NY was the leading producer of hard clams in the United States.

The dockside value of hard clams landed in NY alone from 1972 through 1977 exceeded that of any other fish or shellfish species landed in the state.

Recent cultured and wild caught hard clam and oyster harvests in Long Island Sound have provided over 300 jobs and $30 million in revenue annually (Goldberg et al., 2012). The hard clam harvest has more than tripled in the past decade (Fig. 5), in part because some lobster fishermen have turned to clamming as lobster harvests declined and then closed completely. Recent CT landings data are not available, but NY clam harvest in the Sound declined from 2007 to 2011 partly due to NY clam fishermen harvesting in other waterbodies. Clams are harvested at 4 different sizes for different markets in Long Island Sound with total harvest made up by the following distribution by number: 29% littlenecks (62 mm), 51% topnecks (74 mm), 16% cherrystone (86 mm), 4% chowders (100 mm).

Oyster production grew in the 1980s and 1990s due to culture practices that typically grow oysters on the bottom with no gear. A typical harvest size oyster is about 90g, and the legal length for harvest is 3 inches (7.62 cm). The commercial industry peaked at >200 million oysters harvested in 1992 (>90% harvested in CT), declining by nearly 90% by the early 2000s mainly due to MSX (Multinucleated Sphere Unknown), a parasitic disease caused by *Haplosporidium nelsoni*. Oyster harvests began to rebound in 2006 due to efforts to restore and protect oyster habitats (Fig. 5). Since 2011, CT harvest data have not been available, but resource managers believe that harvests continue to rise.

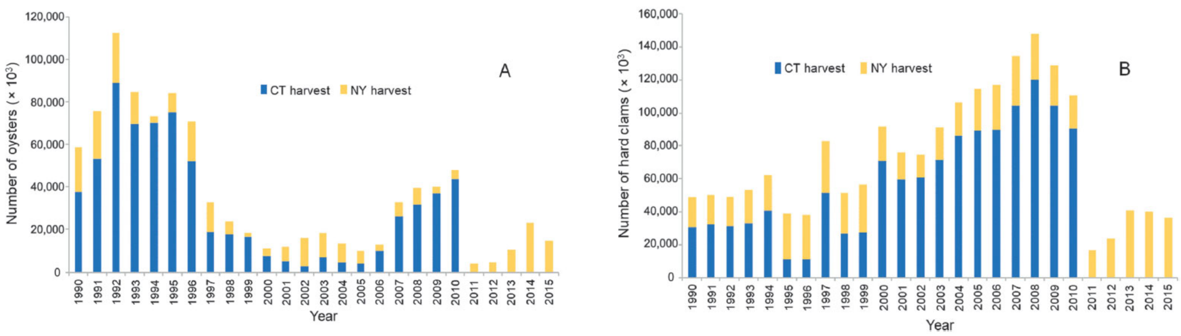


Fig. 5. Long Island Sound (Connecticut, New York) harvest of Oysters (A) and Clams (B) from 1990 - 2015. Note that CT growers stopped reporting harvests in 2011.

In 2012, there were 127,884 acres of approved oyster harvest area in CT, 99,132 acres conditionally approved, 138,849 acres restricted and 23,384 acres classified as prohibited from shellfish harvest (http://longislandsoundstudy.net/). In NY, there were 412,018 approved acres, 1,613 conditionally approved, and 75,499 acres where shell fishing was prohibited (http://longislandsoundstudy.net/). Within the approved and conditional acres in the CT side of Long Island Sound, there were a total of 66,042 acres of leased shellfish growing area. In CT there are about 45 individuals and businesses that hold shellfish leases. In NY, with the exception of one lease in Oyster Bay, 'baymen' can harvest shellfish in any approved waters with the proper permits. One major NY leaseholder accounts for the majority (~80%) of the cultured shellfish landings.

Method for bioextractive nutrient estimation

This bioextraction study (Bricker et al., 2018) used a combination of models, field data, and laboratory experiments to evaluate the impacts of watershed discharges on water quality, and the interaction of shellfish aquaculture and water quality. The objective was to determine the feasibility of using oysters and clams as an additional N management measure and to determine the potential value of the water

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

cleaning ecosystem service that the shellfish provide. The main model used was the ecosystem-scale EcoWin.NET model (EWN; Ferreira, 1995) which used a 2-layer, 42 box model grid to simulate system-scale Eastern oyster (*Crassostrea virginica*) production and associated drawdown of chlorophyll, particulate matter, and nitrogen (N) using relevant components of water transport, biogeochemistry, and shellfish growth models (Bricker et al., 2015). Clam production (*Mercenaria mercenaria*) and N removal was not explicitly simulated, but their impact on the ecosystem was included in the EcoWin.NET ecosystem model as a wild species.

The impact of clam growth was simulated through a model of the drawdown of both phytoplankton and organic detritus (Ferreira et al., 2018) and in this way, the N removal attributed to both oysters and clams could be calculated. The estimated N removal by oyster aquaculture in CT was 656 tons per year, equivalent to 1.31% of total N inputs to the Sound. The clam related N removal of 279 tons per year increases total removal to an equivalent of 1.87% of total N inputs. While this seems small, it is optimistic to note that areal removal rates compare favorably to reported N removal effectiveness of approved agricultural best management practices and stormwater control measures (Rose et al., 2015). Also, denitrification associated losses are not included an can be significant (Kellogg et al., 2013; Ray & Fuleiler, 2020) thus this should be considered a conservative estimate. The next step was to determine the value of the bioextractive nutrient removal service to evaluate potential economic benefits.

An avoided or replacement costs economic analysis was used to estimate the value of the removed N. This method assumes that the N removal service provided by oyster aquaculture would have to be replaced by an alternative management measure (King & Mazzotta, 2000). The costs to remove one kg of N by alternate reduction strategies were used to provide estimates of the economic value of the removed N, giving a range of potential values. Typically the least cost option will be used to determine the value of bioextracted N. The alternative management costs used to assign value to oyster related N removal were; improvements to wastewater treatment plants at three different treatment levels (8, 5, and 3 mg L^{-1}, \$32, \$37 and \$100 kg^{-1} $year^{-1}$), implementation of agricultural BMPs (\$8 kg^{-1} $year^{-1}$), and urban non-point controls (\$350 kg^{-1} $year^{-1}$; Evans, 2008). The range in potential economic value of boiextractively removed N in CT Long Island Sound is \$7.5 - \$330 million per year. If shellfish growers were included in a fully functioning nutrient credit trading program, they could receive payment for the N removal ecosystem service that their shellfish provide.

Reduction of eutrophication symptoms – chlorophyll

Although oyster and clam related N removal in Long Island Sound is small compared to total N inputs, there are additional valuable ecosystem services provided such as water clearance of chlorophyll and other particulates. A calculation of the drawdown of chlorophyll, one of the most recognized indicators of eutrophication (Bricker et al., 2003), shows that the filtration by oysters and clams in CT Long Island Sound has the potential to reduce chlorophyll concentrations. The importance of this is that reduction of chlorophyll increases water transparency which encourages re-growth of seagrasses, and thus increases in habitat, as well as reducing the phytoplankton biomass that contributes to development of hypoxia in bottom waters. Table 1 shows the potential chlorophyll drawdown in CT Long Island Sound based on four model scenarios (Ferreira et al., 2018). The total drawdown by combined oysters and clams is >6% over the total area of the Sound, with greater reductions in the Narrows and Western Sound (6-9% reductions) where eutrophication impacts are most severe (Bricker et al., 2007), compared to the Eastern Sound (0.4% reductions). Clams provide a greater chlorophyll reduction than oysters (Table 16). While no economic valuation was made of the increased water transparency and marine habitat services provided by shellfish, increases in seagrass coverage and increases in bottom water dissolved oxygen are likely also highly valuable, providing habitat for estuarine species such as fish and crabs that are sought after by both recreational and commercial fishermen. Additional loss of N by denitrification has also not been included in the calculation of N losses in this study, though it has

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

been shown to be an important loss pathway in aquaculture, and in some cases can almost double the loss of N from aquaculture sites (e.g. Bricker et al., 2018; Ray & Fulweiler, 2020; Rose et al., 2021). Thus, these estimates of N removal and economic value should be considered as conservative estimates of the potential contribution by oysters and clams to nutrient remediation in coastal waters.

Table 16. Chlorophyll (Chl) drawdown by shellfish in CT Long Island Sound, considering four alternate modelling scenarios (adapted from Ferreira et al., 2018). The drawdown of Chl is greatest in Scenario 1 which includes filtration of both oysters and clams - the concentration of Chl with no shellfish is 11.8 μg L^{-1} but is reduced by 6.1% to 11.1 μg L^{-1}. The reductions are not as great with clams only (Scenario 3, reduction of 4.7%), and oysters only (Scenario 4, reduction of 1.9%).

Scenario	Total Chl drawdown (μg L^{-1})
1. Oysters + Clams (μg chlorophyll L^{-1})	11.1
2. No Shellfish (μg chlorophyll L^{-1})	11.8
3. No oysters (μg chlorophyll L^{-1})	11.3
4. No clams (μg chlorophyll L^{-1})	11.6
Change due to shellfish (oysters + clams) (%)	-6.1
Change due to oysters only (%)	-1.9
Change due to clams only (%)	-4.7

Conclusions and Policy implications

- Bioextractive removal of N through oyster aquaculture and clam harvest does not contribute significantly to the removal of nitrogen compared to total inputs. The 656 tons N per year removed by oysters and 279 tons N removed by clams is equivalent to 1.87% of the total annual N input;
- Based on a cost avoided method of estimation, the value of the N removed through oyster cultivation and harvest in Long Island Sound ranges from $7.5 to $330 million depending on the alternative management measure used to estimate costs. If the shellfish growers were included in a nutrient trading program, the N removal by their shellfish could be credited toward fulfilment of required N reductions and the growers could be compensated for the removal service provided by their shellfish. This is already the case in Chesapeake Bay (Cornwell et al., 2016; Wheeler, 2020);
- Although not valued, there are additional ecosystem services provided by shellfish such as denitrification N losses (Bricker et al., 2018), and increased water transparency that can lead to valuable improvements to water quality such as increased seagrass and habitat, and improved bottom water dissolved oxygen. Thus, the economic value of the shellfish ecosystem service is likely much greater;
- Recent research has shown that costs and N removal efficiency of shellfish cultivation compare favorably with approved Best Management Practices (BMPs; Stephenson et al., 2010; Rose et al., 2015). Nutrient credit trading has been proposed, and in some states implemented, as a tool to achieve water quality goals (Lal, 2010; Branosky et al., 2011; Cornwell et al., 2016; Ferreira & Bricker, 2016). These programs establish a market-based approach to provide economic incentives for achieving nutrient load reductions to meet pollution reduction targets. They could create new revenue opportunities for farmers, entrepreneurs, and others who are able to reduce discharges below allocated levels at low cost and sell credits received to dischargers facing higher-cost reduction options;
- The Connecticut Nitrogen Credit Exchange was created in 2002 to improve nutrient-related hypoxia conditions in Long Island Sound, providing an alternative compliance mechanism for 79 wastewater treatment plants throughout the state. During 2002-2009, 15.5 x 10^6 N credits

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

were exchanged at a value of $46 million, with estimated cost savings of $300-400 million (CT DEP, 2010). The trading of point sources was successful but non-point sources are not yet included. These results provide a basis for inclusion of oyster aquaculture into the trading program as has been done in Chesapeake region (Cornwell et al., 2016; Wheeler, 2020);

- Even if a nutrient credit trading program in Long Island Sound does not include shellfish growers, this valuation of the bioextraction ecosystem services associated with shellfish culture will enhance public awareness of water quality issues and could help shift attitudes to allow increased opportunities for shellfish aquaculture and stimulate local economies.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

7. Policy recommendations

Eutrophication due to anthropogenic nutrient enrichment of marine waters is a major issue in European seas. These 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 which end up in the coastal zones are a more complicated and expensive issue to solve. 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 communities.

The bioextraction capacity of bivalve shellfish is a key regulatory 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 a part of the overall nitrogen budget as a part of integrated catchment management.

The major challenge to the use of bivalves for nutrient management is the determination of the quantity of nutrients that are removed by shellfish species. The range of environments and the diversity of bivalve culture 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 farm bivalves, across a large number of countries, often with significant production volumes.

The approach used in GAIN 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 a 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 as a complement to the well-established bottom-up approach of emissions reduction.

Results show that different shellfish species have different N removal rates. Location and farm culture practices can also have a role to play in removal rates. The Mediterranean mussel, mainly farmed in Spain, had the highest nitrogen removal.

95% of all shellfish farmed in the EU is composed of the five species we analysed: blue mussel, Mediterranean mussel, Pacific oyster, native 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 estimated nitrogen removed from European seas varied between 4 900 tonnes and 13 425 tonnes, depending on the assessment method used. Expressed as population equivalents, these numbers correspond to offsetting the emissions of 1.5-4.1 million Europeans.

In financial terms, the benefits of incorporating cultivated shellfish into a catchment-scale nutrient management scheme are significant (Table 17). The remediation costs of different measures such as stormwater control or agricultural best practices are taken from Rose et al (2015) and Ferreira & Bricker (2019).

Table 17. Financial benefits of an EU-wide nutrient credit trading framework to include shellfish farmers.

Nitrogen removal		Minimum (analytical)	Maximum (FARM)
Nitrogen removed by shellfish (tonnes per year)		4922	13425
Population-Equivalents (PEQ @ 3.3 kg N per ind.)		1490000	4068076
Value of eco-intensification	**Remediation cost (€ kg-1 N)**	**Credit valuation (Millions of €)**	**Credit valuation (Millions of €)**
Stormwater control measures	3388	16601	45483
Approved agricultural BMP	435	2132	5840
Wastewater treatment upgrades	7047	34530	94604
Average credit valuation (millions of €)		17754	48642

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

If we consider only the lower estimate of nitrogen removal by shellfish, the average overall value of nutrient removal totals almost eighteen billion €. The maximum estimates are considerably higher, and it is recognised that the potential valuation is associated to nutrient loading that is for the most part diffuse and is therefore challenging to reduce in many rural areas in Europe without severe 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. Top-down control of eutrophication via shellfish aquaculture is recognised in qualitative terms but there has been no associated policy development at a European or national level.

The draft of policies to control eutrophication should have at the core 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: there are other major societal benefits including greater food security, local employment 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.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

8. References

Abbe, G. R., C. B. McCollough, L. S. Barker & C. F. Dungan. 2010. Performance of disease-tolerant strains of eastern oyster (Crassostrea virginica) in the Patuxent River, Maryland, 2003 to 2007. J. Shellfish Res. 29:161–175.

Ali, K.H.M., Karim, O.A., 2002. Simulation of flow around piers, Journal of Hydraulic Research, 40(2), pp161-174.

APROMAR (2019) La acuicultura en España. http://www.apromar.es/content/la-acuicultura-en-espa%C3%B1a-2019. 91 pp

Beckensteiner, J., D.M. Kaplan and A.M. Scheld. 2020. Barriers to Eastern oyster aquaculture expansion in Virginia. Frontiers in Marine Science 7: Article 53. doi: 10.3389/fmars.2020.00053

Boithias, L., Sauvage, S., Lenica, A., Roux, H., Mikayilov, F., Larnier, K., Dartus, D., Sánchez Pérez, J.M., 2017. Simulating Flash Floods at Hourly Time-Step Using the SWAT Model. Water. 9. 929. 10.3390/w9120929.

Borja, A., Bricker, S.B., Dauer, D.M., Demetriades, N.T., Ferreira, J.G., Forbes, A.T., Hutchings, P., Jia, X., Kenchington, R., Marques, J.C., Zhu, C.B. 2008. Overview of integrative tools and methods in assessing ecological integrity in estuarine and coastal systems worldwide. Marine Pollution Bulletin, 56, 1519–1537.

Bouraoui F, Grizzetti B, Aloe A (2011) Long term nutrient loads entering European seas, EUR 24726 EN, Publications Office of the European Union, Luxembourg, 2011, ISBN 978-92-79-19319-4, doi:10.2788/54513, JRC 62873. 82 pp.

Branosky, E.; Jones, C.; Selman, M. Comparison Tables of State Nutrient Trading Programs in the Chesapeake Bay Watershed; World Resources Institute: Washington, DC, 2011. www.wri.org/publication/ comparison-tables-of-state-chesapeake-baynutrient-trading-programs.

Bricker, S. B., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks & J. Woerner. 2008. Effects of nutrient enrichment in the nation's estuaries: a decade of change. Harmful Algae 8:21–32.

Bricker, S., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks and J. Woerner. 2007. Effects of Nutrient Enrichment in the Nation's Estuaries: A Decade of Change, National Estuarine Eutrophication Assessment Update. NOAA Coastal Ocean Program Decision Analysis Series No. 26. National Centers for Coastal Ocean Science, Silver Spring, MD. 322 pp.

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. Role of Shellfish Aquaculture in the Reduction of Eutrophication in an Urban Estuary. Environ. Sci. Technol. 52:173–183. 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 43:23–38. https://doi.org/10.1007/s12237-019-00661-8

Bricker, S.B., J. Ferreira, C. Zhu, J. Rose, E. Galimany, G. Wikfors, C. Saurel, R. Landeck Miller, J. Wands, P. Trowbridge, R. Grizzle, K. Wellman, R. Rheault, J. Steinberg, A. Jacob, E. Davenport, S. Ayvazian, M. Chintala, and M.Tedesco. 2015. An Ecosystem Services Assessment using bioextraction technologies for removal of nitrogen and other substances in Long Island Sound and the Great Bay/Piscataqua Region Estuaries. NCCOS Coastal Ocean Program Decision Analysis Series No. 194. National Oceanic and Atmospheric Administration, National Centers for Coastal Ocean Science, Silver Spring, MD and United States Environmental Protection Agency, Office of Research and Development, Atlantic Ecology Division, Narragansett, RI. 154 pp + 3 appendices.

Bricker, S.B., J.G. Ferreira, C. Zhu, J.M. Rose, E. Galimany, G.H. Wikfors, C. Saurel, R.L. Miller, J. Wands, P. Trowbridge, R.E. Grizzle, K. Wellman, R. Rheault, J. Steinberg, A.P. Jacob, E.D. Davenport, S. Ayvazian, M. Chintala, and M.A. Tedesco. 2018. The role of shellfish aquaculture in reduction of eutrophication in an urban estuary. Environmental Science & Technology 52 (1): 173– 183.

Bricker, S.B., J.G. Ferreira, T. Simas, 2003. An Integrated Methodology for Assessment of Estuarine Trophic Status. Ecological Modelling, 169(1), 39-60.

Butenschön, M., Clark, J., Aldridge, J.N., Allen, J.I., Artioli, Y., Blackford, J., Bruggeman, J., Cazenave, P-, Ciavatta, S., Kay, S., Lessin, G., van Leeuwen, S., van der Molen, J., de Mora, L., Polimene, L., Sailley, S., Stephens, N., Torres, R., 2016. ERSEM 15.06: a generic model for marine biogeochemistry and the ecosystem dynamics of the lower trophic levels. Geosci. Model Dev. 9, 1293-1339.

Carrere L., F. Lyard, M. Cancet, A. Guillot, N. Picot: FES 2014, a new tidal model - Validation results and perspectives for improvements, presentation to ESA Living Planet Conference, Prague 2016.

Cheng, X., Chen, L., Sun, R. & Jing, Y., 2018. An improved export coefficient model to estimate non-point source phosphorus pollution risks under complex precipitation and terrain conditions. Environmental Science and Pollution Research, Volume 25, p. 20946–20955.

Churchill, Jr., E.P. 1920. The oyster and the oyster industry of the Atlantic and Gulf Coasts. Department of Commerce, Bureau of Fisheries. Bureau of Fisheries Document No. 890. Government Printing Office, Washington, D.C.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

Coffey, R., Benham, B., Wolfe, M., Dorai-Raj, S., Bhreathnach, N., O'Flaherty, V., Cormican, M., Cummins, E., 2016. Sensitivity of streamflow and microbial water quality to future climate and land use change in the West of Ireland. Regional Environmental Change. 16. 10.1007/s10113-015-0912-0.

Collins, A.L., McGonigle, D.F., 2008. Monitoring and modelling diffuse pollution from agriculture for policy support: UK and European experience. Environmental Science & Policy, 11(2), 97-101.

Congrove, M. S. 2008. A bio-economic feasibility model for remote setting: potential for oyster aquaculture in Virginia. Dissertations, Theses, and Masters Projects. Paper 1539617864. Available at: https://dx.doi.org/doi:10.25773/v5-dpen-ek98.

Connecticut Department of Environmental Protection (CT DEP). An Incentive-based Water Quality Trading Program; The Connecticut Department of Environmental Protection, Bureau of Water Protection and Land Reuse: Hartford, CT, 2010

Cooper, S.R. and G.S. Brush. 1993. A 2,500-Year History of Anoxia and Eutrophication in Chesapeake Bay. Estuaries 16, No. 3B, p. 617-626.

Cornwell, J., J. Rose, L. Kellogg,M. Luckenbach, S. Bricker, K. Paynter, C. Moore, M. Parker, L. Sanford, B. Wolinski, A. Lacatell, L. Fegley, and K. Hudson. 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. Report to the Chesapeake Bay Program. http://www.chesapeakebay.net/documents/Oyster_BMP_1st_Report_Final_Approved_2016-12-19.pdf. Accessed 4 Jan 2017.

Cubillo, A. M., S. B. Bricker, M. Parker & J.G. Ferreira. 2018. NCCOS Ecological Assessment to support NOAA's Choptank Complex Habitat Focus Area: eutrophication and shellfish aquaculture/ restoration ecosystem services modeling: final report. Submitted to NOAA NCCOS, Cooperative Oxford Laboratory by Longline Environmental, Ltd., National Oceanic and Atmospheric Administration, National Ocean Service, National Centers for Coastal Ocean Science, Silver Spring, MD.

Cubillo, A.M., Ferreira, J.G., Kamermans, P., Bricker, S., Parker, M., Filgueira, R., 2017. Modelling bivalve growth in different food environments [Conference presentation]. In: European Aquaculture Society 2017. Dubrovnik, Croatia

Cubillo, A.M., Ferreira, J.G., Pearce, C.M., Marshall, R., Cheney, D., Hudson, B., 2018. Ecosystem services of geoduck farming in South Puget Sound, USA: a modeling analysis. Aquac. Int. 26:1427–1443. https://doi.org/10.1007/s10499-018-0291-x

Cubillo, AM, Ferreira, JG, Lencart-Silva, J, Taylor, NGH, Kennerley, A, Guilder, J, Kay, S, Kamermans, P., 2021. Direct effects of climate change on productivity of European aquaculture. Aquaculture International (2021). https://doi.org/10.1007/s10499-021-00694-6

Deltares, 2010. Delft3D-FLOW, Simulation of multi-dimensional flows and tranport phenomena, including sediments, Deltares, User Manual. V.3.14, Delft, the Netherlands.

Diaz R.J., Rosenberg, R., 2008. Spreading dead zones and consequences for marine ecosystems. Science 321, 926–929.

DOENI. 2009. Tables of Co-ordinates of Shellfish Waters. http://www.doeni.gov.uk/niea/schedule_to_legal_notice_for_shellfish_waters_designations_june_2009.pdf. Accessed January 2011.

Dvarskas, A., S.B. Bricker, G.H. Wikfors, J. Bohorquez, M.S. Dixon, J.M. Rose, 2020. Quantification and valuation of nitrogen removal services provided by commercial shellfish aquaculture at the subwatershed scale. Environmental Science and Technology. 54 (24), 16156-16165 DOI: 10.1021/acs.est.0c03066

EC (2017) CAP monitoring and evaluation indicators 2014-2020, CAP context indicators. Water quality (https://ec.europa.eu/agriculture/cap-indicators/context_en or https://ec.europa.eu/agriculture/cap-indicators/context/2017/c40_en.pdf). Accessed 27 March 201

EUMOFA. (2019). THE EU FISH MARKET 2019. https://doi.org/10.2771/168390;

EUMOFA. (2020). Coronavirus response EUMOFA's weekly data and trends analysis week 17. European Commission. (2);

EUMOFA. (2020). Coronavirus response EUMOFA's weekly data and trends analysis week 19. European Commission. (1);

EUMOFA. (2020). Coronavirus response EUMOFA's weekly data and trends analysis Weeks 24 - 25. European Commission. (3);

Eurostat. (2020). Eurostat. Retrieved 12 October 2020, from https://ec.europa.eu/eurostat/data/database;

Evans, B.M. 2008. An evaluation of potential nitrogen load reductions to Long Island Sound from the Connecticut River Basin. Report Submitted to New England Interstate Water Pollution Control Commission. University Park: Penn State Institutes of Energy and the Environment.

Fabinyi, M., Liu, N., Song, Q., Li, R., 2016. Aquatic product consumption patterns and perceptions among the Chinese middle class. Reg. Stud. Mar. Sci. 7, 1–9. doi:10.1016/j.rsma.2016.01.013

FAO, 2007. Future prospects for fish and fishery products. 4. Fish consumption in the European Union in 2015 and 2030. Part 1. European overview., FAO Fisheries Circular. Rome.

Ferreira J.G., Bricker S.B., 2019. Assessment of Nutrient Trading Services from Bivalve Farming. In: Smaal A., Ferreira J., Grant J., Petersen J., Strand Ø. (eds) Goods and Services of Marine Bivalves. Springer, Cham. https://doi.org/10.1007/978-3-319-

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

96776-9_27

Ferreira, J. G. EcoWin - An object-oriented ecological model for aquatic ecosystems. Ecol. Mod. 1995, 79, 21−34.

Ferreira, J. G., A. J. S. Hawkins & S. B. Bricker. 2007. Management of productivity, environmental effects and profitability of shellfish aquaculture - the Farm Aquaculture Resource Management (FARM) model. Aquaculture 264:160–174.

Ferreira, J. G., A. Sequeira, A. J. S. Hawkins, A. Newton, T. D. Nickell, R. Pastres, J. Forte, A. Bodoy & S. B. Bricker. 2009. Analysis of coastal and offshore aquaculture: application of the FARM model to multiple systems and shellfish species. Aquaculture 289:32–41.

Ferreira, J. G., J. H. Andersen, A. Borja, S. B. Bricker, J. Camp, M. Cardoso da Silva, E. Garcés, A.-S. Heiskanen, Christoph Humborg, L. Ignatiades, C. Lancelot, A. Menesguen, P. Tett, N. Hoepffner, U. Claussen. 2011. Indicators of human-induced eutrophication to assess the environmental status within the European Marine Strategy Framework Directive. Estuarine, Coastal and Shelf Science 93: 117-131.

Ferreira, J.G., Andersen, J.H., Borja, A., Bricker, S. B., Camp, J., Cardoso da Silva, M., Garcés, E., Heiskanen, A. S., Humborg, C., Ignatiades, L., Lancelot, C., Menesguen, A., Tett, P., Hoepffner, N., Claussen, U., 2010. Marine Strategy Framework Directive: A tool for eutrophication assessment and management in marine waters. JRC Sci. Tech. Reports Plenary lecture. Eutro2010, Denmark.49 pp.

Ferreira, J.G., Andersen, J.H., Borja, A., Bricker, S. B., Camp, J., Cardoso da Silva, M., Garcés, E., Heiskanen, A. S., Humborg, C., Ignatiades, L., Lancelot, C., Menesguen, A., Tett, P., Hoepffner, N., Claussen, U., 2011. Overview of eutrophication indicators to assess environmental status within the European Marine Strategy Framework Directive. Estuar. Coast. Shelf. Sci. 93:117–131. https://doi.org/10.1016/j.ecss.2011.03.014

Ferreira, J.G., Bricker, S.B., 2016. Goods and services of extensive aquaculture: shellfish culture and nutrient trading. Aquaculture International, 24 (3), 803–825.

Ferreira, J.G., Hawkins, A., Monteiro, P., Service, M., Moore, H., Edwards, A., Gowen, R., Lourenco, P., Mellor, A., Nunes, J., Pascoe, P., Ramos, L., Sequeira, A., Simas, T., Strong, J., 2008a. SMILE - Sustainable Mariculture in northern Irish Lough Ecosystems. Assessment of Carrying Capacity for Environmentally Sustainable Shellfish Culture in Carlingford Lough, Strangford Lough, Belfast Lough, Larne Lough and Lough Foyle

Ferreira, J.G., Hawkins, A.J.S., Monteiro, P., Moore, H., Service, M., Pascoe, P. L., Ramos, L., Sequeira, A., 2008b. Integrated assessment of ecosystem-scale carrying capacity in shellfish growing areas. Aquaculture 275:138–151. https://doi.org/10.1016/j.aquaculture.2007.12.018

Ferreira, J.G., R.A. Corner, H. Moore, M. Service, S.B. Bricker, and R. Rheault. 2018. Ecological carrying capacity for shellfish aquaculture – Sustainability of naturally occurring filter-feeders and cultivated bivalves. Journal of Shellfish Research 37 (3): 1–18.

Ferreira, J.G., S.B. Bricker. 2016. Goods and services of extensive aquaculture: shellfish culture and nutrient trading. Aquaculture International, 24 (3), 803–825. DOI 10.1007/s10499-015-9949-9

Ferreira, J.G., Taylor N.G.H., Cubillo, A., Lencart-Silva, J., Pastres, R., Bergh, Ø., Guilder, J., 2021. An integrated model for aquaculture production, pathogen interaction, and environmental effects. Aquaculture 536:736438. https://doi.org/10.1016/j.aquaculture.2021.736438

FWR, 1993. A framework for marine and estuarine model specification in the UK. March 1993, pp 69.

Gerritsen, J., Holland, A.F., Irvine, D.E., 1994. Suspension-feeding bivalves and the fate of primary production: an estuarine model applied to Chesapeake Bay. Estuaries 17 (2), 403–416.

Girard, S., Mariojouls, C., Paquotte, P., Wisner-bourgeois, C., 1998. An analysis of seafood consumption survey methods in France; in: IXth IIFET Conference. Tromso, Norway, pp. 8–11.

Goldberg, R., R. Mercaldo-Allen, J. M. Rose, P. Clark, C. Kuropat, S. Meseck, and J. Pereira. 2012 Effects of hydraulic shellfish dredging on the ecology of a cultivated clam bed. Aquacult. Environ. Interact. 3: 11–21.

Gunningham, N., Sinclair, D., 2005. Policy Instrument Choice and Diffuse Source Pollution. J. Env. Law, 17 (1), 51–81.

Guo, J., 2014. Semi-analytic model for temporal clear water scour at prototype piers, Journal of Hydraulic Research, 52(3), pp. 366-374.

HELCOM (2015) Updated Fifth Baltic Sea pollution load compilation (PLC-5.5). Baltic Sea Environment Proceedings No. 145

HELCOM (2018) Sources and pathways of nutrients to the Baltic Sea. Baltic Sea Environment Proceedings No. 153

HELCOM (2018): HELCOM Thematic assessment of eutrophication 2011-2016. Baltic Sea Environment Proceedings No. 156.

Higgins, C.B., K. Stephenson, B.L. Brown. Nutrient Bioassimilation Capacity of Aquacultured Oysters: Quantification of an Ecosystem Service. Journal of Environmental Quality, 2011; 40 (1), 271-277.

Howarth, R. W. & R. Marino. 2006. Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: evolving views over three decades. Limnol. Oceanogr. 51:364–376.

Hudson, K. 2019. Virginia Shellfish Aquaculture Situation and Outlook Report: Results of the 2018 Virginia Shellfish Aquaculture Crop Reporting Survey. Virginia Institute of Marine Science, Marine Advisory Program and Virginia SeaGrant.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

Jones, C., E. Branosky, M. Selman & M. Perez. 2010. How nutrient trading could help restore the Chesapeake Bay, WRI Working Paper. Washington, DC.

Kellogg, M., J. Cornwell, M. Owens&K. Paynter. 2013. Denitrification and nutrient assimilation on a restored oyster reef. Mar. Ecol. Prog.Ser. 480:1–19.

Kemp W. M., W. R. Boynton, J.E. Adolf, D.F. Boesch, W.C. Boicourt, G. Brush, J.C. Cornwell, T.R. Fisher, P.M. Glibert, J.D. Hagy, L.W. Harding, E. D. Houde, D.G. Kimmel, W.D. Miller, R.I.E. Newell, M.R. Roman, E.M. Smith, J.C. Stevenson. 2005. Eutrophication of Chesapeake Bay: historical trends and ecological interactions. Marine Ecology Progress Series 303: 1–29.

King, D. M.; Mazzotta, M. J. Ecosystem Valuation. Methods, Section 1. Market Price Method, 2000. www.ecosystemvaluation.org/market_price.htm (accessed February 28, 2013).

Kingsley-Smith, P. R., H. D. Harwell, M. L. Kellogg, S. M. Allen, S. K.Allen, D. W. Meritt, K. T. J. Paynter & M. W. Luckenbach. 2009. Survival and growth of triploid Crassostrea virginica (Gmelin, 1791) and C. ariakensis (Fujita, 1913) in bottom environments of Chesapeake Bay: implications for an introduction. J. Shellfish Res. 28:169–184.

Krause, G.; Hörterer, C., Petereit, J. (2020): Report on consumer and stakeholder acceptance of eco-intensification measures, including impact assessment of improved information availability. Deliverable 3.7. GAIN - Green Aquaculture INtensification in Europe. EU Horizon2020 project grant nº. 773330. 74 pp.

Kristensen, P., Whalley, C., Zal, F. N. N., Christiansen, T. (2018). European waters assessment of status and pressures 2018. EEA Report, (7/2018).

Kurlansky, M. 2006. The Big Oyster: History on the half shell. Random House LLC, New York. 307 p.

Lal, H., J. A. Delgado, C. M. Gross, E. Hesketh, S. P. McKinney, H. Cover, M. Shaffer. 2009. Market-based approaches and tools for improving water and air quality. Environmental Science & Policy 12:1028–1039.

Lefebvre A. and D. Devreker. 2020. First Comprehensive Quantitative Multi-Parameter Assessment of the Eutrophication Status from Coastal to Marine French Waters in the English Channel, the Celtic Sea, the Bay of Biscay, and the Mediterranean Sea. J. Mar. Sci. Eng. 2020, 8(8),561; https://doi.org/10.3390/jmse8080561

Lesser, G.P., Roelvink, J.A., van Kester, J.A.T.M. and Stelling, G.S., 2004. Development and validation of a three-dimensional morphodynamic model. Coastal Engineering, 51, 883-915.

Longline Environment Ltd. (2020). META - Maritime and Environmental Thresholds for Aquaculture. Retrieved 12 March 2021, from www.longline.co.uk/meta.

Lopes, A. S., Ferreira, J. G., Vale, C., & Johansen, J. (2017). The mass balance of production and consumption: Supporting policy-makers for aquatic food security. Estuarine, Coastal and Shelf Science, 188, 212–223. https://doi.org/10.1016/j.ecss.2017.02.022

Lopes, A., Ferreira, J.G, 2020. Mass balance of EU consumption and production of aquatic products in the European Union, and implications for eco-intensification of the aquaculture sector. GAIN - Green Aquaculture INtensification in Europe. EU Horizon 2020 project report. 48 pp.

Lyard F., L. Carrere, M. Cancet, A. Guillot, N. Picot: FES2014, a new finite elements tidal model for global ocean, Ocean Dynamics, 2016.

Malagó A, Bouraoui F, Grizzetti B, De Roo A (2019) Modelling nutrient fluxes into the Mediterranean Sea. Journal of Hydrology: Regional Studies, Volume 22. https://doi.org/10.1016/j.ejrh.2019.01.004

Malone, T. C., D. J. Conley, T. R. Fisher, P. M. Glibert, L.W. Harding, K. G. Sellner, S. Estuaries, P. B. Dedicated, I. Nutrients, W. Jun, T. C. Malone, D. J. Conley, T. R. Fisher, P. M. Glibert & L. W. Harding. 1996. Scales of nutrient-limited phytoplankton productivityin Chesapeake Bay. Estuaries 19:371–385.

Marteleira, R., Roxo, M. J., Lasco, R. & Coelho, P. S., 2018. Hydrological modelling to assess water resources resilience towards climate change impacts in Tacloban, Philippines. Climate, Disaster and Development Journal, 3(1), pp. 13-22.

Maryland Aquaculture Coordinating Council (MD ACC) Shellfish Aquaculture Update. December 10, 2020.

Maryland Chesapeake Bay Oyster Management Plan - Fisheries Management Plan (MD FMP). 2019. Maryland Department of Natural Resources, Fishing and Boating Services 580 Taylor Ave. B-2, Annapolis, MD.

Maryland Department Natural Resources (MD DNR). 2019. Aquaculture coordinating council shellfish aquaculture update. Accessed November 14, 2019. Available at: https://calendarmedia.blob.core.windows.net/assets/5692bc27-1a94-4096-b29f-e7e2088025d3.pdf.

Meritt, D. & D. Webster. 2013. Remote setting systems. Oyster aquaculture technology series. AGNR-AO-0-11-05.

Miller, A., 1958. The Effect of winds on water levels in the New England coast. Limnology & Oceanography, 3(1), 1-14.

Moriasi, D., Gitau, M., Pai, N., Daggupati, P., 2015. Hydrologic and Water Quality Models: Performance Measures and Evaluation Criteria. Transactions of the ASABE (American Society of Agricultural and Biological Engineers). 58. 1763-1785. 10.13031/trans.58.10715.

Nagy H, Lyons K, Nolan G, Cure M, Dabrowski T., 2020. A Regional Operational Model for the North East Atlantic: Model Configuration and Validation. Journal of Marine Science and Engineering 8(9):673. https://doi.org/10.3390/jmse8090673

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

Nobre, A.M., Ferreira, J.G., Nunes, J.P., Yan, X., Bricker, S., Corner, R., Groom, S., Gu, H., Hawkins, A. J. S., Hutson, R., Lan, D., Lencart e Silva, J. D., Pascoe, P., Telfer, T., Zhang, X., Zhu, M., 2010. Assessment of coastal management options by means of multilayered ecosystem models. Estuar. Coast. Shelf. Sci. 87:43–62. https://doi.org/10.1016/j.ecss.2009.12.013

Norderhaug, K. M., Gundersen, H., Høgåsen, T., Johnsen, T. M., Severinsen, G., Vedal, J., Sørensen, K., Walday, M. (2016). Eutrophication status for Norwegian waters. National report for the third application of OSPARs Common Procedure. Norwegian Environmental Agency.

Nunes, J.P., Jacinto, R., Keizer, J., 2017. Combined impacts of climate and socio-economic scenarios on irrigation water availability for a dry Mediterranean reservoir. The Science of the total environment. 584-585. 219-233. 10.1016/j.scitotenv.2017.01.131.

NYSDEC; CTDEP. A Total Maximum Daily Load Analysis to Achieve Water Quality Standards for Dissolved Oxygen in Long Island Sound. Prepared in Conformance with Section 303(d) of the Clean Water Act and Long Island Sound Study; NYSDEC: Albany, NY, 2000.

OSPAR (2017) Eutrophication Status of the OSPAR Maritime Area. OSPAR Commission, London, OSPAR publication no. 694/2017, ISBN: 978-1-911458-34-0, 166 pp.

OSPAR Commission. 2017. Eutrophication Status of the OSPAR Maritime Area. Third Integrated Report on the Eutrophication Status of the OSPAR Maritime Area https://www.ospar.org/documents?v=37502

OSPAR, 2010. Moffat, C., R. Emmerson, A. Weiss, C. Symon, L. Dicks (Eds.) Quality Status Report 2010.

Parker, M. and S. Bricker. 2020. Sustainable oyster aquaculture, water quality improvement, and ecosystem service value potential in Maryland Chesapeake Bay. Journal of Shellfish Research 39(2): 1–13.

Parker, M., D. Lipton & R. M. Harrell. 2020. Impact financing and aquaculture: Maryland oyster aquaculture profitability. J. World Aquacult. Soc. 2020:1–22.

Pauly, D., Christensen, V., Guénette, S., Pitcher, T.J., Sumaila, U.R., Walters, C.J., Watson, R., Zeller, D., 2002. Towards sustainability in world fisheries. Nature 418, 689–695. doi:10.1038/nature01017

Petersen, J.K., B. Hasler, K. Timmermann, P. Nielsen, D.B. Tørring, M.M. Larsen, M. Holmer, 2014. Mussels as a tool for mitigation of nutrients in the marine environment. Marine Pollution Bulletin 82, 137–143.

Pieniak, Z., Vanhonacker, F., Verbeke, W., 2013. Consumer knowledge and use of information about fish and aquaculture. Food Policy 40, 25–30. doi:10.1016/j.foodpol.2013.01.005

Ray, N.E. and R. W. Fulweiler. 2021. Meta-analysis of oyster impacts on coastal biogeochemistry. Nature Sustainability 4: 261–269.

Roberts, D., Davies, C., Mitchell, A. Moore, H., Picton, B., Portig, A., Preston, J., Service, M., Smyth, D., Strong, D., Vize, S. 2004. Strangford Lough Ecological Change Investigation (SLECI). Work Package 8: Aquaculture activity in Strangford Lough. Report to Environment and Heritage Service by the Queen's University Belfast.

Rodgers, P., Dawson, A., Espino, D.C., Dias, J.F., Hoyo, J.J.G. de, Ordaz, F.G., Toríbiio, R.J., Kapantagapis, A., Lasch, R., Le Grel, L., Mendonça, M., Pirazzoli, C., 2008. Marketing and price formation of fisheries and aquaculture products 161.

Rose, J.M., J. Stephen Gosnell, S. Bricker, M.J. Brush, A. Colden, L. Harris, E. Karplus, A. Laferriere, N.H. Merrill, T. Murphy, J. Reitsma, J. Shockley, K. Stephenson, S. Theuerkauf, D. Ward, R.W. Fulweiler. 2021.. Opportunities and challenges of including oyster-mediated denitrification in nitrogen management plans. Estuaries and Coasts. accepted March25. 2021.

Rose, J.M., S.B. Bricker, J.G. Ferreira. 2015. Modeling shellfish farms to predict harvest-based nitrogen removal. Marine Pollution Bulletin 91: 185–190.

Saurel, C., Ferreira, J.G., Cheney D., Suhrbier, A., Dewey, B., Davis, J., Cordell, J., 2014. Ecosystem goods and services from Manila clam culture in Puget Sound: a modelling analysis. Aquac. Environ. Interact. 5:255–270. https://doi.org/10.3354/aei00109

Scholten, H., Smaal, A.C., 1998. Responses of Mytilus edulis L. to varying food concentrations: Testing EMMY, an ecophysiological model. J. Exp. Mar. Bio. Ecol. 219:217–239. https://doi.org/10.1016/S0022-0981(97)00182-2

Scholten, H., Smaal, A.C., 1999. The ecophysiological response of mussels (Mytilus edulis) in mesocosms to a range of inorganic nutrient loads: Simulations with the model EMMY. Aquat. Ecol. https://doi.org/10.1023/A:1009995823741

Seed, R. 1969. The ecology of Mytilus edulis L. (Lamellibranchiata) on exposed rocky shores 2. Growth and mortality. Oecologia, 3, 317-350.

Sequeira, A., Ferreira, J.G., Hawkins, A.J.S., Nobre, A., Lourenço, P., Zhang, X.L., Yan, X., Nickell, T., 2008. Trade-offs between shellfish aquaculture and benthic biodiversity: A modelling approach for sustainable management. Aquaculture 274:313–328. https://doi.org/10.1016/j.aquaculture.2007.10.054

Skarbøvik E, Stålnacke P, Austnes K, Selvik JR, Pengerud A, Tjomsland T, Høgåsen T, Beldring S (2013) Riverine inputs and direct discharges to Norwegian coastal waters – 2012, Norwegian Institute for Water Research Report No. M-80/2013, ISBN 978-82-577-6319-0

Skarbøvik E, Stålnacke P, Kaste Ø, Selvik JR, Tjomsland T, Høgåsen T, Aakerøy PA, Beldring S (2010) Riverine inputs and direct

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

discharges to Norwegian coastal waters – 2009. Climate and Pollution Agency TA-2726/2010; 75 pp.

Skarbøvik E, Stålnacke P, Selvik JR, Aakerøy PA, Høgåsen T, Kaste Ø (2011) Elvetilførselsprogrammet (RID) – 20 års overvåking av tilførsler til norske kystområder (1990-2009). NIVA report 6235-2011. Klima- og forurensningsdirektoratet TA-2857/2011; 55 pp.

Skarbøvik E, Stålnacke PG, Kaste Ø, Selvik JR, Tjomsland T, Høgåsen T, Beldring S (2012) Riverine inputs and direct discharges to Norwegian coastal waters – 2011. NIVA Report 6439-2012. Climate and Pollution Agency TA-2986/2012; 66 pp.

Stålnacke P, Haaland S, Skarbøvik E, Turtumøygard S, Nytrø TE, Selvik JR, Høgåsen T, Tjomsland T, Kaste Ø, Enerstvedt KE (2009) Revision and assessment of Norwegian RID data 1990-2007. Bioforsk Report Vol. 4 No. 138. SFT report TA-2559/2009. 20pp.

Stephenson, K., S. Aultman, T. Metcalfe, and A. Miller. 2010. An evaluation of nutrient non-point offset trading in Virginia: A role for agricultural non-point sources?: Water Resources. Res. 46: WO4519.

Tjomsland T, Kaste Ø, Enerstvedt KE (2009) Revision and assessment of Norwegian RID data 1990-2007. Bioforsk Report Vol. 4 No. 138. SFT report TA-2559/2009. 20pp.

United States Environmental Protection Agency (US EPA). 2001. Nutrient criteria technical guidance manual - estuarine and coastal marine waters. Washington, DC: United States Environmental Protection Agency, Office of Water. EPA-822-B-01-003.

van Senten, J., C. Engle, M. Parker, and D. Webster. 2019. Final Report: Analysis of the Economic Benefits of the Maryland Shellfish Aquaculture Industry. Virginia Tech and Engle-Stone Aquatics.

Virginia Department of Environmental Quality (VA DEQ). 2017. Exchange Compliance Plan 2017 annual update.

Virginia Nutrient Credit Exchange Association (VNCEA). 2017. Exchange Compliance Plan 2017 annual update. Submitted to the Virginia Department of Environmental Quality.

Wheeler, T. 2020. Oyster growers hope polluters will shell out for nutrient credits. Bay Journal 130:14–16.

Xiao, Y. et al., 2007. Trophic Assessment in Chinese Coastal Systems - Review of Methods and Application to the Changjiang (Yangtze) Estuary and Jiaozhou Bay. Estuaries and Coasts, 30(6), pp. 901-918.

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

9. Appendix 1

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
	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

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

	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

The project has received funding from the European Union's Horizon 2020 Framework Research and Innovation Programme under GA n. 773330

Policy white paper

Can bivalve shellfish be used for nutrient management of European seas? A roadmap for a bioextraction nutrient management program

Moore, H.; Ferreira, J.G.; Bricker, S.B.; Cubillo, A.M.; Lopes, A.S.; Service, M.

Summary

Nutrient pollution, or eutrophication, of marine coastal waterbodies is a concern in European seas, some of which have exhibited eutrophication impacts for decades. Nutrient discharges from land-based sources such as wastewater treatment plants, agricultural and urban use of fertilizer, and atmospheric sources from burning of fossil fuels cause a predictable sequence of water quality and environmental degradation.

The symptoms of eutrophication are more serious and progress more quickly in sensitive waterbodies that have a reduced capability to dilute or flush incoming nutrients—as a consequence, tolerance and resilience of ecosystems to nutrients are variable. Nutrient-related impacts include intense algal blooms that cloud waters and kill seagrass, low dissolved oxygen in bottom waters that kill fish and other organisms that are unable to leave the area, and blooms of harmful or toxic algae, called ‘red tides’, that cause closure of fisheries for public health reasons and can cause respiratory distress if aerosolized in the surf zone, leading to reductions in tourism. Although management strategies have reduced nutrient discharge in recent years, additional management is needed because eutrophic symptoms continue to degrade water quality and habitat of many European seas.

Bivalve shellfish (mussels, oysters, clams) remove nutrients from the water through filtration of suspended particles as they feed. This nutrient removal capacity, called ‘bioextraction’, is a *key regulatory ecosystem service* that contributes to eutrophication control; to date it has not been used in Europe as part of a management framework. In the USA, several jurisdictions include harvest of cultivated bivalves as part of a comprehensive nutrient management program. Because the water cleaning service has an economic value, the shellfish farmers might also be compensated for the nutrients removed by their shellfish. The EU shellfish aquaculture industry is well developed and growing—it should be used for nutrient remediation and protection of coastal marine resources in addition to provision of seafood.

This policy document is based on a study aimed at evaluating the nutrient removal capability of current shellfish aquaculture in the EU. The main objective was to illustrate how bivalve aquaculture could contribute to the provision of sustainable locally grown shellfish for consumption while at the same time, improving water quality. This will potentially provide an extra source of income for shellfish farmers for nutrient removal and helps fulfil obligations for compliance with water-related legislation, such as nutrient reductions required by the Water Framework and Marine Strategy Framework Directives.

Figure 1. Roadmap for a nutrient management programme that includes shellfish bioextraction.

Current European bivalve shellfish aquaculture could remove 4900 – 9800 tonnes of nitrogen per year, equivalent to water treatment for 1.5 to 3 million people. The estimated avoided cost economic value of the N removal water cleaning service is €18 billion to €36 billion. This analysis and lessons learned from case studies will be key to development of an effective nutrient management program that includes bivalve shellfish growers.

This document describes the work done, and its main conclusions, at policy level, and provides a roadmap for development of a roadmap for a bioextraction nutrient management programme in the European Union.

Why we care

Nutrient discharges cause eutrophication in European seas. Nitrogen is a particular concern in marine and coastal waters because it is often the limiting nutrient for primary production.

The European Union Marine Strategy Framework Directive (MSFD) describes eutrophication as 'a process driven by enrichment of water by nutrients, especially compounds of nitrogen and/or phosphorus, leading to: increased growth, primary production and biomass of algae; changes in the balance of organisms, and water quality degradation. The consequences of eutrophication are undesirable if they appreciably degrade ecosystem health and/or the sustainable provision of goods and services.' Eutrophic conditions have been observed in European seas for decades.

Recent assessments of the eutrophication status of European coastal waterbodies showed that 97% of the Baltic and 7% of the North-East Atlantic are considered eutrophic. Parts of the Greater North Sea, the Celtic Sea, and Bay of Biscay, and the inner coastal areas of the Skagerrak were classified as eutrophication problem areas. Additional assessments of the English Channel, southern bight of the North Sea, the Celtic Sea, the Bay of Biscay and the Western Mediterranean Sea showed that, despite efforts in recent decades to reduce inputs, the nutrient pressure on coastal marine ecosystems was still high. These assessment results confirm that continuing nutrient management is needed to protect marine ecosystems of Europe.

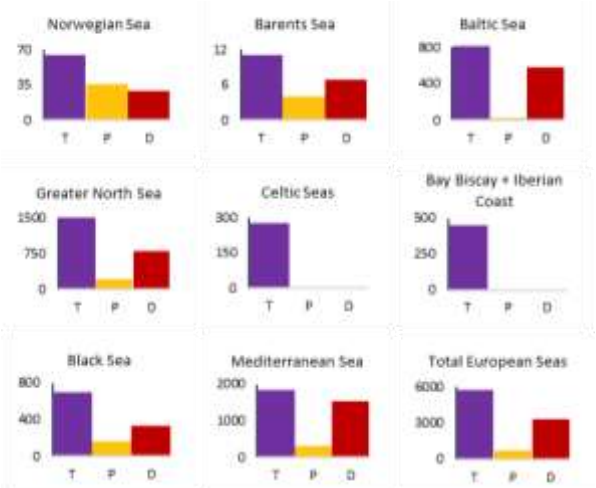


Figure 2. Proportion of N inputs to the various European seas from point sources (P), diffuse sources (D) and total loading (T). Discharges to the Mediterranean Sea and the total European seas include loading from North Africa.

The primary driver of eutrophication is nutrient discharge from human-related activities in the coastal zone. Here we focus on nitrogen because it is frequently the nutrient that leads to eutrophication development in estuarine and marine coastal waters.

Traditional nutrient management strategies include reductions from *point sources* such as urban and industrial wastewater treatment facilities that discharge nutrients through a pipe.

Management of and *non-point or diffuse sources* such as runoff of fertilizer from agricultural, urban and suburban areas, animal feeding operations, and atmospheric nutrient deposition from burning of fossil fuels, present a greater challenge.

Point sources are more easily controlled and management strategies are less costly than non-point source reduction strategies. Non-point source reduction is logistically difficult and costly, and in some cases may have serious social consequences if, for example, rural areas are required to implement costly solutions to reduce nutrient leakage from farms.

In many EU areas, point sources have been significantly reduced and diffuse sources are now the major nutrient source stimulating eutrophication development. Quantification of nitrogen loads to European seas shows that, with the exception of the Norwegian Sea, where salmon and trout industry discharges are significant point source inputs, the greatest proportion of N discharges are diffuse (Figure 2).

This is a challenge from both cost and logistics perspectives for future nutrient management planning.

Can bivalve shellfish aquaculture help reduce eutrophication impacts?

One promising nutrient management measure that could complement traditional land-based strategies is to use the power of cultivated filter-feeding bivalve shellfish (mussels, oysters, clams) which remove particulates directly from the water as they feed. Known as 'bioextraction', microscopic algae and detritus are digested by the shellfish, part is used for growth and the remainder is expelled as faeces, pseudofaeces, and ammonia. When the shellfish are harvested, the N is removed from the system, thereby recycling nutrients from sea to land. In addition, while the shellfish are growing, they clear the water of phytoplankton and detritus, thereby increasing transparency, reducing decomposition of organic matter, and allowing restoration of bottom-growing plants such as seagrasses (Figure 3).

Shellfish farms can additionally remove N via a process called denitrification where bio-deposits from aquaculture operations lead to loss of nitrogen gas to the atmosphere. Some studies show that this process may remove an amount of N equivalent to harvesting shellfish. Our study evaluated the removal within bivalve meat and shell and thus might be considered a conservative estimate of the true value of this ecosystem service.

The EU has a well-developed and growing bivalve shellfish industry that could be considered for possible inclusion in a comprehensive nutrient management program where growers are credited for the nutrients that their shellfish remove from the water.

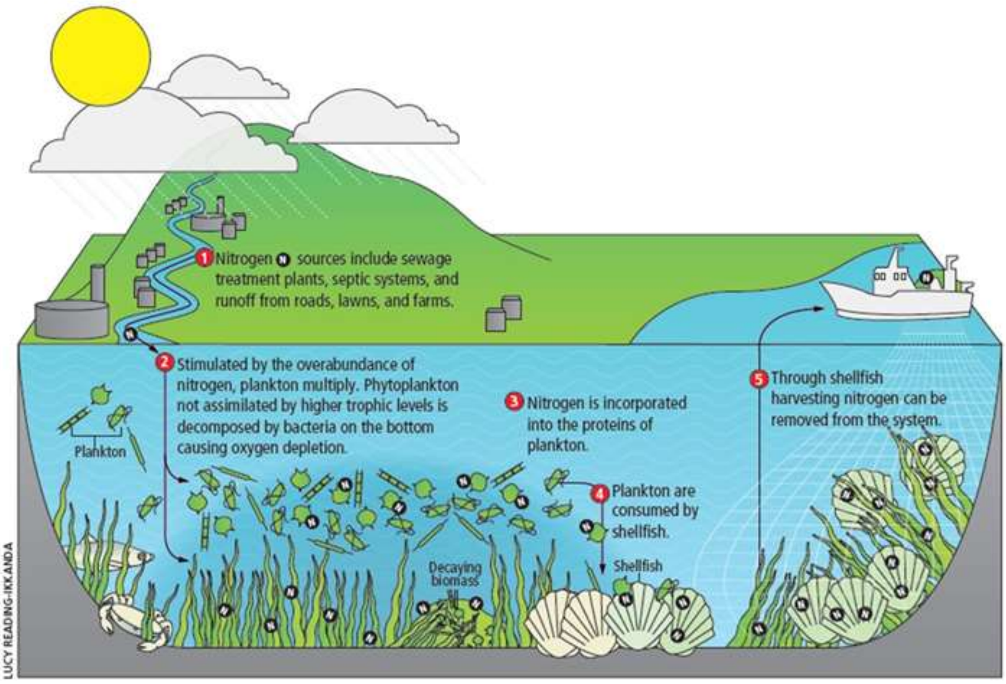


Figure 3: modified from Long Island Sound Study[1], 2010.

Nutrient credit trading programs are designed to fulfil water quality goals in the most practical and cost-effective way possible. In a well-developed nutrient credit trading program, shellfish growers

[1] https://longislandsoundstudy.net/our-vision-and-plan/clean-waters-and-healthy-watersheds/nutrient-bioextraction-overview/

should be compensated for the value of the removed nutrients and their consequences, i.e. excess of phytoplankton and organic detritus, which would otherwise need to be addressed by means of alternate management measures.

Can bioextraction be implemented as nutrient management policy?

The bioextractive capacity of bivalve shellfish is a key regulatory ecosystem service that can contribute to eutrophication reduction, but to date it has not been included as part of an EU nutrient management framework. Bivalve shellfish are being used by several US jurisdictions to help improve water quality and at the same time, increase domestic production of seafood. Nutrient credit trading has been proposed in the US, and already implemented in some states, as a tool to achieve water quality goals. These programs establish a market-based approach to provide economic incentives for achieving nutrient load reductions to meet pollution reduction targets. They could create new revenue opportunities for shellfish farmers, entrepreneurs, and others who are able to reduce discharges below the allowable levels at low cost and sell credits received to dischargers facing higher-cost reduction options.

In Chesapeake Bay, USA, oyster growers have been included in a nutrient credit trading program. The N removed by farmed oysters is credited toward fulfilment of required N reductions and growers can be compensated for the removal service provided. Oyster related N removal (tissue only) can be credited within the Maryland Chesapeake Bay Nutrient Credit Trading Program and growers have already received payment for N removal services.

In the USA, the Connecticut Nitrogen Credit Exchange was created in 2002 to improve nutrient-related hypoxia conditions in Long Island Sound. The program provided an alternative compliance mechanism for seventy-nine wastewater treatment plants throughout the state. Between 2002-2009, 15.5 x 10^6 N credits were exchanged at a value of US$46 million, with estimated cost savings of US$300-400 million. The trading of point sources was successful but non-point sources are not yet included. These results provide a basis for inclusion of oyster aquaculture into the trading program as has been done in Chesapeake region.

In order to develop robust policy measures such as a nutrient credit trading program, appropriate datasets are required to estimate, model, and analyse potential policy impacts. Policies that aim to promote aquaculture production also benefit from high quality datasets. Consumer demand for more and better seafood traceability is also a driver for improved seafood datasets and development of innovative methodologies for production, trade, and consumption estimates. Importantly, nutrient removal by shellfish farms must be quantified and a framework established to validate and apply nutrient credits in a consistent manner. This includes determination of the economic value that is assigned to the removed nutrient that could be compensation to the growers for the water quality ecosystem service being provided by their shellfish. Finally, a framework must be established to integrate shellfish nutrient removal into the larger nutrient management program.

What We Did

Using lab analyses, models, and previous work we estimated the amount of N that can be removed by bivalves individually and within a farm, the total EU bivalve harvest of the top 5 EU commercial species (blue mussel, Mediterranean mussel, Pacific oyster, Manila clam and flat oyster), the total N load to EU seas, and the economic value of the removed N.

How much nitrogen can bivalve shellfish aquaculture remove?

Filter feeding bivalves have complex interactions with their environment. The impact of these interactions depends on a number of factors, including farming practice (stocking density of bivalves, husbandry practice, etc.), physical conditions such as temperature and salinity, and water quality, particularly insofar as it determines the food supply.

Determination of the amount of nutrient removed can be estimated in two ways; 1) by measuring the nutrient content in the tissue and shell of a single bivalve shellfish and multiplying by the number of shellfish harvested; and 2) modelling the net mass balance of nutrients (intake minus loss) by the entire bivalve shellfish population of an aquaculture farm. In this study we used both methods to provide a range of potential N removal estimates; both were scaled up for the whole of the European Union by using production values for EU shellfish producing nations.

The proximate analysis is straightforward: tissue and shell are analysed in the lab to determine nutrient content of a mussel or oyster. Modelling the interaction of bivalves with the ecosystem is more complex and the physiological responses and growth at the individual level must be considered. The main objective of such models is to represent growth in individual biomass based on physical and biogeochemical determinants, as well as simulating environmental interactions. The individual shellfish models used in this work are based on a generic modelling framework called AquaShell™ that has been developed and parameterized for a range of species and validated at different locations worldwide. The individual shellfish models are then incorporated into the well-tested Farm Aquaculture Resource Management (FARM) model to determine both production and environmental effects at the farm-scale. FARM simulates bivalve production over the culture cycle and calculates phytoplankton and detritus removal by shellfish due to feeding. The mass balance (intake minus loss) provides an estimate for net removal of N from the water column by the farmed shellfish, which effectively equates to a drawdown of phytoplankton and detritus, primary and secondary symptoms of eutrophication.

What is the bivalve aquaculture production in Europe?

Bivalve production data was sourced from the META[2] and Eurostat[3] websites. The production volumes and economic values for the selected years and species were compared to determine production trends and variations in production between 2016 and 2018. A short economic analysis was also conducted for these bivalve products. This was done for each species and the main producing countries were identified.

What is the nutrient loading to European seas?

Previously determined nutrient loading and load source-apportionment was expanded and improved to include more detailed and up-to-date loading data for different European regional seas. The area covered by each of the regional European seas was used to calculate the normalized N loading per unit of surface area (kg km^{-2} y^{-1}). Where possible, we have separated the N sources into point-sources, land-based diffuse sources, and atmospheric deposition (Figure 1).

What is the economic value of nutrient bioextraction from European seas?

The economic value of the nutrient removal by shellfish farms can be estimated by assuming it is a water cleaning service that if not performed by the bivalves would have to be done by an alternate nutrient management measure such as wastewater treatment or agricultural or urban best management practices (BMPs). This is called an 'avoided' or 'replacement' cost approach, whereby the value of the N removed by bivalves can be determined by using costs for other management strategies to remove one kg of N (€ kg^{-1} N). We apply that cost to the amount of removed N to estimate the cost savings represented by bivalve bioextraction. Additionally, this is one way to estimate the potential compensation that would be provided to the grower in a nutrient credit trading program, though typically payments are less than avoided cost values. Here we use costs for wastewater treatment (€7047 kg^{-1} N), urban stormwater controls (€3888 kg^{-1} N), and for approved agricultural BMPs (€435 kg^{-}

2 https://longline.co.uk/meta

3 https://ec.europa.eu/eurostat

[1] N) to estimate the value of the N removed by bivalves.

What We Found

Nitrogen loading to European seas

Our study reported on both nitrogen and phosphorus but here we focus on N because it is typically the nutrient that causes eutrophication development in marine waters. According to current estimates, the EU waters that receive the highest N discharges are the Greater North Sea (1 500 × 10^3 tonnes y^{-1}), Mediterranean Sea[4] (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 N loadings normalised to surface area of each European Sea are shown in Table 1. The N loading estimate to all European seas including N. African discharges is 731 kg km^{-2} y^{-1} for N but is reduced to 641 kg km^{-2} y^{-1} if only European discharges are considered.

Table 1. Nitrogen loading to European seas (10^3 tonnes y^{-1}) and nutrient loading per unit area (kg km^{-2} y^{-1}). Nutrient loads include (i) point sources, such as wastewater treatment plants (WWTPs); (ii) diffuse sources, such as agricultural and urban run-off; and (iii) atmospheric deposition from fossil fuel combustion in power plants and vehicles.

	Total N (kt y^{-1})	N per unit area (kg km^{-2} y^{-1})
Norwegian Sea	64.7	58.3
Barents Sea	11	6.90
Baltic Sea	826	2 096
Greater North Sea	1 500	1 957
Celtic Sea[5]	275	750
Bay of Biscay and Iberian Coast	450	835
Black Sea	700	1 517
Mediterranean Sea – only Europe	950	377
Mediterranean Sea - Europe and N. Africa	1 845	733
Total including North Africa	5 672	731
Total incl. N. Africa & excluding Black Sea[6]	4 972	641

In most European seas the main load of N is diffuse which are more difficult to address than point source loads (Figure 1). An exception to this is the Norwegian Sea, due to high N loadings from salmon and trout aquaculture in Norway that are considered as point-source inputs. The N discharges from Norwegian finfish aquaculture affect both the Norwegian and the Barents Seas, accounting for 49% and 35% of the N loading to those Seas, respectively.

Bivalve production and value in Europe

The five species selected for inclusion in this analysis (blue mussel, Mediterranean mussel, Pacific oyster, Manila clam and flat oyster) account for over 95% of shellfish farm production in the EU, and 40% to 45% of all farmed aquatic organisms.

Table 2. Shellfish production (tonnes) and production value (k€) in the EU in 2018 for five key species (left to right): Mytilus galloprovincialis, Mytilus edulis, Magallana gigas, Ruditapes philippinarum and Ostrea edulis.

EU Production totals (t)	Mediterranean mussel	Blue mussel	Pacific oyster	Manila clam	Flat oyster	Total
2018	334 685	122 552	106 254	32 743	1 631	581 179
EU Production value (k€)	**Mediterranean mussel**	**Blue mussel**	**Pacific oyster**	**Manila clam**	**Flat oyster**	**Total**
2018	205 957	177 180	429 471	174 538	12 536	973 818

[4] 950 kt y^{-1} is the N loading from European countries to the Mediterranean Sea, excluding North African discharges. If we consider N discharges from N. Africa this value doubles to 1,845 kt y^{-1} and thus the Mediterranean Sea would be the main contributor to N loading.

[5] Previous literature research shows only the discharges to the Bay of Biscay, the North Sea, and the Celtic Seas as a whole.

[6] Included to allow comparison with the loading estimates from the previous research.

Overall EU bivalve production increased by 5.4% since 2016: EU shellfish production is growing. The largest producer countries are Spain and France, and the highest volume is Mediterranean mussel in Spain and Pacific oyster in France. The EU produced over 580 000 tonnes of bivalve shellfish in 2018 with a total economic value above €970 million (Table 2).

Pacific oysters are the most valuable shellfish species, with higher value than both mussel species combined (Table 2). Although oysters are more valuable, both oyster and mussel consumer markets were negatively impacted during the COVID-19 pandemic. In France, the drop of demand caused a decrease in oyster prices and an increase in risks of pathogen spread in overstocked farms. The farmed mussel sector suffered an 80% decrease in sales and drop in prices, leading to financial problems in several companies.

Bioextraction of nitrogen by cultivated shellfish in European seas

Nitrogen removal estimates based on both the proximate and modelling approaches, coupled with the detailed shellfish production volumes for the top 5 commercial species allowed a calculation of the total N removed in 2018 by the main shellfish producing countries. Total N removal by the combined five species was 4 922 – 13 425 tonnes (Table 3). To put into a different perspective, the removal is equivalent to water treatment for 1.5 – 4.1 million people.

Table 3. N removed for each species in the key EU producer countries, estimated using the proximate analysis and FARM model approaches. The lower values are based on the proximate analysis estimation. Volumes shown in tonnes for 2018.

	Nitrogen removed in EU in 2018 (tonnes)					
	Blue mussel	Med. mussel	Pacific oyster	Flat oyster	Manila clam	Total
UK	125 – 272					125 – 272
Netherlands	398 – 868			0.9 – 6.3		399 – 875
France	386 – 842	47 – 113	339 – 2 080	1.9 – 12.6	2.8 – 16.9	777 – 3 065
Ireland	122 – 265		31 – 191	0.7 – 4.9		153 – 461
Spain		2 426 – 5 809		1.0 – 6.6	2.8 – 16.6	2 430 – 5 833
Italy		614 – 1 470			99 – 592	713 – 2 062
Others	42 – 93	258 – 618	24 – 146	0.3 – 1.8		325 – 858
Total EU-28 production	1 073 – 2 340	3 346 – 8 010	393 – 2 417	4.8 – 32.2	105 – 625	4 922 – 13 425

The bulk of N removal is due to mussel production: an average of 64% of the total removal is attributed to Mediterranean mussel harvest and 20% to blue mussel harvest. Spain is responsible for the highest removal, the high production volume of Mediterranean mussel accounts for an average of 46% of the total N removed. France has the second highest value of N removal (average 19% of total), due mainly to its production of blue mussels and Pacific oysters. Italy, the main clam producer, has the third highest removal, average of 15%, due mainly to the additional >61 000 tonnes harvest of Mediterranean mussel.

The total N inputs (tonnes) for each regional sea are shown in Table 4, the average credit valuation (millions of €) for N removal by shellfish is shown, based on the proximate and modelling approaches.

Table 4. Nitrogen loading in European seas and the potential economic benefits on each sea. Nitrogen removal credits calculated based on the country of production, according to proximate analysis and FARM modelling approaches.

European Seas	Nitrogen loading (tonnes)	Average credit valuation (millions of €)
Norwegian Sea	64 600	-
Barents Sea	11 010	-
Baltic Sea[7]	601 900	-
Greater North Sea[7]	1 000 000	1 898 – 4 155
Celtic Seas	275 000	556 – 1 670
Bay of Biscay and Iberian Coast	450 000	8 805 – 21 134
Black Sea	700 000	-
Mediterranean Sea - only Europe	950 000	5 398 – 18 574
Total	4 052 610	17 833 – 48 642

[7] Atmospheric deposition values were not included in loads used to estimate N removal, because those data were available only for the Baltic and Greater North. Thus, N removal estimates could be overestimated in locations where atmospheric deposition is a large part of the total load.

The economic value of nitrogen bioextraction from European seas

Production of the five bivalve species surpassed 580 000 tonnes in 2018, and the estimated N removed by their harvest from European seas ranged from 4 922 to 13 425 tonnes, depending on the assessment method used. The economic value of the N removed by bivalve bioextraction was estimated using an avoided or replacement costs approach and ranges from 2 132 to 94 604 million euros.

In financial terms, the benefits of incorporating cultivated shellfish into a catchment-scale nutrient management scheme could be significant (Table 5). If we consider only the lower estimate of N removal by shellfish, the average overall value of N removal totals almost €18 billion. The maximum estimates are considerably higher, and it is recognised that the potential valuation is associated with nutrient loading that is for the most part diffuse and is therefore challenging to reduce in many rural areas in Europe without severe social consequences.

Table 5. Financial benefits of an EU-wide nutrient credit trading framework to include shellfish farmers. Remediation costs of stormwater control, agricultural best management practices, and wastewater treatment are from the literature.

Nitrogen removal		
Nitrogen removed by shellfish (tonnes per year)		4 922 – 13 425
Population-Equivalents (106 PEQ @ 3.3 kg N per ind.)		1.49 – 4.07
Value of eco-intensification	**Remediation cost (€ kg^{-1} N)**	**Credit valuation (Million €)**
Stormwater control measures	3 388	16 674 – 45 483
Approved agricultural BMP	435	2 141 – 5 840
Wastewater treatment upgrades	7 047	34 683 – 94 604
Average credit valuation (millions of €)		17 833 – 48 642

What can case studies teach us?

We summarize two bioextraction case studies from Europe and the United States to illustrate how bivalve shellfish modelling was used to facilitate consideration of bivalve shellfish inclusion in comprehensive nutrient management programs. Common to both locations is that they support bivalve aquaculture and have high nutrient loads to the waterbody, leading to moderate to high levels of eutrophication that require additional nutrient management measures to remediate and protect the waterbody from further degradation.

Chesapeake Bay, USA – Up-scaled farm-scale model results used in approval of oyster tissue BMP

In Chesapeake Bay the local-scale FARM production model was used to simulate farm-scale oyster production and associated N removal via assimilation into tissue and shell. The focus is on N because it is, globally, most often the limiting nutrient in estuarine waters and has been the focus of coastal nutrient management. An avoided or replacement costs economic analysis was used to estimate the value of the removed N.

Key findings/lessons

Based on model results, the average N content of an oyster is 0.9 g for the 6 farms tested. This value was upscaled to the total number of oysters harvested in Chesapeake Bay to estimate the potential N removal by oyster aquaculture for all of Chesapeake Bay. Oysters removed 60 tons N per year, equivalent to 0.052% of total N inputs. While seemingly small, this removal rate compares favourably to reported effectiveness of approved agricultural best management practices (BMPs). Additionally, this does not include denitrification losses, which can be significant, and thus this can be considered a conservative estimate.

Alternative management costs specific to Chesapeake Bay were used to assign value to oyster related N removal: improvements to wastewater treatment plants at three treatment levels (8, 5, and 3 mg L^{-1}, \$35 - \$104 kg^{-1} year^{-1}), agricultural BMPs (\$7 - \$1034 kg^{-1} year^{-1}), and urban stormwater BMPs (\$66 - \$4873 kg^{-1} year^{-1}). We compared results to the 2017 value for N credits in the VA Nutrient Credit

Exchange program ($8.33 credit^{-1}). The range of values for oyster aquaculture related N removal is $0.132 to $292 million year^{-1} which compare to the VCNEA based value of $0.500 million per year. This illustrates that the avoided costs or 'costs saved' are typically greater than the costs paid to growers, which are determined by the market.

Notably, some of these data were used in the Chesapeake Bay Program Oyster Best Management Practice Panel evaluation that led to approval of harvested oyster tissue as a nutrient BMP for Chesapeake Bay in 2016. In this program, harvest of one million diploid 3-inch oysters receives credit for 198 lbs (90 kg) of N removed, and 287 lbs (130 kg) N removed for triploids, to count toward fulfilment of US Environmental Protection Agency mandated nutrient reductions.

Dundrum Bay, Northern Ireland—modelling shows that bivalves reduce chlorophyll

Similar to the analysis of potential capacity for oyster aquaculture associated nitrogen removal in Chesapeake Bay (see case study above), in Dundrum Bay, Northern Ireland, we applied an ecosystem level modelling framework that includes integration of estimated nutrient discharge from the catchment, water circulation), and bivalve growth and associated nutrient removal. The aim was to analyse the role of bivalves in control of eutrophication symptoms, using chlorophyll and particulate matter as indicators. The results will be used to inform the development of a bivalve aquaculture related nutrient removal best management practice and, more broadly to support development of a nutrient credit trading program within the waterbody that could be applied and developed in other/ all EU waterbodies within a national program.

Key findings/lessons learned

The bay-scale evaluation of chlorophyll reductions —one of the primary symptoms of eutrophication— attributed to farmed shellfish can only be done by means of an ecosystem model. Local-scale models such as FARM can determine food depletion within a farm, but they cannot predict what the resulting effect will be at the full bay scale. Since phytoplankton abundance, biomass, and composition is one of the biological quality elements in the WFD, and since both abundance and biomass are usually represented by chlorophyll as a proxy, system-scale models are a valuable management tool for considering different scenarios for eutrophication management.

Shellfish cultivation reduce extreme chlorophyll concentrations within the system. The typical chlorophyll maximum increases from 3.8 to 5.1% when we remove the shellfish from the system, depending on the location. These results suggest that the effect of bivalve filtration related removal of phytoplankton occurs in a broader area of the bay, since the benthic (bottom) filter-feeders are removing food from the water passing through the cultivation sites. Changes to cultivation practice will thus be reflected in a more general way on bay-scale eutrophication.

Impacts of this study

This study showed that the bivalve aquaculture associated N reductions in the EU are significant and should be considered for inclusion in a comprehensive nutrient management strategy. The major challenge to the use of bivalves for nutrient management is the determination of the quantity of nutrients that are removed by shellfish species. The range of environments and the diversity of bivalve culture practices is also a challenge in estimating nutrient removal, although bivalve aquaculture in European coastal ecosystems is well developed and growing, often with significant production volumes.

Results show that different shellfish species have different N removal rates. Location and farm culture practices can also have a role to play in removal rates. The Mediterranean mussel, mainly farmed in Spain, had the highest nitrogen removal. The total N removed (4 922 – 13 425 tonnes) is equivalent to water treatment for 1.5 – 4.1 million people and is worth €18 – €48 billion.

Nutrient management at the catchment scale is consistent with other policy instruments such as the

WFD, which aim to manage watersheds in an integrated manner across the various types of waterbodies. Reduction of eutrophication through installation of shellfish aquaculture is recognised in qualitative terms but there has been no associated policy development at a European or national level. These results with the growing body of work supporting this concept and lessons learned from locations where bivalve shellfish are included as nutrient management tools should help to further the discussion of implementing this tool in the EU.

Services provided by shellfish are not limited to nutrient removal: there are other major societal benefits including greater food security, local employment and cleaner waters, beneficial for local populations and for tourism—shellfish are the quintessential nature-based solution.

Bivalve 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.

Roadmap for a bioextraction nutrient management program

- Form an expert panel on Best Management Practice to review and evaluate studies in support (or not) of the potential to include shellfish ecosystem services in existing and future nutrient credit trading programs, including nutrient credits for nutrients removed from the ecosystem in both tissue and shell via harvest. This group will develop protocols for determination of the N credits that will apply for various species, validation protocols, etc;
- Use different approaches such as those proposed herein, i.e. elementary analysis or modelling estimates, to quantify the average N removed by each harvestable-sized shellfish species and recommend to the panel determining how shellfish can be added to existing nutrient management programs;
- Upscale the N removal per individual to the total number of oysters harvested at the system-scale to estimate the potential N removal by shellfish aquaculture within the analysed waterbody;
- Determine the value of the bioextractive nitrogen removal to evaluate potential economic benefits of the shellfish aquaculture ecosystem service. An avoided or replacement costs economic analysis can be used to estimate the value of the removed N. The costs to remove one kg of N by alternate reduction strategies are typically used to provide estimates of the economic value of the removed N, giving a range of potential values. Typically the least cost option will be used to determine the value of bioextracted N;
- There are additional ecosystem services provided by shellfish that are currently not valued, such as denitrification N losses and increased water transparency, that can lead to valuable improvements to water quality such as increased seagrass and habitat, and improved bottom water dissolved oxygen. Thus, the economic value of the shellfish ecosystem service is likely to be much greater than that proposed above. Given the low trophic level that bivalve shellfish represent, their potential value as a food source, and the range of ecosystem services they supply, this group of cultivated organisms is emblematic as a nature-based solution;
- The valuation of the bioextraction ecosystem services associated with shellfish culture will enhance public awareness of water quality issues and could help shift attitudes to allow increased opportunities for shellfish aquaculture and stimulate local economies.