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Review Paper

# Sustainable growth of non-fed aquaculture can generate valuable ecosystem benefits

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ABSTRACT

Investment in extractive or 'non-fed' aquaculture has been proposed as a partial solution for sustainable food provision. An important aspect is the potential for aquaculture-environment interactions to influence the provision of ecosystem services. Here, we quantify and monetise the impacts of bivalve and seaweed farming on a regulating service (removal of nitrogen from nearshore waters) and a supporting service (habitat provision for species with fisheries value). We estimate that on average, 275–581 kg N ha$^{-1}$ yr$^{-1}$ (in harvest units: 4–25 kg N t$^{-1}$) is removed via bioextraction at oyster, mussel and seaweed farms, with much smaller contributions from enhanced sediment denitrification beneath farms compared to reference sites. Based on nitrogen offset values in the United States and Europe, this additional nitrogen removal could be worth 84–505 USD t$^{-1}$ in locations where nutrients are a management priority. Additionally, the habitat structure offered by aquaculture is estimated to support 348–1110 kg ha$^{-1}$ yr$^{-1}$ of additional fish compared to reference habitats, potentially worth an additional 972–2504 USD ha$^{-1}$ yr$^{-1}$ to commercial fishers or 1087–2848 USD ha$^{-1}$ yr$^{-1}$ to recreational fishers. Habitat values assume equal mortality rates at farms and comparable natural habitats, although the direction of effect is robust to small increases in mortality at farms. New policy perspectives may improve the capacity of non-fed aquaculture to sustainably meet the increasing demand for food while enhancing the provision of these two ecosystem services. Responsible development will be crucial to ensure that ecological benefits are not eroded by suboptimal site selection or farming practices that diminish the same or other ecosystem services.

## 1. Introduction

Healthy ecosystems provide services that benefit society. In the coastal marine environment, ecosystem services (ES) can include support for fisheries, amelioration of nutrient inputs, coastal defence, and a range of other non-extractive uses (Costanza et al., 2014). However, coastal ecosystems worldwide face an accumulation of environmental stressors including habitat loss, overfishing, and eutrophication, all of which are gradually eroding the productivity and resilience of marine environments (Halpern et al., 2019, Halpern et al., 2008). The estimated value of lost ES is in the trillions of USD (Costanza et al., 2014), and has left coastal communities searching for cost-effective solutions to restore ecosystem function and livelihoods. This coincides with the need to provide nutrition for a growing global population without exceeding environmental limits (Springmann et al., 2018).

Strategic development of marine farming (aquaculture, or specifically, mariculture) using ecological principles will play an important role in supporting sustainable food delivery, together with established strategies such as fisheries management and conservation/restoration efforts. Aquaculture is a diverse industry with a range of social and environmental interactions, and negative environmental outcomes, especially from 'fed' aquaculture of finfish and shrimp, have been well documented (Dierberg and Kiattisimkul, 1996; Taranger et al., 2015). However, there is growing evidence that 'non-fed' aquaculture of

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bivalves and seaweeds—taxa which do not require feed inputs—can enhance certain ES (Alleway et al., 2019; Brugère et al., 2019; Carranza and Zu Ermgassen, 2020; Costa-Pierce, 2010; Gentry et al., 2019; Hehre and Meeuwig, 2016; Kim et al., 2017; Lindahl et al., 2005; Petersen et al., 2014; Petersen and Taylor, 2020; Petrolia et al., 2020; Rose et al., 2014; Smaal et al., 2018; van der Schatte Olivier et al., 2020). A recent review noted that potential benefits of aquaculture typically do not receive fair coverage in the literature (Weitzman, 2019).

Undersupply of shellfish makes up around 30 % of a projected 28 million t seafood supply gap (Cai and Leung, 2017; Costello et al., 2020; FAO Fisheries and Aquaculture Department, 2020a), and active investment of 150–300 billion USD is needed to guide the growth trajectory of aquaculture toward the most sustainable activities (O’Shea et al., 2019). This coincides with a trend of institutional investment in ventures that are expected to yield social and environmental benefits as well as profits, termed ‘impact investing’ (Bugg-Levine and Emerson, 2011). However, the capacity of impact investment to support sustainable aquaculture is hindered by policies that view aquaculture solely as an environmental impact to be managed through risk assessment (Arthur et al., 2009; Fletcher et al., 2004; Fletcher, 2015). This policy perspective, together with other economic and social factors (e.g. Beckensteiner et al., 2020), currently places a greater constraint on the development of this industry than biophysical parameters (Costello et al., 2020; Ruff et al., 2020).

There is evidence that the transition to a ‘blue economy’ is failing to deliver sustainable outcomes (Nahuelhual et al., 2019). This is partly because most growth has occurred in finfish rather than non-fed aquaculture (FAO Fisheries and Aquaculture Department, 2020), and is partly a result of unsuitable siting, intensive clustering of farming activity, and insufficient biosecurity measures (Nahuelhual et al., 2019). However, a more holistic view of the role of aquaculture may help to prioritise non-fed aquaculture in areas where it can enhance ES without exceeding the local carrying capacity. One approach is to assign monetary values to ES, creating an economic argument for investment into nature, and a standardised method to assess the benefits and costs that arise from aquaculture-environment interactions (Costanza et al., 2014; Liquete et al., 2013; Worm et al., 2006). Several recent syntheses have considered the economic value of ES provided by bivalve aquaculture, primarily focusing on regulating services such as nutrient removal (Ferreira and Bricker, 2016; Gentry et al., 2019; Petersen et al., 2014; Petrolia et al., 2020; Rose et al., 2015; Smaal et al., 2018; van der Schatte Olivier et al., 2020). However, there has been comparatively little quantification of regulating services provided by seaweed farming (Alleway et al., 2019; Chopin and Tacon, 2020; Gentry et al., 2019; Kim et al., 2017). Moreover, while some valuations have discussed supporting services such as habitat provision (Petrolia et al., 2020; van der Schatte Olivier et al., 2020) and shoreline erosion protection (Grabowski et al., 2012; Petrolia et al., 2020; Sutton-Grier et al., 2015), there has not yet been an attempt to value such services at regional or global scales. Recent syntheses have considered the types of interactions that occur between wildlife and aquaculture habitat (Barrett et al., 2019; Callier et al., 2018, Theuerkauf et al. 2021), but without an assessment of the economic outcomes from such interactions.

Here, we quantify the impacts of global bivalve and seaweed aquaculture on ES provision. Our analysis is limited to two relatively tractable examples of ES that are expected to be influenced by non-fed aquaculture, specifically (i) removal of nitrogen from nearshore waters, and (ii) habitat provision for species of value to fisheries. We quantify effects of farm activities on of these ES by aggregating effect sizes from published control-impact studies, and then apply monetary values to those effects by synthesising available market prices and non-market valuations in the literature. In doing so, we provide a framework that can be updated and expanded to include other ES as more data become available.

## 2. Aquaculture-environment interactions and ecosystem service provision

Like all farming activities, non-fed aquaculture increases provision of some ES but trades off against others. Social and economic trade-offs (especially between competing spatial uses) have been the major factor limiting expansion of aquaculture globally, more so than biophysical parameters (Sanchez-Jerez et al., 2016; Gentry et al., 2017a; Oyinlola et al., 2018; Theuerkauf et al., 2019; Beckensteiner et al., 2020; Galparsoro et al., 2020). Complex aquaculture-environment interactions also result in a range of trade-offs between various ES, wherein some services are enhanced while others are diminished.

### 2.1. *Potential benefits to ecosystem service provision*

#### 2.1.1. *Provisioning and cultural services*

The most direct ES provided by bivalve and seaweed aquaculture is the provision of nutrient-rich seafood and other commercially valuable products that are less impactful than most alternatives in terms of carbon intensity and habitat loss (Hilborn et al., 2018; Ray et al., 2019; Williamson et al., 2015). In some regions, even waste shell is a valuable commodity (van der Schatte Olivier et al., 2020). The production of sustainable local seafood also provides cultural services in the form of economic opportunities (including a range of value-adding ecotourism and farm-to-table food tourism opportunities depending on social attitudes to specific forms of aquaculture (Chand et al., 2015; Kim et al., 2017), and strengthened connections between communities and local environments (Michaelis et al., 2020; Sandifer et al., 2015). Provisioning and cultural services have been the focus of several recent reviews and are not further evaluated here (Alleway et al., 2019; Gentry et al., 2019; Smaal et al., 2018; van der Schatte Olivier et al., 2020).

#### 2.1.2. *Regulating services*

Bivalves and seaweeds are influential regulators of the marine environment. Seaweeds take up carbon, nitrogen and phosphorus from the water column (Roleda and Hurd, 2019), and in estuaries and bays affected by anthropogenic eutrophication (Malone and Newton, 2020), harvesting of seaweed biomass (‘bioextraction’) has the potential to remove significant amounts of nitrogen and phosphorus from the system (Xiao et al., 2017). Carbon sequestration by seaweeds can also occur where the use of the product is controlled (Chung et al., 2013; Froehlich et al., 2019). Bivalves, as filter-feeders, remove particulate matter from the water column and assimilate the nutrients contained in planktonic biomass and detritus, while also reducing concentrations of bioavailable nitrogen in the water column via biodeposition and denitrification (conversion of bioavailable nitrogen to non-bioavailable gases) or sequestration in the sediment (Smaal et al., 2018). Many, but not all (Anaïs et al., 2020), human-impacted inshore waters have elevated nutrient loads and planktonic biomass due to anthropogenic nutrient inputs and/or historical declines in bivalve populations, and thus have scope for improvement of water quality via bivalve and seaweed farming (Beck et al., 2011; Byron et al., 2011; Nielsen et al., 2016; Theuerkauf et al., 2019). The physical structure of seaweed and bivalves also serves to stabilise sediments, attenuate waves and reduce current velocity. This can provide measurable defence against erosion and storm surge in some situations, while slowed water movement, together with sediment stabilisation, reduces suspension of sediments (Morris et al., 2018; Spalding et al., 2014). Organic carbon sequestration has been proposed as a possible regulating service, but it is not yet clear whether bivalves are a carbon source or sink because the shell formation process is a net producer of $CO_2$ (Fodrie et al., 2017).

#### 2.1.3. *Supporting services*

Aquaculture provides complex 3-dimensional habitat structure that, in many systems, is associated with abundant and diverse populations of wild fauna both within and around the farm footprint (Barrett et al., 2019; Callier et al., 2018; Costa-Pierce and Bridger, 2002; Smaal et al., 2018). In general, benefits or costs to supporting services will be determined by local context and habitat quality (fitness of individuals that use farm habitats). Given that farms appear to be attractive habitats for many animals, high mortality risk or other factors could cause farms to act as ecological traps rather than productive habitats for susceptible species (Barrett et al., 2019; Swearer et al., 2021). The role of farm structures may be most beneficial in areas where structured habitats such as seagrass and shellfish reefs have historically been lost (Beck et al., 2011; Orth et al., 2006). In bivalve and seaweed aquaculture, habitat provision for wild fauna arises from high densities of farmed organisms (i.e. bivalve shells and seaweed foliage), as well as farm infrastructure. Farm infrastructure varies according to farmed taxa and location, including rafts or longlines, stakes or poles, cages/baskets/bags (either on- or off-bottom), or in the simplest form, bivalves and seaweeds may be seeded directly onto the seabed, with or without protective gear. While bivalves and seaweeds do not require feed inputs, the farmed biomass and presence of aquaculture gear can also increase food availability for wildlife (DeAlteris et al., 2004; Sardenne et al., 2019). Farmed seaweeds are an obvious food source for grazers (Hehre and Meeuwig, 2016; Yang et al., 2015), while various invertivores have been observed consuming farmed bivalves either directly from the lines (Šegvić-Bubić et al., 2011; Žydelis et al., 2009) or upon falling to the seabed (D'Amours et al., 2008; Inglis and Gust, 2003; Sardenne et al., 2019). Both bivalve and seaweed farms also support abundant biofouling that provides a food source for browsing herbivores and invertivores, which in turn are prey for the larger invertivores and piscivores that are sometimes found at farms (Barrett et al., 2019; Callier et al., 2018). Finally, aquaculture may improve habitat quality for wildlife by excluding more damaging activities such as bottom trawling (Gristina et al., 2017).

## 2.2. *Potential costly mechanisms*

### 2.2.1. *Degradation of benthic habitat within the farm footprint*

Non-fed aquaculture can have opposing effects on near- and far-field benthic communities (Weitzman et al., 2019). This is most apparent at bivalve farms, where the filter-feeders remove organic matter from the water column and deposit it on the seabed. This nutrient concentrating process promotes sediment hypoxia when stocking densities are high and/or water exchange is low, and combined with shading by farm structures, can shift the benthic community from one dominated by filter-feeders and photosynthesisers to one dominated by detritivores (Tallis et al., 2009; McKindsey et al., 2011; Ferriss et al. 2019). Maintenance and harvesting activities can also disturb resident populations (Spencer et al. 1998), and as a result, aquaculture can reduce habitat quality for a range of benthic organisms. However, such impacts not universal, even in the presence of intensive disturbance (e.g. Goldberg et al., 2014), and where they do occur, near-field impacts on benthic communities may be offset by far-field effects, as filter feeding by farmed bivalves improves light penetration and reduces organic sedimentation over a wide area (Guyondet et al., 2015; Schröder et al., 2014). Consideration of pre-existing habitat value is essential for the realisation of net benefits. Siting farms over seagrass or reef habitats, for example, can deliver a net cost to fisheries enhancement if the original habitat is more productive than the farm habitat that replaces it. Also, some habitats have intrinsic value that may not be represented by the biomass of fisheries targets (e.g. broadly unproductive habitats that are critical for a particular species of conservation concern). Responsible approval processes must consider all such information.

### 2.2.2. *Food depletion and larval capture*

The filter feeding activity of farmed bivalves is a key driver of environmental benefits, but also presents some potential costs. For example, farmed bivalves compete with wild populations for suspended food and consume the planktonic larvae of numerous fish and invertebrate species, while wild spat collectors and farm structures themselves also capture or attract wild larvae that might have otherwise settled into natural habitats. These processes can combine to impact the performance and recruitment of wild populations in areas with very high densities of farmed or wild filter feeders, especially when non-selective spat collection is employed (Molinet et al., 2017; Smaal et al., 2013). These processes are difficult to value as they are both complex and context-dependent (e.g. status of wild populations, farming and spat collection methods). To minimise the costs of any such externalities, farmed biomass must remain within the local carrying capacity. The term 'carrying capacity' has at least two relevant definitions in this context: the production carrying capacity, at which harvests are optimised, and the ecological carrying capacity, at which ecological effects are deemed unacceptable (McKindsey et al., 2006). Biomass limits should be set based on the lower of the two, which is not always the ecological carrying capacity. This is because most nearshore marine waters have concentrations of suspended organic matter well above natural levels and some depletion is therefore considered acceptable or even desirable. Non-fed aquaculture is also highly sensitive to ambient food availability. Models and field data indicate that because of the high local density of bivalves within a farm footprint, the downstream portion of the farm tends to become food-limited even without ecosystem-wide food depletion (Strohmeier et al., 2005). In such cases, the interests of existing leaseholders and environmental advocates may be aligned with respect to permissible farm biomass. Either way, assessments of new lease applications must consider the potential for food depletion and larval capture.

### 2.2.3. *Biological invasions and genetic introgression*

Like most marine infrastructure, farms are vulnerable to biofouling by invasive species, and can act as stepping-stones that supply invasive propagules to downstream habitats (Mineur et al., 2012). Farming of non-native species or selectively bred variants also risks invasion or genetic introgression into populations of locally adapted relatives if farmed individuals are allowed to reproduce while at sea (Crego-Prieto et al., 2015; McKindsey et al., 2007). There have been numerous reports of species imported for aquaculture impacting surrounding ecosystems. For example, the red alga *Kappaphycus alvarezii* has been imported into numerous tropical countries for aquaculture, where it spreads to coral reefs (Bindu and Levine, 2011), while the Pacific oyster *Crassostrea gigas* has been introduced to at least 45 ecoregions, both unintentionally and for aquaculture (Molnar et al., 2008). The economic impacts of invasions originating from aquaculture are not well documented. InvaCost, a recent compilation of the economic costs of biological invasions, contains > 13000 estimates (Diagne et al., 2020). As of 2021, only 253 estimates concern bivalves or seaweeds, and of those, only 3 are relevant to aquaculture. All 3 concerned *C. gigas*. Estimated costs of invasive *C. gigas* in the Netherlands and Denmark were similar at 1.5 and 1.6 million USD $yr^{-1}$ for each country, respectively (Strandberg, 2017; van der Weijden et al., 2007). In Mexico, where the aquaculture-derived invasion is relatively recent and potentially controllable, the cost was estimated at only 7774 USD $yr^{-1}$ (Bonilla and Paez, 2019). To our knowledge, none of these analyses accounted for positive effects of *C. gigas*, which can be substantial in areas where native oyster populations have already declined and the invader performs similar functions (McAfee and Connell, 2020). In general, we expect that net costs of invasions by farmed species will depend on the competitiveness of the invader in its new environment and the relative value of the invader compared to impacted native populations. These will need to be calculated on a case-by-case basis.

### 2.2.4. *Displacement or mortality of marine birds and mammals*

Sensory disturbances and habitat alterations associated with non-fed aquaculture can displace some birds and mammals (Becker et al., 2011; Godet et al., 2009; Markowitz et al., 2004), while boat strikes and entanglements with certain types of gear can be lethal (Bedriñana-Romano et al., 2021; Price et al., 2017). Given the potential lifetime value (millions USD) of individual cetaceans in areas where whale or dolphin tourism is conducted (Cisneros-Montemayor et al., 2010; Wiener et al., 2020), any measurable impacts on cetaceans will quickly offset the ES benefits of non-fed aquaculture. Because reports of entanglements are rare in practice (Price et al., 2017), and the literature is equivocal on the prevalence of other negative impacts (Barrett et al., 2019; Callier et al., 2018; Díaz López and Methion, 2017), it is difficult to estimate costs at any scale. In jurisdictions where entanglements or other interactions are recorded and made public, it may be possible to estimate costs to ecotourism. The high consequence of any negative interaction makes it critical for aquaculture to be sited and constructed in ways that minimise impacts on marine birds and mammals.

### 2.2.5. *Disease transmission*

Pathogen spread is often facilitated by dense and highly connected populations. Oysters appear to be particularly disease-prone, with a range of viral, bacterial or parasitic threats present in oyster farming regions globally (King et al., 2019). Outbreaks are devastating for farmers, and have driven industries to near-extinction in areas where outbreaks occur frequently (King et al., 2019). However, it is not clear to what extent aquaculture is to blame for outbreaks of shellfish disease. Most diseases were likely endemic before the aquaculture, and mass shellfish mortality events regularly occur in areas without aquaculture, often linked to environmental stressors that allow latent pathogen populations to take hold. Where outbreaks do occur, a high biomass of farmed bivalves may well amplify the outbreak by increasing the density of infective particles in the water column. Conversely, there is some evidence that filter feeding by farmed bivalves can reduce disease risk for wild counterparts (Ben-Horin et al., 2018).

### 2.2.6. *Algal blooms*

High terrestrial nutrient inputs can lead to algal or cyanobacterial blooms that pose risks to ecosystems and human health (Grattan et al., 2016; Ye et al., 2011). Non-fed aquaculture is expected to downregulate harmful blooms by reducing ambient nutrient levels, but where blooms occur, toxin build-up can occur in both wild and farmed bivalves (Grattan et al., 2016). In some cases, the high biomass of farmed or biofouling macroalgae on farms can also trigger macroalgal blooms. Most notably, *Ulva prolifera* growing on nori (*Porphyra yezoensis*) farm structures likely provided the starting culture for 'green tides' impacting the Yellow Sea (Liu et al., 2009). The clean-up following an outbreak in Quingdao prior to the 2008 Olympic Games reportedly cost > 87 million USD in 2008 dollars (Wang et al., 2009). However, it is not clear how to apportion responsibility between terrestrial nutrient inputs, which fuelled the bloom, and the local aquaculture industry, which may have triggered it. Aquaculture-seeded algal blooms are also rare from a global perspective, and costs are highly specific to affected locations. Key recommendations for the Yellow Sea include reduced terrestrial nutrient inputs (Ye et al., 2011) and more hygienic biofouling management to prevent the release of large volumes of *U. prolifera* (Liu et al., 2009).

### 2.2.7. *Pollution and refuse disposal*

Shells are the major by-product of farming bivalves for food, yet shell disposal is unlikely to be a significant limitation. Landfill incurs relatively small costs (Yan and Chen, 2015), and a range of existing and potential uses for waste shell mean that landfill need not be used. These include agricultural feed/soil additives, construction materials, and substrate for reef restoration projects (Yan and Chen, 2015; van der Schatte Olivier et al., 2020). As shell formation creates acidity and shell degradation/dissolution creates alkalinity, it may be preferable to return waste shells to the sea to maintain acid balance and provide low-acidity benthic microhabitats (Waldbusser et al., 2013).

While non-fed aquaculture has low use of industrial chemicals, all aquaculture relies on plastic products that can contribute to plastic waste and debris (Hong et al., 2014). Such outcomes may be become less common in jurisdictions where relevant standards are regulated and enforced (e.g. South Australian Government 2020), yet challenges remain from a global perspective. No precise global estimates of costs to ES are available, but a recent review conjectured > 500 billion USD annually in lost value (Beaumont et al. 2019). Sources of marine plastic have also not been well partitioned, although terrestrial inputs account for the vast majority (4.8–12.7 Mt in 2010: Jambeck et al., 2015), followed by lost and abraded fishing gear, with relatively small contributions from aquaculture (Lusher et al., 2017). Yet given the potential costs of plastic pollution, all aquaculture sources require attention.

## 3. Methods

The two focal ES, nutrient removal and habitat provision, are evaluated individually using methodology specific to each (Fig. 1).

### 3.1. *Nutrient removal*

We use two approaches to quantify effects of aquaculture on nutrient removal services. The first is to obtain estimates of annual harvest volume and the nitrogen content of that harvest (bioextraction). Bioextraction is assumed to be independent from other nitrogen removal mechanisms and other nitrogen fluxes, such that the full nitrogen content of the harvest can be treated as 'additional' nitrogen removal, without making comparison to reference sites. The second is to compare rates of sediment denitrification at farms and reference sites (i.e. control-impact design). The difference between denitrification rates at farms and reference sites represents an effect of farms on denitrification, which can be a negative effect if denitrification rates are lower at farms than reference sites. The total impact of farming on nutrient removal is the sum of these two mechanisms (bioextraction and denitrification). We did not quantify biodeposition and/or burial rates due to a low number of estimates in the literature (Carlsson et al., 2012; Wei et al., 2019; Zhou et al., 2006) and uncertainty about the long-term fate of buried nitrogen.

### 3.1.1. *Literature search and data selection*

To discover literature relevant to the effect of bivalve or seaweed farms on nutrient fluxes, we searched ISI Web of Science and Google Scholar catalogues during June 2020 for any publications up until that time, using terms targeted at nitrogen removal by bioextraction (harvesting), enhanced biodeposition and denitrification (Table A.1, Suppl. Text A.1). We did not assess phosphorus, as nitrogen is more often the limiting nutrient for eutrophication in marine systems (Malone et al.,

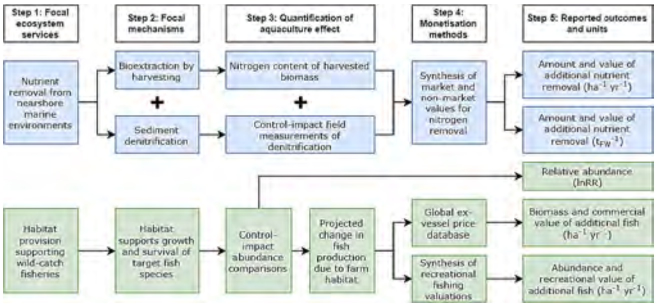

**Fig. 1.** Conceptual diagram of the focal ecosystem services, mechanisms, methods and key outcomes from the present analysis. Outcomes are expressed as 'additional' value, but in cases where farms have a detrimental effect on the focal mechanism, that additional value will be negative. At Step 5, species-level estimates for additional biomass, abundance and corresponding economic values are aggregated to the level of habitats within studies (i.e. assemblages). Where relevant, we also report outcomes according to the farmed taxa (e.g. mussels, oysters and seaweeds), the farming method (on– or off-bottom), nitrogen removal mechanism (bioextraction or denitrification), or characteristics of the reference habitat (presence or absence of natural habitat structure).

1996; Paerl, 2018; Ryther and Dunstan, 1971). Moreover, because most nutrient mitigation approaches remove both nutrients simultaneously, the value of phosphorus removal is largely captured by the nitrogen value (Ferreira and Bricker, 2018).

Peer-reviewed articles, theses and technical reports returned by the literature search were screened by reading the title and abstract, and where necessary, the full text. Studies were included in the database if they provided estimates of nitrogen bioextraction, and/or changes in nitrogen biodeposition rates at working farms, modelled farms, or experimental farms intended to simulate conditions at working farms. For denitrification rates, only studies that measured denitrification on working or experimental farms, relative to uncultivated controls, were included. We did not account for denitrification measurement methods in the analysis, although a recent review notes that the acetylene block method underestimates true denitrification rates (Ray et al. 2021). Our inclusion of 3 effect size estimates derived using the acetylene block method (cf. 11 using isotope pairing and 5 using $N_2$/Ar methods) will not bias the direction of effects but may produce a slightly conservative global value for denitrification enhancement/diminishment. We did not consider data from wild bivalve or seaweed populations, nor from systems identified as integrated multi-trophic aquaculture (IMTA) with crustaceans or finfish, as such systems are not representative of predominant aquaculture practices worldwide. Where possible, we omitted sites for which the primary economic driver was nutrient mitigation, including those that were unable to produce bivalves of marketable size within a typical production cycle, because such sites are rare in the global context and our valuation relies on an assumption of nutrient removal as an incidental benefit.

### 3.1.2. Calculation of nutrient removal effect sizes

Effect sizes were calculated for nitrogen removal by bioextraction and denitrification, respectively, with values converted to nitrogen removal per area (kg N $ha^{-1}$ $yr^{-1}$) and per fresh weight harvest volume (kg N $t_{FW}{}^{-1}$) whenever possible. Farm area was taken as reported by the study authors, preferably the footprint of the area being actively farmed (including the space between any gear), or else the area of the entire lease. Nitrogen removal per harvest volume was more often available for bioextraction estimates than denitrification. We primarily present nitrogen removal rates in per-area levelised units, as this is the most relevant metric for spatial management of aquaculture activities. However, nitrogen removal per area depends on the intensity of farming among other factors (Parker and Bricker, 2020), and accordingly, we also provide values per harvest volume where available.

### 3.1.3. Economic valuation of nutrient removal

Nutrient reductions can be monetised via three main approaches. The replacement/avoided cost method is perhaps the most common in the ES literature, such that the unit value of nutrient removal is based on the cost of applying the least-cost alternative mitigation option that is equivalent and suitable for the local context (Freeman et al., 2014). The second approach is to observe payments made for nutrient offsets, such as within a nutrient credit trading program, on the assumption that observed prices will tend to reflect public preferences. This is not always the case, because in many jurisdictions, prices are set by regulators based on estimated costs of mitigation using approaches that are not necessarily the least-cost option. The third approach is to use stated preferences, whereby resource users are asked about their willingness to pay for a given nutrient reduction (Interis and Petrolia, 2016). To apply monetary values to nitrogen removal associated with bivalve and shellfish aquaculture, we compiled 75 valuations of nitrogen removal in coastal systems from 1996 to 2020. We considered values derived from observed payments within nutrient offset trading programs and those derived from replacement cost methods. We omitted replacement cost estimates that we considered unlikely to be a least-cost option in any context (all were $> 5000$ USD $kg^{-1}$: Melbourne Water, 2019; Stephenson et al., 2010). The remaining valuations ranged from 0-2384 USD $kg^{-1}$ in 2020 currency (central values 2–1359).

## 3.2. Habitat provision

### 3.2.1. Literature search and data selection

Literature on the effect of bivalve or seaweed farms on fish and mobile macroinvertebrate populations were discovered following the same process as the nutrient removal literature search (Table A.1, Suppl. Text A.1). Hereafter, the terms ‘fish’ and ‘fishes’ refer to both fishes and invertebrates.

Studies were included in the literature database if they provided species-level data on relative abundance of wild fish at farm (impact) sites and relative to control sites. We limited the dataset to species with some demonstrable fisheries value, provided at least 10 individuals had been recorded in at least one of the two habitat types. Suitable relative abundance data included population density estimates as well as catch-per-unit-effort from stationary traps, nets or video stations. Other types of experiments such as tagging and tracking studies generally did not provide usable data for this purpose, while several relevant studies were not included because they did not report species-level data. Because the farm footprint is often not a homogeneous habitat, we considered whether the sampling method was likely to representatively sample the whole farm footprint and corrected the estimate accordingly (Supp. Text A.2).

### 3.2.2. Calculation of relative abundance

Species abundance responses to fish farms were standardised to a common effect size, the natural log of the response ratio (Hedges et al., 1999): $lnRR = \ln(F/R)$, where $F$ is non-zero abundance at farm sites and $R$ is non-zero abundance at reference sites. This is a typical approach for meta-analyses of ecological data, as taking the natural log of the relative abundance ($F/R$) normalises the distribution, with positive values indicating positive responses and vice versa.

### 3.2.3. Estimation of production enhancement

Productivity was estimated at farm and reference sites using a mechanistic life-history-based approach, starting from an estimate of juvenile density and applying published life-history parameters to model surviving biomass over the expected lifetime of a cohort. Juveniles were used as the starting point because they are generally easier to sample than adults and more resident in inshore habitats, such that their density can be estimated with higher confidence. This approach has been used previously to quantify enhancement of fish production by specific inshore habitats, most notably on oyster reefs (Peterson et al., 2003; zu Ermgassen et al., 2016). Those studies inferred production enhancement by identifying ‘enhanced’ species and fully or partially crediting their lifetime production to oyster reef habitat based on evidence of habitat-limitation (reliance on reefs or other structured habitats) or foraging behaviour (growth enhancement by feeding at reefs). However, omitting species displaced by a shift from unstructured to structured habitat will not count their lost production potential as a negative effect, affecting the additionality of the habitat value estimate. We opted not to limit the dataset to enhanced species, but instead credited a positive or negative effect on production according to the observed difference between habitat types (for all valuable species that were effectively sampled). Because production at farms is compared to a range of reference habitats, this approach also makes it possible to directly compare the relative benefit (or cost) of situating farms over benthic habitats with high or low levels of pre-existing fisheries productivity, and identify optimal placement where, for example, natural habitat structure has been lost (Beck et al., 2011; Greening and Janicki, 2006; Orth et al., 2006). The main drawback is that production estimates are less rigorously linked to the species’ ecology, such that spurious habitat effects could arise in either direction. This could be addressed by omitting comparisons when the direction of the effect does not conform to prior expectations, but to avoid confirmation bias (Mykoniatis and Ready, 2020), expectations would need to be strongly justified. We elected not to take this approach, given the lack of evidence for the habitat requirements of many species in our dataset.

Age- and size-frequency distributions were simulated using published growth and mortality parameters, broadly following recent applications of this method (Jänes et al., 2020; Lai et al., 2020; zu Ermgassen et al., 2016). Full details are given in Supp. Text A.2 and Figure A.1). Necessary species life history parameters (the growth coefficient $K$, length-at-infinite-age $L_{inf}$, maximum age $t_{max}$, and weight-at-length coefficients $a$ and $b$) were mostly obtained from the FishBase database, with help from the R package rfishbase (Boettiger et al., 2012; Froese and Pauly, 2020). For some species, it was necessary to find single estimates in the literature, or take values from related species with a preference for the lowest and closest taxonomic level for which an estimate was available, usually congenerics. Production estimates also depend on the instantaneous natural mortality rate $M$, yet $M$ varies according to estimation methods and local drivers of mortality (Hamel, 2014). As few species in our dataset have directly estimated $M$ values in the literature, and considering the large uncertainties in transferring $M$ values across locations and habitat types, we instead considered four $M-$estimation methods developed for data-deficient species (Then et al., 2015; Supp. Text A.2).

### 3.2.4. *Economic valuation of fish production*

Value to commercial fishers was estimated via ex-vessel prices obtained from the Sea Around Us ex-vessel price database (Sumaila et al., 2007), aggregated at the level of species if available, or else genera. Where ex-vessel prices were not available at genus level or lower, we instead applied published median ex-vessel estimates at the level of ISSCAAP classifications (Melnychuk et al., 2017). Species that are exclusively targeted by subsistence fisheries or for the aquarium trade were not assigned a monetary value. Species in data-poor fisheries were sometimes considered commercially valuable based on anecdotal evidence such as availability at fish markets and assigned to the most relevant ISSCAAP value category. Value to recreational fishers was informed by previous estimates of willingness-to-pay, usually based on stated preference methods. We only considered estimates that provided marginal values (i.e. the value of catching an additional fish while already on a fishing trip) and concerned coastal marine or sea-run diadromous recreational fisheries, as these are most relevant to the environments where bivalve and seaweed farming occurs.

## 3.3. *Synthesis of ecosystem services and values*

Central estimates are reported with nonparametric bootstrapped 95% confidence intervals (percentile method, simpleboot package for R: (Peng, 2019; R Core Team, 2020)), unless otherwise stated.

### 3.3.1. *Analysis of nutrient removal effects*

The final nutrient removal dataset included 102 estimates of nitrogen removal per area from 55 studies (Suppl. Text A.1, Appendix C). To allow a basic transfer of nutrient removal rates, we fitted two linear models to compare nitrogen removal rates according to taxa, culturing methods and removal mechanisms. The response variables (Model 1: kg N $ha^{-1}$ $yr^{-1}$; Model 2: kg N $t_{FW}^{-1}$) were each shifted (translated) to positive values by adding a constant, and then log-transformed, after which a Gaussian model provided an acceptable fit. Both models were specified with factors for the removal mechanism assessed (2 levels: bioextraction or denitrification), the taxa (4 levels, limited to those with sufficient sample sizes: oyster, mussel, clam, seaweed) and the culturing method (2 levels: on– or off-bottom). We tested for an interaction between taxa and removal mechanisms under the expectation that the relative importance of bioextraction and denitrification would differ across taxa; only per-area denitrification estimates (Model 1) had sufficient representation to fit this term. Significance was tested using type II sums of squares (car package: (Fox and Weisberg, 2019). Adjusted predictions were extracted using the ggeffects package (Lüdecke, 2018) and returned to the original scale.

### 3.3.2. *Analysis of habitat provision effects*

We omitted two very influential studies that sampled oyster gear directly, as we could not confidently standardise sampling effort across habitats. The first reported very high densities of juvenile blue crabs (*Callinectes sapidus*) in both oyster baskets and macrophyte habitats, with very few in unstructured habitats (Stewart, 2015). The second reported very high densities of shrimp, tautog and tomcod. Abundance of both tautog and tomcod appeared to be greatly enhanced by farm structure (DeAlteris et al., 2004). The final habitat provision dataset included 182 abundance comparisons of 128 unique species from 26 studies (Suppl. Text A.1, Appendix C).

To identify factors predicting habitat provision effects and resulting monetary values, we fitted linear models to (i) species-level relative abundance data, as the most direct measure of the effect of farm habitats on fish populations, and then to (ii) the assemblage-level monetary values resulting from the sum of species-level responses within each habitat type within each study. We did not fit a model to species-level monetary values, as they mean little without reference to the rest of the fish assemblage. The relative abundance model (Model 3) was fitted with *lnRR* as the response variable and four factors: farmed taxa (3 levels: mussels, oysters, seaweed), culturing method (3 levels: elevated, on-bottom with gear, on-bottom without gear), the choice of reference habitat structure (2 levels: primarily unstructured or structured), and a ‘taxa × culturing method’ interaction term. Samples were weighted by the square root of the study sample size. Model terms were significance-tested and adjusted predictions extracted as for the nutrient removal model. Two similar models were fitted to assemblage-level data, and were specified and tested as above, with Model 4 assessing predictors of additional commercial value and Model 5 recreational value.

### 3.3.3. *Standardisation and application of economic values*

Monetary values, as presented here, highlight the benefits to society that arise from enhanced ES provision, but do not guarantee an actual exchange of money. Realised values depend on local regulations and logistics, as well as access to suitable markets. Otherwise, nutrient credit or avoided cost values can be viewed as a proxy for the value of nutrient removal to society.

Global currencies were converted to USD based on OECD purchasing power parity (PPP) estimates for the relevant year, with all values then adjusted for inflation to the equivalent of 2020 USD. Monetary values for both nutrient removal and additional fish were highly variable and right-skewed (Figure A.2), likely reflecting valuation methods as well as local contexts and perceptions of value. For conservatism, the median rather than mean is used as the central estimate of monetary value.

Estimates of enhanced nitrogen removal were multiplied by the median per-kg nitrogen value returned by the literature review. Valuations derived from observed offset market prices were slightly lower than those from non-market methods (Figure A.2). Economic value transfer across global regions was done naively, due to a lack of data on (i) the value of nutrient removal services outside North America and Europe (Figure A.3), and (ii) the proportion of aquaculture production that occurs in areas with an excess of nutrients. As such, the values reported here are best applied where nutrient removal is equivalently valued.

The monetary value of enhanced fish populations was monetised separately for commercial and recreational fisheries. For commercial fisheries, the change in landable biomass was multiplied by the mean ex-vessel price (FAO Fisheries and Aquaculture Department, 2020a). For recreational fisheries, the change in density of landable fish was multiplied by the median marginal value of an additional fish within the relevant value category (7 possible categories: bait, table crab, panfish, table fish, flatfish, prized table fish, gamefish, lobster), and reported together with the range of central estimates. Species that are primarily

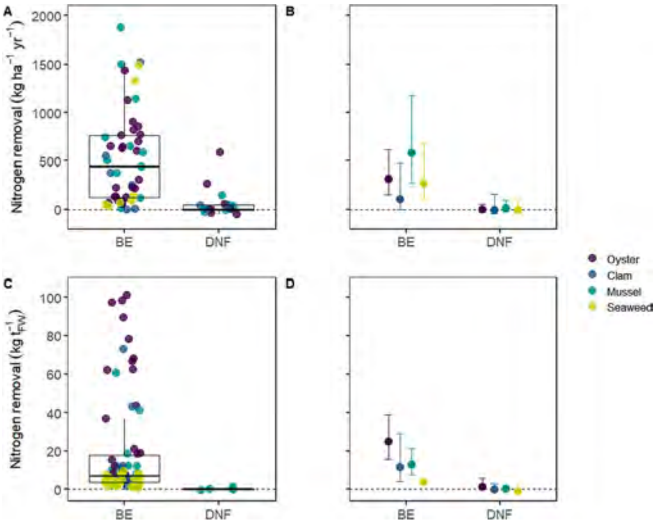

**Fig. 2.** Comparison of nitrogen removal by bioextraction (BE) and additional denitrification (DNF) rates at bivalve and seaweed farms according to the farmed taxa. **Panels A and B** show nitrogen removal by farm area, Panels B and C show nitrogen removal by harvest weight. Boxplots in Panels A and C indicate 25%, 50% (median) and 75% percentiles; whiskers are set at 1.5x interquartile range. **Panels B and D** present adjusted predictions and 95% confidence intervals from a linear model of nutrient removal rates with model terms for the farmed taxa, nitrogen removal mechanism, culturing method (on/off-bottom) and for Panel B only, a ‘mechanism × taxa’ interaction term (Model 1: Table S1). Predictions are conditional on off-bottom culturing. Model predictions for DNF by harvest weight (Panel D) are based on studies of mussel farms only, and are not included in the valuation.

harvested as bycatch or for bait are assigned a nominal ex-vessel value of 500 USD t$^{-1}$, while lobsters are assigned a nominal recreational value of 10 USD each. We credit habitats with the full value of increased or decreased landable production, assuming that (i) the absolute cost of fishing does not increase with catch size, and (ii) because we did not include a fishing mortality parameter, that the fishery is perfectly efficient – all fish that reach landable size are caught, but not until the moment before natural mortality. The true value will certainly be lower and dependent on local fisheries context.

## 4. Results

### 4.1. *Value of nitrogen removal*

Bioextraction via harvesting, combined with effects of farming on sediment chemistry, led to an increase in nitrogen removal rates within farm footprints compared to non-farmed habitats. This effect held true whether nitrogen removal was quantified per area of farm footprint or per harvest volume (Fig. 2). Bioextraction via harvesting removed by far the most nitrogen per area across all species and farming systems, with a mean removal of 644 kg N ha$^{-1}$ yr$^{-1}$ (median 505) across all 49 estimates (Table 2). In harvest volume units, mean nitrogen removal by bioextraction was equivalent to 19 kg N $t_{FW}^{-1}$ (median 6.8) across all 74 estimates (Table 2). Enhanced denitrification appears to be less important than bioextraction, although the effect of farms is still positive, with an additional 59 kg N ha$^{-1}$ yr$^{-1}$ (median 3.2) at farms (Table 2). The statistical model indicated that the nitrogen removal mechanism in question (bioextraction or denitrification) was a significant predictor of nitrogen removal per area, but the farmed taxa was not, and nor was the choice of on– or off-bottom cultivation (Model 1: Table B.1). There were significant effects of taxa and nitrogen removal mechanism on nitrogen removal per harvest volume (Model 2: Table B.1).

**Table 1**

Potential benefits and costs of non-fed bivalve and seaweed aquaculture from the perspective of ecosystem service provision.

| Ecosystem service | Beneficial mechanisms | Costly mechanisms |
|---|---|---|
| ***Provisioning*** | | |
| Seafood, pharmaceuticals and other products | Non-fed aquaculture is highly productive relative to its spatial allocation, yet has low carbon intensity. Some aquaculture-environment interactions may benefit provisioning by wild harvests. | Spatial exclusion of wild catch fisheries from aquaculture leases. Some aquaculture-environment interactions may negatively affect provisioning by wild harvests, e.g. larval capture, food competition, microplastic pollution. |
| ***Cultural*** | | |
| Employment | Rural and urban jobs in aquaculture and supply chains. | Lost jobs in industries spatially displaced or otherwise negatively affected by aquaculture, e.g. commercial fishing. |
| Lifestyle and ‘sense of place’ | Direct involvement in aquaculture or consumption of locally-grown seafood can increase connection to the environment. | General aesthetic or ecological effects of aquaculture may detract from the ‘sense of place’ and connection to the environment. |
| Tourism and recreation | Aquaculture can provide new tourism experiences (e.g. farm-to-table oyster tours). Regulating and supporting services, if enhanced by aquaculture, will benefit ecotourism and recreation. | Aesthetic losses due to aquaculture infrastructure and any plastic pollution, impacts on charismatic megafauna, spatial exclusion of some activities from aquaculture leases. In very eutrophic waters, farms may seed macroalgal blooms and impact amenity. |
| ***Regulating*** | | |
| Remediation of water quality | Nitrogen and phosphorus from terrestrial inputs are assimilated into tissues and removed from the marine environment during harvest. Some evidence that sediment denitrification rates are elevated beneath farms. Filter feeding by bivalves removes suspended sediment and planktonic biomass. Farm structure slows water movement and encourages sedimentation. | Impacts on benthic macrophytes by organic sedimentation or shading could increase sediment resuspension. |
| Coastal defence | Farm structures attenuate waves and slow down storm surge. | Farm structures alter flow with undesirable effects on shorelines. |
| Carbon sequestration | Carbon is assimilated into tissues, although long-term sequestration depends on eventual fate of tissues. | Bivalve shell formation may be a net producer of $CO_2$ |
| ***Supporting*** | | |
| Habitat for wildlife | Farm structures provide habitat with shelter, feeding opportunities and aggregation points for social behaviours. | Frequent or severe disturbance regimes could reduce habitat quality for resident animals, especially if mortality risk is increased. Species that prefer soft sediment habitats may be negatively affected by the addition of structure. Nutrient-rich deposition within the farm footprint alters benthic communities. |
| Protection of wildlife | Farms exclude ecologically damaging activities such as bottom trawling. | Ropes and other gear may be an entanglement risk for large marine animals, e.g. whales. Some species may avoid farming areas due to noise and be disadvantaged as a result. Occasionally, nuisance animals may be culled. |

**Table 2**

Estimated nitrogen removal by bioextraction of farmed bivalves and seaweeds. Estimates are grouped according to farmed taxa and pooled across global regions. Nitrogen removal rates are adjusted predictions from statistical models (Models 1 and 2: Table S1) together with 95% confidence intervals. Denitrification values are close to zero (not shown here). Monetary values assume a nitrogen price of 32.3 USD kg $N^{-1}$, the median value returned by a review of nitrogen valuations across Europe and North America.

| | Nitrogen removal by area | | | Nitrogen removal by harvest volume | | |
|---|---|---|---|---|---|---|
| Taxon | kg $ha^{-1}$ $yr^{-1}$ | USD $ha^{-1}$ $yr^{-1}$ | n estimates | kg $t_{FW}^{-1}$ | USD $t_{FW}^{-1}$ | n estimates |
| Clam | 107 (-3–477) | 3452 (-99–15410) | 7 | 11 (4–29) | 128 (368–937) | 6 |
| Mussel | 581 (275–1172) | 18,756 (8900–37865) | 12 | 13 (8–21) | 245 (416–690) | 11 |
| Oyster | 314 (150–612) | 10,147 (4854–19781) | 22 | 25 (16–39) | 505 (801–1255) | 23 |
| Scallop | 52 [a] | 1670 | 0 | 4.2 [a] | 136 | 0 |
| Seaweed | 275 (96–678) | 8889 (3084–21886) | 8 | 3.8 (2.6–5.4) | 84 (124–175) | 34 |

[a] In the absence of direct studies of scallop bioextraction, we estimated scallop bioextraction based on modelled production intensity of Chinese scallop *Chlamys farreri* farms in north-east China (12.4 $t_{FW}$ $ha^{-1}$ $yr^{-1}$: Ferreira et al. 2007) and estimates of nitrogen content from van der Schatte Olivier et al. (2020), who in turn used data from Hardy & Smith (2001) and Zhou et al. (2002).

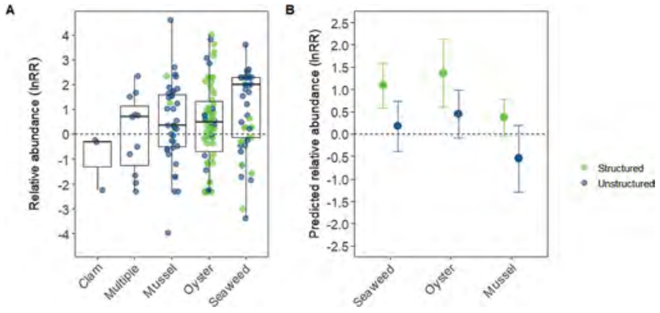

**Fig. 3.** Relative abundance of targeted fish species at aquaculture sites relative to natural reference habitats, according to the farmed organism and degree of structure at the reference habitat type (unstructured = soft sediment or patchy, structured = macrophytes or reef). Values are presented as (**panel A**) observed relative abundance expressed as the natural log response ratio, and (**panel B**) adjusted model predictions of relative abundance according to the farmed taxa and reference habitat structure (Model 3, Table A.2). The overall enhancement effect is significantly different from zero. Insufficient data were available for clam, scallop or mixed bivalve aquaculture to be included in the statistical model (panel A). Boxes indicate the 25%, 50% (median) and 75% percentiles, whiskers indicate 1.5x the interquartile range. Error bars indicate 95% confidence intervals around model predictions.

For bivalve farms with access to a suitable nutrient credit trading scheme or other incentive (and applying values from Europe and North America), nutrient removal by bioextraction and denitrification together could be worth 3264–19124 USD $ha^{-1}$ $yr^{-1}$, depending on the species, 366–845 USD $t_{FW}^{-1}$ (Table 2). Scallops are data-deficient, but based on available evidence are also likely to be of considerable value for nutrient mitigation (Table 2). For seaweed, bioextraction is the primary mechanism studied, potentially worth 1861–24981 USD $ha^{-1}$ $yr^{-1}$ or 47–172 USD $t_{FW}^{-1}$ (Table 2).

### 4.2. Value of habitat provision

Farm habitats were associated with a higher relative abundance of targeted fish species than reference habitats, with a weighted mean *lnRR* of 0.48 (median: 0.61). This corresponds to a 1.6x increase in abundance per species assessed (Fig. 3). The choice of reference habitat was the only significant predictor of the observed effect on species-level *lnRR* (Model 3, Table B.1). The weighted mean sampled density was 273 individuals $ha^{-1}$ at farms cf. 143 $ha^{-1}$ at reference sites, giving an additional 130 individuals $ha^{-1}$ (95% CI: 67–198) at farms (median: 12). This includes studies for which it was necessary to convert abundance per sample to density (n = 95 species comparisons); without those comparisons, the weighted mean density is 288 individuals $ha^{-1}$ at farms and 131 individuals $ha^{-1}$ at reference sites (Figure B.1).

Annual production was equivalently enhanced (Fig. 4). Our standardised estimation of *Djuv* returned a mean juvenile density of 187 $ha^{-1}$ at farms (95% CI: 133–248) and 105 $ha^{-1}$ at reference sites (95% CI: 69–146), resulting in an additional 105 kg $ha^{-1}$ $yr^{-1}$ $species^{-1}$ produced at farms. These values assume equal mortality at farms and reference sites and are affected by the M−estimation method, however, the finding of enhancement is robust to small relative increases in mortality at farms (Table 3). The effect was, in general, driven by a subset of highly productive fisheries targets.

We report economic values at the level of fish assemblages (26 assemblages from 21 studies with minimum two species effectively sampled, mean 6.8 (range 2–32) species per assemblage: Table 4). Farm-associated fish assemblages were estimated to produce an additional 703 kg $ha^{-1}$ $yr^{-1}$ (median: 125) if not caught. If all fish are available to commercial fishers and are caught at the optimal time, individuals that

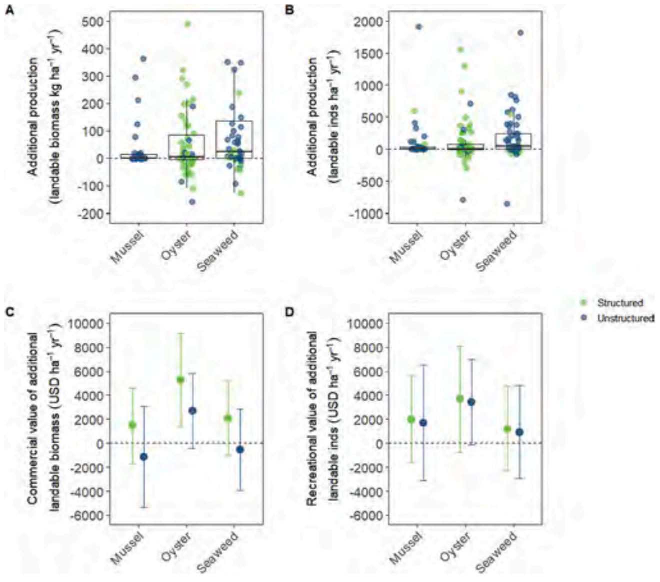

**Fig. 4.** Projected additional production of targeted fish species due to aquaculture habitat, relative to structured and unstructured reference habitats. Positive values indicate species that are enhanced by the presence of aquaculture habitat, while negative values indicate a decrease in productivity for that species. **Panels A and B** show the projected additional biomass and abundance of individuals of landable size produced annually at farms relative to reference habitats, assuming steady state production. Boxes indicate the 25%, 50% (median) and 75% quartiles, with whiskers indicating 1.5x the interquartile range. **Panels C and D** give adjusted predictions for the assemblage-level monetary value of additional production, conditional on the degree of structure at the reference habitat type (unstructured = soft sediment, structured = macrophytes or reef). Error bars indicate 95% confidence intervals. Commercial values (panel C) are based on mean ex-vessel prices, while recreational values (panel D) assume the median recreational value for each recreational fish category.

reach landable size have a potential ex-vessel value of 1528 USD ha$^{-1}$ yr$^{-1}$ (median: 157). The equivalent value to recreational fishers is estimated at 1926 USD ha$^{-1}$ yr$^{-1}$ (median: 585). Assemblage-level enhancement was not significantly predicted by the farmed taxon, gear type or choice of reference habitat (commercial and recreational value: Models 4 and 5, Table B.1). However, given evidence that the choice of reference habitat predicts relative abundance effects at species level (Model 3, Table B.1; Fig. 3), we consider it worthwhile to make predictions that are conditional on farms being placed on, and compared to, unstructured natural habitats. Seaweed, oyster and mussel farms are predicted to generate additional fish production with a value to commercial fishers of (seaweed) 2059, (oyster) 5267, and (mussel) 1453 USD ha$^{-1}$ yr$^{-1}$, respectively, relative to unstructured reference habitats (Fig. 4).

### *4.3. Global growth scenario*

To place these monetary values within the global context, we applied them to a simple scenario of 3-fold higher global non-fed aquaculture production by 2050, enabled by policy reform that supports the most sustainable forms of aquaculture. This scenario assumes uniform growth across global regions and commodities, and is conditional on all additional production occurring where nutrient removal is desirable and where farm infrastructure does not compete with natural habitat structure (other conditions and details are given in Suppl. Text B.1). Given these conditions are met, the nitrogen removal and habitat provision associated with the additional production volume (relative to 2018) could have value on the order of 17–56 billion USD annually (Table B.2).

## 5. Discussion

Nutrient removal and habitat provision services were selected for valuation due to the availability of numerous independent effect sizes in the literature, and their relative amenability to monetisation. In aggregating these effect sizes, we find that the positive effects on these services tend to outweigh negative effects on the same services. Within the scope of our valuation, a hectare of inshore zone allocated to oyster, mussel or seaweed farming is estimated to remove nitrogen worth between 3084–37865 USD yr$^{-1}$ (Table 2). Simultaneously, that same hectare of farm footprint may provide habitat that supports additional production of landable fish potentially worth 972–2504 USD yr$^{-1}$ to commercial fishers or 1087–2848 USD yr$^{-1}$ to recreational fishers (Table 3). However, we wish to emphasize that by focusing on these two ES, we do not consider costs of production, market impacts and other externalities, such as potential trade-offs among ES (Table 1). Only a full benefit-cost analysis, which is beyond the scope of this work, would ascertain whether non-fed aquaculture yields net positive impacts across the board at local, regional or global scales. In addition, we also note the biased regional representation of effect sizes and economic valuations. The lack of representation across farmed taxa and methods within each region precluded any meaningful interpretation of geographic differences in effects on ES values, but we expect that certain values will be more or less susceptible to geographic bias. For instance, nutrient bioextraction per harvest volume is relatively consistent within species across geographic distance, as species-level parameters such as tissue: shell ratio and nitrogen content vary within a narrow range (Appendix C). Alternatively, annual nitrogen removal per farm area will be strongly influenced by stocking density and growth rates, which in turn depend on local farming practices and environmental parameters. The economic valuation of ES or disservices may be most susceptible to geographic bias, as willingness-to-pay is expected to vary among localities and countries. Key limitations and assumptions, including those noted above, are summarised in Table 5 and discussed below.

**Table 3**
Comparison of species biomass production estimates according to the method of estimating the instantaneous natural mortality rate (*M*). Methods are detailed in Appendix A. Production rates are presented as weighted means and weighted nonparametric bootstrapped 95% confidence intervals. $M_{mean} + 10\%$ increases mortality at farms by 10%, but leaves mortality at reference sites unchanged.

| M | Farm habitat production (kg ha$^{-1}$ yr$^{-1}$) | Reference habitat production (kg ha$^{-1}$ yr$^{-1}$) | Production enhancement at farms (kg ha$^{-1}$ yr$^{-1}$) |
|---|---|---|---|
| $M_{mean}$ | 190 (114–283) | 84 (46–132) | 105 (43–180) |
| $M_{nls}$ | 518 (259–846) | 191 (94–319) | 328 (101–619) |
| $M_{Pauly}$ | 248 (145–376) | 119 (58–193) | 129 (43–328) |
| $M_K$ | 158 (100–226) | 81 (41–133) | 77 (33–129) |
| $M_{K2}$ | 104 (67–150) | 46 (28–67) | 59 (29–95) |
| $M_{mean}$ + 10% | 149 (91–222) | — | — |

### *5.1. Nutrient removal valuation*

The approach employed here was aimed at detecting broad patterns to facilitate the application of globally relevant economic values, rather than highlighting the nuances of nutrient dynamics that will govern ES provision at regional and local scales, and for particular farmed species. While we have confidence in the broad findings, especially for well-researched oyster and mussel species, some globally important farmed taxa (especially clams and scallops) are data-poor and valuations should be interpreted with a degree of caution that reflects the sample size of estimates as well as the reported ranges and confidence intervals. Notably, nutrient removal via oyster or mussel farming was over-represented in the dataset, with most estimates coming from farms in the United States and Europe stocked with *Crassostrea virginica*, *C. gigas*, *Mytilus edulis* or *M. galloprovincialis*. The regional imbalance in nutrient removal studies is most obvious for Asia, which was represented by only 10 out of 42 bivalve estimates in our dataset despite accounting for ~ 92% of global mollusc aquaculture in 2018 (FAO Fisheries and Aquaculture Department, 2020a).

There was a clear finding across all farming systems that nitrogen removal services occur primarily via bioextraction, with a relatively small contribution from denitrification. The large mass of nitrogen removed during harvesting means that during their lifetime, farms remove more nitrogen from coastal systems than unharvested wild bivalve habitat, despite many bivalve habitats being associated with enhanced denitrification—likely at comparable rates to farm habitats (Ray and Fulweiler, 2020)—as well as burial of organic nitrogen (Newell et al., 2005a,b). An early comparison of nutrient removal by inshore habitats found that oyster reefs, marshes and submerged vegetation provided similar denitrification enhancement cf. bare sediment habitats in North Carolina, USA, with the mean enhancement at oyster reefs (269–486 kg N ha$^{-1}$ yr$^{-1}$) valued at ~ 4000–7000 USD in 2020 dollars (Piehler and Smyth, 2011). However, most studies have found lower values; for example, Beseres Pollack et al. (2013) estimated that oyster reefs in Texas, USA, removed 7.5 kg N ha$^{-1}$ yr$^{-1}$ through denitrification and burial of sediments, worth ~ 70 USD (2020 dollars). Sustainable harvest of shellfish reefs does provide bioextraction services, but there is likely to be a substantial trade-off in lost filtration and natural nitrogen removal services as the standing biomass is reduced (DePiper et al., 2017; Kasperski and Wieland, 2009; Newell et al., 2005a,b). Presence of macroalgae has also been found to enhance denitrification rates in some contexts, although it likely depends on the biomass of macroalgae at the sediment interface where it can regulate nutrient fluxes (Dalsgaard, 2003), rather than the standing biomass above the sediment. Overall, large-scale in-situ nutrient removal is likely to be best achieved by responsible placement and use of bivalve or seaweed aquaculture for human consumption and nutrient bioextraction, together with protection and restoration of natural habitats in service of other conservation goals.

**Table 4**

Descriptive statistics for effects of aquaculture habitat on local production of targeted fish species. Values are for assemblages, i.e. the net effect across all targeted species effectively sampled by a study (minimum 2 species per study for inclusion). Values are presented as means and bootstrapped 95% confidence intervals (both weighted by the number of species effectively sampled). Additional population density can take negative values if aquaculture is associated with lower population density. Values are aggregated regardless of reference habitat (structured or unstructured). Clam and scallop aquaculture are data-deficient for habitat value.

| Farmed taxa | n | Relative abundance (lnRR) [a] | Additional production (total kg $ha^{-1}$ $yr^{-1}$) [b] | Additional production (landable individuals $ha^{-1}$ $yr^{-1}$) [b] | Additional production (landable kg $ha^{-1}$ $yr^{-1}$) [b] | Recreational value (USD $ha^{-1}$ $yr^{-1}$) [bc] | Commercial value ex-vessel (USD $ha^{-1}$ $yr^{-1}$) [bc] |
|---|---|---|---|---|---|---|---|
| Oysters | 12 | 0.86 (0.36–1.37) | 1147 (172–2346) | 456 (34–1066) | 1110 (158–2237) | 2848 (476–6603) | 2504 (180–5290) |
| Mussels | 5 | 0.53 (-0.08–1.12) | 363 (59–764) | 244 (34–478) | 348 (57–741) | 1919 (336–4125) | 997 (139–2042) |
| Seaweeds | 7 | 0.69 (0.25–1.22) | 529 (-144–2452) | 680 (60–2129) | 494 (-158–2339) | 1087 (143–3454) | 972 (-538–4994) |

[a] lnRR = ln(farm/ref), where farm is individuals per sample at farms and ref is individuals per sample at reference sites

[b] For studies that reported relative abundance only, measures of density were obtained by estimating the area effectively sampled. Production was based on juvenile density. For studies that did not specifically target juveniles, we estimated the juvenile density based on the reported density and assumptions about the size-selectivity of the methods used. Landable production is based on the abundance and weight of individuals that are expected to survive to landable size.

[c] Commercial values are based on ex-vessel price estimates (Sumaila et al. 2007, Melnychuk et al. 2017); recreational values are based on the following median price estimates for an additional fish caught (2020 USD): panfish $2.35; table fish $6.95; flatfish $10.27; gamefish $10.51; prized table fish $22.90

We were not able to meaningfully quantify differences in nutrient removal between regions, as research is generally skewed toward a few locally important species. Future analyses will benefit from new data on species, farming systems and regions that are currently under-researched relative to their importance. Benefit transfers across cultures and economies are difficult to do with confidence, and even meta-regression models produce benefit functions that have considerable margins of error when applied to new locations (Ready and Navrud, 2006; Rosenberger, 2015). Studies indicate that willingness-to-pay for ES may be more strongly correlated with income levels than cultural differences, and transferring valuations with a correction for purchasing power can give reasonably accurate results in some situations (Hynes et al., 2013; Jacobsen and Hanley, 2009). We corrected for inflation and purchasing power when aggregating valuations into international currency (2020 USD), but did not attempt to make a similar correction when applying those aggregated values to global aquaculture production, which takes place across numerous countries with differing wealth. Values should be interpreted accordingly.

Where nutrient mitigation is a management goal, existing or hypothetical nutrient credit markets are expected to find the least-cost solutions for nutrient abatement within a watershed, such that even our lower 95% confidence limit for nitrogen value (27 USD $kg^{-1}$) could be a severe overestimate for regions where ‘low hanging fruit’ remain. Even in the EU and US, where there has already been considerable investment in wastewater and stormwater management, some valuations and credit schemes have estimated replacement costs as low as 3–11 USD $kg^{-1}$ in 2020 dollars (Beseres Pollack et al., 2013; CT DEEP, 2020; Lindahl et al., 2005; Newell et al., 2005a,b; Stephenson et al., 2010). However, there are places where least-cost alternatives may still be very expensive. A recent valuation in Maryland, USA, estimated that nitrogen removal by six oyster farms (1.6–8.9 ha) was worth 8900–266000 USD $yr^{-1}$ if wastewater treatment plant upgrades (3–8 mg N $L^{-1}$) were taken to be the avoided cost (Parker and Bricker, 2020), while in Melbourne, Australia, new developments that do not meet on-site treatment standards are charged $>$ 5,000 USD $kg^{-1}$ for nitrogen offsetting via stormwater wetlands (Melbourne Water, 2019). Rates have risen over time as costs of abatement increase, from $<$ 1000 USD $kg^{-1}$ in 2005 (Whiteoak, 2019). This program was not included in our valuation as it may not have targeted the least-cost abatement option. In North Carolina, USA, nutrient credit rates have remained relatively consistent over time: inflation-corrected credit rates were generally flat between 2001–2014, increased 4-fold in 2014–2019, before falling again to 2001–2014 levels. At the time of writing, 2021 rates were set at 20–238 USD $kg^{-1}$ depending on the watershed (NC DEQ, 2021).

Nutrient credit payments and replacement cost valuation methods both approximate the perceived value of—or willingness to pay for—nutrient removal services. This has been supported by alternative valuation methods. For example, Interis & Petrolia (2016) valued nutrient removal by oyster reefs in the Gulf of Mexico using a willingness-to-pay choice experiment, and found that a 10% reduction in nitrogen and phosphorus levels associated with hypothetical construction of a 607-ha oyster reef was worth 69 USD (95% CI: 47–95) as a one-time payment per household (5196 respondents). However, there were seemingly diminishing returns in perceived value with greater nutrient removal, with a 20% nutrient reduction worth 94 USD (95% CI: 67–127). These diminishing returns in willingness-to-pay interact with the higher cost of achieving more ambitious nutrient mitigation targets (especially if the most cost-effective measures have already been taken), such that it is likely to be beneficial for non-fed aquaculture to be located where nutrient mitigation is most needed (Dvarskas et al., 2020).

### 5.2. *Habitat provision valuation*

This analysis is, to our knowledge, the first regional or global valuation of habitat provision by aquaculture. Production enhancement at seaweed and bivalve farms appears generally lower than in natural nursery habitats (especially seagrass and oyster reefs), although production enhancement and fisheries value is highly variable both within and between studies and habitat types (Aburto-Oropeza et al., 2008; Grabowski and Peterson, 2007; Jänes et al., 2020; Lai et al., 2020; Peterson et al., 2003; Rahman et al., 2018; Raoult et al., 2018; zu Ermgassen et al., 2016). However, inter-study comparisons between aquaculture and natural habitats are difficult due to methodological differences (including our decision to subtract the value of displaced species), and there is some evidence that oyster farms in particular may offer similar habitat value to restored oyster reefs. DeAlteris et al. (2004) reported much higher densities of juvenile tautog, tomcod and other species at oyster cages than in seagrass or bare sediment, while Powers et al. (2007) found that fish communities within clam leases were more similar to those in seagrass than bare sediment. Indeed, oyster farms may play a similar role to restored oyster reefs: Erbland & Ozbay (2008) reported similar or higher densities of targeted species around oyster cages than restored oyster reefs, while rates of bait consumption may be similar on average at oyster farms and oyster reefs (Lefcheck et al., 2021). In general, it is clear that observed fish populations at farms have more in common with structured than unstructured natural habitats.

Natural mortality, *M*, is a critical parameter for production estimation, yet is difficult to measure and can vary considerably within species across habitats and locations (Hamel, 2014). Lacking evidence to the contrary, we assumed equal mortality in farm and reference habitats, which could lead to overestimation of production enhancement if farm habitats tend to attract recruits but do not support their survival and growth, forming an ecological trap (Swearer et al., 2021). This has long been a point of contention for artificial reefs (Brickhill et al., 2005; Folpp et al., 2020; Osenberg et al., 2002; Roa-Ureta et al., 2019), and remains largely unanswered for aquaculture habitats (Barrett et al., 2019). We

**Table 5**
Summary of key limitations relevant to the valuation of nitrogen removal and fisheries enhancement benefits, together with suggestions for future research targeting knowledge gaps.

| LIMITATION | TARGETS FOR FUTURE RESEARCH |
|---|---|
| **General** | |
| Limited scope omits a variety of aquaculture-environment interactions and potential trade-offs between ecosystem services. | Estimate and monetise the effects of aquaculture on ecosystem services not assessed here. |
| No quantitative assessment of temporal dynamics. Aquaculture-environmental interactions and values to society depend on past, present and future ecosystem states and uses. | Consider the impacts of climate change and other influences on ecosystem service value. |
| **Nutrient removal** | |
| Low sample size of independent estimates for most farmed bivalve species (Crassostrea sp. and Mytilus sp. over-represented), and all farmed seaweed species. | Data-deficient farmed taxa. |
| Geographic distribution of study sites is not representative of the distribution of global aquaculture production (Europe and North America over-represented). | Data-deficient regions and/or a synthesis of relevant non-English language research for use by international researchers. |
| Effects of biodeposition and burial of nitrogen poorly understood, especially over long time-scales (for that reason, not assessed in the present study). | Long term fate of biodeposited and buried nitrogen. |
| Synthesis of valuations only includes valuations from Europe and North America, all within areas where nutrient removal is a management priority (i.e. Baltic Sea, human-impacted estuaries in the USA); likely to overestimate global willingness-to-pay for nutrient removal services. | Nutrient removal value in data-deficient areas (e.g. Asia, Africa), and/or mapping of global aquaculture production relative to nutrient status. |
| Benefit transfer, for the reasons above, is likely to overestimate global value. | Global meta-analytic benefit transfer function for coastal nutrient removal. |
| **Habitat provision** | |
| In some cases, converted relative abundance to density units by estimating the effective area sampled; farm and reference habitats were treated equally and resulting densities were not significantly different from those initially reported as density units. | Where possible, use sampling methods that provide abundance per area (i.e. density). |
| Estimated the age 0 + juvenile fraction of the sampled population based on size-selectivity of the sampling methods and a plausible size-frequency distribution reconstructed using species life-history parameters. | Where possible, provide size- or age-frequency data (e.g. supplementary data files). |
| Life-history parameters are not location-specific, and for data-deficient species, are taken from related species or estimated via available parameters. | N/A |
| Production enhancement assumes equal mortality and growth rates at farm and reference sites; assuming 10% higher mortality at farms results in a smaller but still positive enhancement effect. | Empirical studies of fish mortality in aquaculture and natural habitats (tagging or age-structure methods). |
| Valuation of enhanced production assumes that all individuals that reach landable size are caught at the moment before natural mortality. | N/A |
| Recreational fishing values may not apply outside wealthy nations (where recreationally caught fish are highly valued); commercial ex-vessel values are used instead for global calculation. | N/A |

tested the effect of increasing *Mmean* by a small arbitrary value (10%) at farm sites but not reference sites – this penalty reduced but did not cancel the positive effect relative to unstructured reference habitats. For juvenile fish that use farm structure as shelter from predators, disturbance during harvesting or other farm activities may be the factor most likely to increase *M* in farms relative to reference sites (Dumbauld et al., 2009). Behavioural responses to disturbance vary widely among fish age classes and taxa (Kulbicki, 1998), and in general, we cannot determine whether any increased risks during farm activities outweigh the benefits for fish that recruit to aquaculture habitats. Many farm-associated species can be tagged to track loss or emigration of juveniles from farm and reference habitats (e.g. visible implant elastomer (VIE) tags: Astorga et al. 2005). Among the few studies that have done so, Tallman and Forrester (2007) reported lower growth but also lower disappearance rates of tagged scup at oyster farms compared to rocky reefs, consistent with lower mortality and/or higher site fidelity. Adult bream tagged in Japan remained within oyster farm habitat for most of the 2.5-month tracking period (Tsuyuki and Umino, 2017), indicating that fish perceive value in residing and presumably feeding around farms. Incentivising farm management practices that provide spatial and temporal habitat continuity (e.g. partial harvest with overlapping production cycles) may help to avoid mortality risk for vulnerable species and life stages residing within farms.

### 5.3. Supporting growth of ecologically beneficial aquaculture

#### 5.3.1. Managing spatial conflicts

At present, marine aquaculture may have a global footprint of up to ∼ 23000 km$^2$ of seabed (Bugnot et al., 2021). This area, which includes seaweed, mollusc, finfish and crustacean aquaculture, makes up 71% of the total area devoted to marine infrastructure (Bugnot et al., 2021), but is dwarfed by, for example, the spatial impact of trawl fisheries, which impact millions of square kilometres of continental shelf (Amoroso et al., 2018). Aquaculture affects a small area by comparison, but can pose difficulties in balancing spatial management when it takes place in busy nearshore waters and may exclude most other marine and maritime activities from the farm footprint, whether due to management/regulatory approaches or avoidance of farm infrastructure. As a result, overlapping objectives in spatial use remains a significant barrier to aquaculture growth in parts of the world (Sanchez-Jerez et al., 2016), despite large areas of the coastal shelf having suitable environmental, regulatory and human health parameters (Oyinlola et al., 2018; Theuerkauf et al., 2019). Even where direct spatial conflict is minor, potentially profitable farms are not established because of regulatory inefficiencies and negative public perception (Beckensteiner et al., 2020), while aquaculture for food production is precluded in some areas due to health risks posed by accumulation of algal toxins, bacteria or heavy metals (Grattan et al., 2016).

The long-term contribution of non-fed aquaculture depends on improved marine spatial planning policies to facilitate siting of farms where the produce is safe to consume, and where farms have neutral or net positive local environmental effects and low conflict with other spatial uses (Bricker et al., 2016; Gimpel et al., 2018; Sanchez-Jerez et al., 2016). Examples of competing spatial uses include areas targeted for protection or restoration of coastal submerged aquatic vegetation or other habitats, which in some cases may be impacted by aquaculture (Ferriss et al., 2019). Potential restorative effects of aquaculture may be compromised if spatial conflicts force the aquaculture sector to seek room offshore, as some have proposed (Gentry et al., 2017b; Lester et al., 2018). Specifically, nutrient uptake in offshore waters will not address inshore eutrophication issues, and could even be viewed as a negative impact given the nutrient-depleted status of typical offshore pelagic ecosystems. The value of offshore farms as fish habitat also remains highly uncertain (Fernandez-Jover et al., 2009; Hallier and Gaertner, 2008). Offshore aquaculture also brings a range of challenges, including nutrient-limited growth of bivalves and seaweeds, more expensive infrastructure, and higher costs arising from fuel use and the need to ensure a safe working environment. As a result, many economically viable scenarios for offshore aquaculture require either a significant undersupply of seafood leading to higher market prices (Knapp, 2013), or for farms to be placed too close to shore to effectively mitigate spatial conflict (e.g. $< 3$ nm or $\leq 30$ m depth: Froehlich et al., 2017a,b). As long as nearshore aquaculture is able to meet demand, the most promising sustainable development pathway for bivalve and shellfish aquaculture may be optimal siting within the nearshore zone, together with farming practices that maximise provision of ES. Exceptions to this may occur where climate change reduces the suitability of nearshore waters for farming (Callaway et al., 2012; Froehlich et al., 2018). In some contexts, climate change may increase primary productivity, but estuarine and nearshore waters are also more prone to fluctuations in temperature and dissolved oxygen under climate change scenarios (Filgueira et al., 2016; Guyondet et al., 2015). Where it is not feasible to switch to a more resilient species, aquaculture may need to gradually move offshore to mitigate climate stress.

### 5.3.2. Policy and economic mechanisms

If directed to the most ecologically beneficial forms of aquaculture, capital investment can help to ensure that the seafood supply gap is closed in a sustainable and responsible manner (Cai and Leung, 2017; O'Shea et al., 2019). Investment can take many forms, and past growth in the aquaculture sector has primarily been driven by numerous small businesses and cooperatives supported by capital from conventional lenders or government programs (Engle, 2010). While aquaculture continues to grow, the current trajectory suggests that the status quo is unlikely to close the gap (Cai and Leung, 2017). Market forces will attract investors if demand growth outpaces supply, yet there is also a need for new policy mechanisms to support capital investment from sources that seek to grow the industry responsibly (Costello et al., 2020).

Incidental effects on ES provision are usually externalities, in that no payment is made as compensation for the gain/loss of ES. Internalising those effects would provide an economic incentive to farm in a manner that benefits ES provision. Salzman (2005) identified 5 broad policy tools to promote ES provision by landholders: prescription, penalty, persuasion, property rights, and payment. Each of these tools can, to varying degrees, internalise effects of aquaculture on ES. Regulatory prescription is commonly applied to protect the environment from negative effects of aquaculture. It could also be used to mandate specific ecologically beneficial practices, but it would need to be well-researched, targeted and enforced. Poorly targeted regulations are often unpopular, inefficient, and could discourage much-needed capital investment. Similarly, financial penalties (taxes or fees) can be applied to shift farmers from undesirable to desirable practices, but bring many of the same downsides as regulatory prescriptions, and moreover, can be a backward step if they allow for damaging activities that would otherwise have been prohibited by legislation. Persuasion and education can change behaviour without coercion, for example within environmental best management practice programs (BMPs: Tucker and Hargreaves, 2009), especially if the change is not costly for the farmer. Property rights are applied differently in mariculture vs. land-based farming, as mariculture operations are rarely if ever granted freehold of marine space. Instead, they are granted concessions or leases to use a common resource (sometimes exclusively). Nonetheless, comparable incentives can still be applied, such as additional space allocated for operations deemed to be ecologically beneficial (Bosch et al., 2010). So-called 'green concessions' in Norwegian salmon aquaculture demonstrate the feasibility of this approach (Hersoug, 2015). Lastly, payments can be made in the form of direct subsidies, tax/fee reductions or grants to support beneficial forms of aquaculture (Bosch et al., 2010).

Payments may be the most effective tool where applicable, as they can combine market value and regulatory drivers. Offset trading markets provide a useful example, wherein payments by nutrient emitters for nutrient removal services are motivated by regulations that carry penalties for excess nutrient emissions (Ferreira and Bricker, 2016; Lindahl et al., 2005). Existing nitrogen markets are generally administered by—but not otherwise funded by—a regional authority, and are possible because dissolved nitrogen emission and uptake is measurable and a unit of nitrogen has monetary value under the prevailing policy settings. Similar markets could function via voluntary payments for ES provision, motivated by philanthropy or public image (as occurs with carbon offsets). Nutrient trading programs are currently limited in scale and distribution, and only two bivalve species, *Crassostrea virginica* and *Mercenaria mercenaria*, have been approved for inclusion in nutrient management programs in the USA. However, where implemented, they can provide a financial incentive to farm bivalves in areas that will benefit from nutrient removal. For example, Parker (2019) modelled the budgets of oyster farms in Maryland, USA, and found that receiving nutrient payments (typically 2–4% of oyster sales) raised the likelihood of breaking even at small production scales.

It is conceivable that payments could one day be made to compensate farmers for providing habitat in habitat-limited nearshore areas, in support of local restoration or rehabilitation objectives. These could occur as voluntary one-off payments made by governments or non-government entities to promote public good (payments for environmental services, PES: Wunder, 2005), or else could occur within an environmental offset trading program (as with nutrient offsets). Proper accounting of habitat ES gains/losses would require a much better understanding of factors affecting aquaculture habitat quality, but any resulting payments could act as an incentive to farm in a way that improves habitat value, offset any financial costs of wildlife interactions (e.g. depredation: Šegvić-Bubić et al., 2011), and improve societal perceptions of non-fed aquaculture by highlighting ES benefits (Beckensteiner et al., 2020; Froehlich et al., 2017a). In lieu of direct payments or regulatory incentives, eco-certification programs that promote environmental BMPs could include ecologically beneficial aquaculture, potentially attracting a higher price for certified produce (Kuminoff et al., 2008). One advantage of this approach is that end consumers in wealthy economies can indirectly subsidise certified producers in less wealthy economies, who may not otherwise have access to financial incentives. Importantly, any such mechanisms should complement, not replace, those targeted at ensuring equitable access to marine environments and conservation or restoration of natural habitats.

Finally, impact investors could be a key driver of ecologically beneficial aquaculture. Impact investments could include microloans to farmers, venture capital for beneficial start-ups, 'green bonds' to raise funds for large projects (Dupont et al., 2015), or contracts for differences (CFDs) to assist aquaculture businesses transitioning to more ecologically beneficial farming practices. While impact investing is increasingly popular both in concept and practice, investors have been struggling to develop clear metrics by which social or environmental 'impact' can be demonstrated and assessed (Clarkin and Cangioni, 2016; Ormiston et al., 2015). Monetisation of ES contributes to solving this problem, as translating benefits into economic terms makes it possible to compare relative impact across disparate investment options. This is the approach now taken by the 2 billion USD Rise Fund (TPG Growth) and others (Addy et al., 2019; Gandhi et al., 2018). By monetising the ES provided by bivalve and seaweed aquaculture, we have highlighted the potential value gained if aquaculture can be optimised for ecosystem benefits alongside the primary goal of generating profits. In doing so, we also provide a quantitative basis for policy and regulatory frameworks to consider and incentivise beneficial effects, as well as a starting point from which impact investors can assess the potential impact of making farm-level investments into bivalve or seaweed aquaculture.

### 5.3.3. Conclusion

Quantifying and monetising aquaculture's ES contributes to a more holistic view of the various costs and benefits of food production (Weitzman, 2019), and provides a common currency by which to compare the benefits of so-called 'restorative' aquaculture to other remedial activities. More work is needed to account for trade-offs between all relevant ES, test the transferability of values across global regions, and identify specific farm management practices that are most likely to deliver net benefits across the board.

## Declaration of Competing Interest

RJ serves on the Board of Directors for the Global Aquaculture Alliance. The authors declare that they have no other known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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## Appendix A. Supplementary data

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## *Further reading*

FAO Fisheries and Aquaculture Department, 2020b. FishStatJ - Software for Fishery and Aquaculture Statistical Time Series.