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Comparative analysis of modeled nitrogen removal by shellfish farms

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ABSTRACT

The use of shellfish aquaculture for nutrient removal and reduction of coastal eutrophication has been proposed. Published literature has indicated that nitrogen contained in harvested shellfish can be accurately estimated from shell length:nitrogen content ratios. The range of nitrogen that could be removed by a typical farm in a specific estuarine or coastal setting is also of interest to regulators and planners. Farm Aquaculture Resource Management (FARM) model outputs of nitrogen removal at the shellfish farm scale have been summarized here, from 14 locations in 9 countries across 4 continents. Modeled nitrogen removal ranged from 105 lbs acre^{-1} year^{-1} (12 g m^{-2} year^{-1}) to 1356 lbs acre^{-1} year^{-1} (152 g m^{-2} year^{-1}). Mean nitrogen removal was 520 lbs acre^{-1} year^{-1} (58 g m^{-2} year^{-1}). These model results are site-specific in nature, but compare favorably to reported nitrogen removal effectiveness of agricultural best management practices and stormwater control measures.

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1. Introduction

Eutrophication of the estuarine and coastal environment is one of the major challenges facing marine resource managers today. Excess nutrients from land, atmospheric and benthic sources have been linked to a host of problems, including loss of key habitats, algal blooms, hypoxia, and fish kills (Bricker et al., 1999, 2008). The systematic reduction of land-based loading of nutrients to the coastal environment has been a keystone of ecosystem management for decades. Many state/federal nutrient reduction programs have focused on nitrogen as the primary nutrient of interest for improving coastal water quality, largely because nitrogen has historically been identified as the nutrient most often limiting primary production of coastal marine ecosystems (Thomas, 1966; Ryther and Dunstan, 1971; Vince and Valiela, 1973; Malone et al., 1996). A variety of different approaches have been employed to reduce nitrogen inputs, depending on the source. Point sources of nitrogen have been addressed through programs such as upgrading wastewater treatment plants (Greening and Janicki, 2006; Latimer et al., 2014) and managing waste streams from concentrated animal feeding operations (http://www.epa.gov/region7/water/cafo/). Nonpoint sources of nitrogen, due to their diffuse nature, are more challenging to address on an ecosystem scale. Typically, runoff from agricultural fields and stormwater are two of the major focus areas of management plans for reducing nonpoint sources of nitrogen (Collins et al., 2010; Houle et al., 2013; Passeport et al., 2013).

The use of shellfish aquaculture and/or restoration for nutrient removal has been proposed in Europe and the United States (Newell, 2004; Lindahl et al., 2005; Lindahl, 2011; Kellogg et al., 2013; Luckenbach et al., 2013). Shellfish remove particulate nutrients, contained in plankton and organic detritus, directly from the water through their filter-feeding activities. Potential nitrogen removal mechanisms include the incorporation of nitrogen into animal tissue and shell during growth, enhancement of denitrification activities under shellfish reefs or aquaculture gear, the burial of shell as reefs grow, and in the case of aquaculture operations, through the harvest of cultivated shellfish (Lindahl et al., 2005; Higgins et al., 2011; Piehler and Smyth, 2011; Newell and Mann, 2012). The use of shellfish aquaculture for nutrient removal purposes has been termed "nutrient bioextraction" by scientists and resource managers in Long Island Sound, USA (Rose et al., 2014).

In order for shellfish aquaculture to be included as part of a comprehensive approach to nutrient management, it is necessary to be able to measure the amount of nitrogen that a shellfish farm removes from the local environment. Higgins et al. (2011) generated a predictive equation for Chesapeake Bay oyster nitrogen content based on shell length. Grizzle and Ward (2011), Carmichael

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et al. (2012) measured average nitrogen content of harvestable size oysters in the Great Bay estuarine system in New Hampshire and Cape Cod estuaries in Massachusetts, respectively. Knowledge of the quantity of animals and shell lengths would allow a reasonable estimation of total nitrogen removal post-harvest (e.g., Luckenbach et al., 2013). These types of equations are specific to a waterbody, species, and type of cultivation practice, but could be generated for other programs that are interested in quantifying nitrogen removal based on harvest information.

In addition to predictions of nitrogen removal on a per-animal basis, the nitrogen removal typical of a shellfish farm in a specific estuarine or coastal setting would also be of interest to regulators and planners. There are several farm-scale aquaculture models available that use information about local water quality, shellfish physiology, and local aquaculture practices to generate estimates of farm production (e.g. (Ferreira et al., 2007; Silva et al., 2011; Hawkins et al., 2013)). These models include assimilation and growth of the whole population within a farm, to estimate the total nitrogen removal from the water column by the farm population, which results in larger nitrogen removal numbers than those estimates that are based on harvest only (since only a subset of a farm population is harvested each year). The models do not include enhanced sediment denitrification or burial in their calculations. These models have been used for a variety of purposes, including optimizing culture practices, siting new farms, and estimating impacts to local eutrophication. As estimators of nitrogen removal from shellfish cultivation at the local scale, these models could be useful to resource managers who are interested in the potential for shellfish aquaculture to contribute to a larger nutrient management program. These models may be particularly helpful in areas with limited or no shellfish aquaculture, for which data on typical harvest numbers may not exist or may be too limited to make predictions with confidence (Silva et al., 2011).

We summarize here available information about the Farm Aquaculture Resource Management (FARM) model, which has been applied in China, Chile, the United States, and several European countries (Ferreira et al., 2007, 2008, 2009; Silva et al., 2011; Ferreira et al., 2012b; Bricker et al., 2014; Saurel et al., 2014). Previous applications of the FARM model have used a variety of commonly cultivated shellfish species and different kinds of aquaculture practices (i.e. longlines, bottom culture, intertidal). Nitrogen removal is calculated as part of the FARM model output, but has not been typically reported in metrics common to resource management, i.e., pounds of nitrogen acre^{-1} year^{-1} or grams of nitrogen m^{-2} year^{-1}.

2. Materials and methods

The FARM model combines physical and biogeochemical models, shellfish growth models, and eutrophication screening models. The FARM model determines shellfish production, conducts an eutrophication assessment and evaluates farm-related impacts on biodeposition and sedimentation rates. It also provides a marginal analysis of farm production potential and profit maximization, while assessing potential credits for carbon and nitrogen trading (Ferreira et al., 2007, 2009). FARM simulates processes at the farm scale (about 100–1000 m^2), but may also consider smaller areas if required. The general layout for the model is shown in Fig. 1, and is applicable to suspended culture from rafts or longlines as well as to bottom or trestle culture; the model can additionally simulate integrated multi-trophic aquaculture (IMTA), for instance through co-cultivation of macroalgae and/or finfish (Ferreira et al., 2012a).

Model input requirements have been minimized, since the model is aimed at the shellfish farming community and local managers. Inputs fall into three classes: aquaculture practices, suspended food entering the farm, and environmental parameters (www.farmscale.org). Data required for aquaculture practices include species, seed size (weight), stocking density, Julian date of typical first seeding, length of typical cultivation cycle, average mortality, farm size, and culture structure (i.e. bottom, suspended). Data requirements for suspended food entering the farm include chlorophyll, total particulate matter, and particulate organic matter, which allows for estimation of phytoplankton as well as the detrital component of available food. Environmental parameters include temperature, salinity, and current speed and direction (i.e., one way, or inverts with tide). Dissolved inorganic nitrogen and dissolved oxygen are used to determine impacts of the farm on local water quality. Monthly data points for at least one year are required for the food and environmental parameters, with the exception of current speed, which is input as peak flow at neap and spring tides.

FARM uses a range of species-specific shellfish individual growth models to simulate each species represented in the model. The individual growth models use a net energy balance approach (Silva et al., 2011) based partly on functions published in the literature with new formulations and parameterization as appropriate to calibrate the models to each study site. The equation used for calculating net energy balance is:

$$\mathrm{NEB} = C - (F + R + E)$$

where NEB = net energy balance deposited as tissue; C = energy ingested, a function of feeding rate, pseudofeces production, and assimilation; F = energy lost as feces; E = energy excreted; R = energy expenditure through metabolism and spawning.

Growth of the Eastern oyster, *Crassostrea virginica*, Pacific oyster, *Crassostrea gigas*, the Good Clam, *Venerupis decussata*, and Manila clam *Venerupis philippinarum* were simulated based on the AquaShell™ Framework (Silva et al., 2011). The Chinese oyster, *Ostrea plicatula*, and blue mussel *Mytilus edulis* were simulated using the ShellSIM model (http://www.shellsim.com) developed from that of Hawkins et al. (2002, 2013). Growth of the Mediterranean mussel *Mytilus galloprovincialis* was simulated using the growth model of (Brigolin et al., 2009).

The individual models of shellfish growth are integrated into a population dynamics framework using well established equations (e.g. (Nunes et al., 2003; Nobre et al., 2005)) to simulate the biomass production of the whole population. The growth rates for individual shellfish that make up a number of weight classes within the population are calculated on the basis of food supply, environmental parameters, and mortalities, as noted above to give an estimate for population growth.

Model outputs include estimation of the shellfish density for greatest sustainable yield of market-sized animals within a given time period (carrying capacity) and the mass of carbon and nitrogen removed through uptake of phytoplankton and detritus by shellfish filtration, and net nitrogen removal accounting for undigested matter (i.e. feces), excretion, and mortality. Carbon removal is calculated from the energy mass balance, accounting for the portion of food assimilated into tissue. Carbon units are converted to nitrogen based on the Redfield ratio. Note that assimilation by the whole population is used for the calculation of removal, not just the harvestable size oysters since once assimilated into tissue and shell carbon and nitrogen are no longer available to support phytoplankton growth. Nitrogen removed can be converted to population equivalents and to substitution cost of land-based nutrient removal. Changes in dissolved oxygen and chlorophyll a are also examined and the model provides an analysis of impacts of biodeposition on sediment organic enrichment. Cultivation practices, such as stocking density, bottom or long line placement of shellfish can affect removal rates, as can the placement of the farm in locations of faster or slower current speeds. For example, there is an

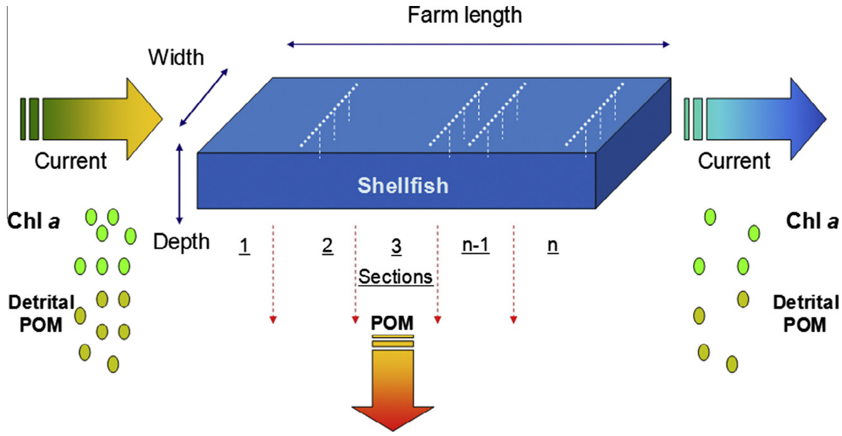

Fig. 1. Farm layout (rope and bottom culture) for the Farm Aquaculture Resource Management Model. Chl *a* = chlorophyll *a*, POM = particulate organic matter. Reprinted from *Aquaculture*, Volume 264, J.G. Ferreira et al. "Management of productivity, environmental effects and profitability of shellfish aquaculture – the Farm Aquaculture Resource Management (FARM) model", p. 162, Copyright 2007, with permission from Elsevier.

optimum seeding density below which production can be increased but above which there will be draw down of dissolved oxygen and potentially undesirable increase in ammonia and biodeposits (Ferreira et al., 2007).

3. Results and discussion

Table 1 summarizes the information compiled for FARM model outputs of nitrogen removal by an individual shellfish farm, which include 14 locations in 9 countries across 4 continents. A more complete list of model outputs can be obtained in Supplemental Table S1. Species analyzed include *C. gigas* (5 locations), *C. virginica* (1 location), *M. edulis* (2), *M. galloprovincialis* (2), *O. plicatula* (1), *V. philippinarum* (1), and *V. decussata* (1). Cultivation practices included ropes, intertidal trestles, a combination of rope and intertidal, longlines and bottom culture. The area of cultivation modeled, or farm size modeled, ranged from 6.0×10^3 m^2 (1.5 acres) to 2.0×10^6 m^2 (50 acres).

Modeled nitrogen removal ranged from 105 lbs acre^{-1} year^{-1} (12 g m^{-2} year^{-1}; *M. galloprovincialis* in Chioggia, Italy) to 1356 lbs acre^{-1} year^{-1} (152 g m^{-2} year^{-1}; *V. philippinarum* in Samish Bay, USA). Mean nitrogen removal across all locations and species was 520 lbs acre^{-1} year^{-1} (58 g m^{-2} year^{-1}).

Several aspects should be noted when considering these FARM model results. First, the model is intended for local or farm-scale simulations and does not account for potential interactions among neighboring farms. For example, if several farms were located in a relatively small waterbody, or if a new farm were sited just upstream of the modeled farm, this could result in depletion of local food resources. Depletion of local food resources through multi-farm interactions should be addressed by system-scale models, and would not be adequately described by FARM model *status quo ante* inputs and would thus potentially result in overestimates of actual production. Second, direct comparison of rates of removal among species and even between the same species in different locations must be done with caution, because nitrogen removal in the model depends on a combination of environmental characteristics of the site, physiological characteristics of the species studied, and cultivation practice employed. Nitrogen removal rates depend on species-specific filtration, pseudofeces production, and assimilation/growth rates, are influenced by food concentration (POM and chlorophyll a), and supply (current speed), by temperature and salinity, and also by culture structure and shellfish seeding density. This complex combination of factors controlling removal makes paired comparisons challenging to interpret, and is why we have limited our analysis to the minimum, maximum, and mean observed across all studies. Third, these removal numbers are also thus site-specific, and estimates of nitrogen removal generated for one location should not be used to predict nitrogen removal at a different site, waterbody, or region.

We were interested in comparing the range of modeled nitrogen removal rates from shellfish farms to establish best management practices for the control of agricultural and stormwater sources of nitrogen. Nitrogen removal efficacy has been reported using a variety of metrics, including the reduction in nitrogen

Table 1
Nitrogen removal outputs by farm scale model simulations of shellfish aquaculture farms.

Location	Species cultivated	Cultivation practice	Modeled area in cultivation (m^2)	Annual nitrogen removal (g m^{-2})	Annual nitrogen removal (lbs acre^{-1})
Sanggou Bay China	Pacific oyster *C. gigas*	Rope	32,000	13	113
Huangdun Bay China	Chinese oyster *O. plicatula*	Rope and intertidal	240,000	65	582
Loch Creran Scotland	Pacific oyster *C. gigas*	Intertidal trestles	164,800	23	205
Pertuis Breton France	Blue mussel *M. edulis*	Longlines	2,000,000	65	581
Piran Slovenia	Mediterranean mussel *M. galloprovincialis*	Longlines	18,000	38	343
Chioggia Italy	Mediterranean mussel *M. galloprovincialis*	Longlines	2,000,000	12	105
Ria Formosa Portugal	Good clam *V. decussata*	Bottom	50,000	38	336
Valdivia Chile	Pacific oyster *C. gigas*	Bottom	60,000	70	627
Tornagaleones Chile	Pacific oyster *C. gigas*	Bottom	60,000	85	759
Niebla Chile	Pacific oyster *C. gigas*	Bottom	60,000	60	536
Isla del Rey Chile	Pacific oyster *C. gigas*	Bottom	60,000	64	571
Carlingford Lough Ireland	Blue mussel *M. edulis*	Bottom	6,000	74	661
Potomac River USA	Eastern oyster *C. virginica*	Bottom	12,141	57	507
Samish Bay USA	Manila clam *T. philippinarum*	Bottom and intertidal	22,500	152	1356

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Table 2
Nitrogen removal by agricultural best management practices in the Chesapeake Bay watershed, as approved by the Virginia Department of Environmental Quality (modified from Stephenson et al., 2010).

Approved agriculture BMP	Minimum nitrogen removal (lbs/acre)	Maximum nitrogen removal (lbs/acre)
Early cover crops	0.04	1.10
15% N reduction	1.11	4.21
Continuous no-till	0.71	1.79
15% N reduction + continuous no-till	1.65	5.01
Crop to forest land conversion	3.71	11.58

Table 3
Nitrogen removal by different types of stormwater control measures, installed at the University of New Hampshire Stormwater Center (modified from Houle et al., 2013; note omission of bioretention systems).

Stormwater Control Measure	Annual nitrogen removed (g)	Installation Area (m^2)	Annual nitrogen removal (lbs $acre^{-1}$)
Vegetated swale	0	260	0
Wet pond	8770	299	261
Dry pond	6640	299	198
Sand filter	0	15	0
Gravel wetland	19900	179	991
Porous asphalt	0	523	0

concentration downstream of an installation (mg L^{-1}), nitrogen removal efficiency as a percentage of nitrogen input, and nitrogen removal per volume of installation (g m^{-3}) (e.g., (Collins et al., 2010, Geosyntec Consultants Inc. and Wright Water Engineers Inc., 2010, Passeport et al., 2013). None of these metrics are easily comparable to the FARM model outputs of mass of nitrogen removed per unit area. We found two recent examples of nitrogen removal efficacy reported in mass of nitrogen removed per unit area (Tables 2 and 3); one for state-approved agricultural best management practices in the State of Virginia, USA (Stephenson et al., 2010), and the other for a variety of low impact development (LID) and conventional stormwater management systems (Houle et al., 2013).

The Virginia Department of Environmental Quality has published a guidance document that assigned nitrogen removal rates for approved agricultural best management practices (BMP) (VADEQ, 2008); as cited in (Stephenson et al., 2010). Removal rates varied for each agricultural BMP depending on location within the part of the Chesapeake Bay watershed that is within the state of Virginia. Approved rates were determined for each of five watershed "basins", and within each basin on the western shore of Virginia, for BMPs that were located to the east or to the west of Interstate 95. We have compiled the minimum and maximum annual nitrogen removal (in terms of lbs $acre^{-1}$) for each of the five approved agricultural BMPs in Table 2, in order to provide a range of expected nitrogen removal for that state. Nitrogen removal rates by shellfish farms compare very favorably, on a per-acre basis, to expected nitrogen removal by agricultural best management practices. Agricultural best management practices ranged from 0.04 lbs N removed $acre^{-1}$ $year^{-1}$, for early planted cover crops in Piedmont watersheds west of Interstate 95 in the York Basin, to 11.58 lbs N removed $acre^{-1}$ $year^{-1}$ for conversion of cropland to forest in Coastal Plain watersheds east of Interstate 95 in the Shenandoah-Potomac Basin.

The University of New Hampshire Stormwater Center has implemented a variety of types of stormwater control measures (SCMs) in a controlled setting on their campus, including both conventional and LID designs (Roseen et al., 2009; Houle et al., 2013). This controlled setting has allowed scientists to quantify performance and cost of commonly-used SCMs and compare among the traditional and low impact design categories. Houle et al. (2013) report nitrogen removal by the SCMs in grams per year, and separately report the area of each SCM installation. We have combined this information to calculate annual nitrogen removal in terms of lbs $acre^{-1}$ for comparison to the agricultural BMPs and shellfish farms (Table 3). Bioretention systems were not included in Table 3 since nitrogen removal by installations of different sizes were reported as a single average in the paper, thus, lbs $acre^{-1}$ could not be back-calculated. In general, nitrogen removal rates by stormwater control measures were higher, on a per-acre basis, than those reported for agricultural best management practices. Nitrogen removal rates by shellfish farms also compared favorably to expected nitrogen removal by stormwater control measures. Several stormwater control measures had no detectable nitrogen removal; including vegetated swales, sand filters, and porous asphalt (these measures instead targeted total suspended solids and/or phosphorus). Stormwater control measures that demonstrated nitrogen removal included wet ponds, dry ponds, and gravel wetlands, and annual removal rates were in the same approximate range as those observed for shellfish farms (198–991 lbs $acre^{-1}$).

Although cost-effectiveness of available best management practices was not the focus of this study, we have compiled published information from eight studies and converted to a common currency (i.e., USD lb^{-1} nitrogen removed) (Gren, 2008; Gren et al., 2009; Jones et al., 2010; Stephenson et al., 2010; Jacobsen, 2012; Houle et al., 2013; Hasler et al., 2014; Petersen et al., 2014). Nitrogen removal strategies were grouped into one of six categories: shellfish, agricultural, urban stormwater, wastewater treatment upgrades, wetlands, and other. The range of reported costs for each category is summarized in Table 4. A detailed table including information about location, strategy, cost, and reference for each of the 47 entries is available as Supplemental Table S2. The range of reported cost-effectiveness was several orders of magnitude for each category of best management practices. The range of potential costs for shellfish aquaculture as a nitrogen removal strategy was similar to that of the other best management practices.

None of these nitrogen removal measures will solve the problem of coastal eutrophication alone, and a successful watershed-scale nitrogen management program will likely incorporate aspects of all three categories. Balancing cost, efficacy of nitrogen removal, and available space for implementation will be a common challenge faced by resource managers looking to implement comprehensive nitrogen control plans. It may be the case that agricultural BMPs reduce less nitrogen annually on a per-acre basis than some SCMs or shellfish aquaculture. In many watersheds, however, the acreage available for application of agricultural BMPs is far greater than for either stormwater control measures or for shellfish farms. Therefore, agricultural BMPs could still easily dominate many nutrient management programs. Stormwater control measures can be extremely expensive to implement, and may not be as cost-effective as other options (in terms of dollars per pound of nitrogen removed), but at the local level, many municipalities

Table 4
A summary of reported costs for six categories of nitrogen removal strategies. Reported costs have all been converted to USD lb^{-1} nitrogen. For detailed information about location, practice, cost, and reference, please see Supplemental Table S2.

Strategy	Cost (USD/lb N)
Shellfish	5.7–150
Agricultural	0.1–470
Urban stormwater	30–3629
Wastewater treatment upgrades	0.5–7610
Wetlands	0.60–214
Other	2.8–218

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may have limited other options when faced with implementation of EPA Phase II stormwater regulations. Lampe et al. (2005) noted several factors that could reduce costs for stormwater control measures, including new construction versus retrofitting, a local workforce experienced in SCM construction, environmental factors like rainfall and soil type, and degree of maintenance required after installation. It is possible that as stormwater control measures become more commonplace, costs for installation and maintenance may decrease. Implementation of shellfish aquaculture for nutrient management purposes will likely be limited by spatial constraints in the coastal zone. It is not unusual for state-level management plans for shellfish aquaculture to limit leased areas to a few percent of the total seafloor. For example, the Rhode Island Coastal Resources Management Council established a 5% cap on the area of Rhode Island's estuaries that could be used for aquaculture, despite ecosystem models indicating a much higher ecological carrying capacity (Byron et al., 2011). Recent legislative action to redevelop the shellfish aquaculture industry in Delaware has limited leased acreage to 5% of the seafloor in two bays, and 10% of the seafloor in a third bay (Schwartzkopf et al., 2013). Acreage devoted to aquaculture leases faces competition by many recreational and commercial users. Additional pressure can come from coastal homeowners and/or communities that do not want aquaculture in their nearshore waters.

In closing, it is worth noting that shellfish can provide a number of other ecosystem services in addition to nitrogen removal, and nutrient management plans that include shellfish aquaculture will likely receive many of these ancillary benefits. Oyster reefs or mussel ropes may increase provision of habitat for other macrobenthic species by an order of magnitude (Segvic-Bubic et al., 2011; Kellogg et al., 2013). Shellfish may also increase photic depth (Pollack et al., 2013), potentially helping restore submerged aquatic vegetation (SAV), which in turn provides oxygenation of bottom water and habitat for juvenile fish. These services, together with the supply of goods that generate additional local income and employment, and reduce dependency on imported seafood, leverage the positive externalities of shellfish aquaculture as a complement to other methods of nutrient abatement.

4. Conclusions

Our results suggest that nitrogen removal from shellfish farms compares favorably on a per-acre basis to commonly applied best management practices for agricultural and stormwater runoff. It is unlikely that any single nitrogen removal mechanism will be able to solve the problem of coastal eutrophication, and resource managers will most likely have to incorporate many kinds of nitrogen removal strategies in order to attain water quality standards. The combination of nitrogen removal with other ecosystem services provided by shellfish farms makes them a good candidate for inclusion in comprehensive nutrient management programs.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.marpolbul.2014.12.006.

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