Environmental Impacts of Shellfish Aquaculture: Filter Feeding to Control Eutrophication

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Abstract

In many areas, coastal residents and others oppose establishment of bivalve molluscan aquaculture projects on the basis of perceived negative environmental impacts. Often overlooked are positive environmental impacts of shellfish aquaculture that can potentially mitigate the impacts of other anthropogenic activities. Filter feeding by populations of bivalve mollusks is reviewed with respect to their ability to act as an estuarine filter, increase clarity of coastal waters and facilitate the removal of nitrogen and other nutrients from eutrophic coastal waters. Most species of cultured bivalve mollusks clear particles from waters at rates of 1 to 4 L/h, and populations of shellfish in healthy assemblages can filter a substantial fraction of the water in coastal estuaries on a daily basis. Actively growing shellfish incorporate nitrogen and other nutrients into their tissues as they grow. On average, 16.8 g of nitrogen is removed from estuaries for every kilogram of shellfish meats harvested. In addition to removal of nutrients through shellfisheries and molluscan aquaculture, shellfish beds may act to promote removal of nitrogen from estuaries by increasing organic nitrogen deposition to the sediments that stimulate denitrification processes. It is suggested that shellfish restoration projects and establishment of small-scale molluscan shellfish aquaculture operations may mitigate the effects of coastal housing development or other activities that promote excessive coastal eutrophication.

Introduction

The input of inorganic nutrients such as nitrogen and phosphorus into coastal embayments and estuaries and resultant eutrophication is seen as a significant environmental problem that is expected to intensify as coastal development continues (Nixon 1995). There is good evidence that terrestrial inputs of nitrogen and other nutrients into estuaries either by surface runoff or groundwater discharge were considerably less in pre-colonial times. In fact, evidence suggests that most of the nutrients entering coastal embayments and estuaries such as Buzzards Bay or Narragansett Bay may have been from offshore upwelling (Nixon 1997). Terrestrial anthropogenic sources of nutrients in southern New England have been demonstrated to include point sources such as community sewage treatment facilities (Nixon et al. 1995), individual sewage disposal systems (Postma et al. 1992), as well as non-point sources, including fertilization and other agricultural practices (Gold et al. 1989), management practices for home lawns (Morton et al. 1988), and other runoff and non-point sources (Nixon et al. 1995). There is evidence that some poorly sited finfish aquaculture projects have contributed to coastal eutrophication (e.g. Gowen and Bradbury 1987).

There is a growing body of evidence that coastal eutrophication is resulting in changes in estuarine and coastal benthic communities (Pearson and Rosenberg 1978; Valente et al. 1992; McClelland and Valiela 1998; Newell and Ott 1999). Coastal eutrophication is implicated in increases in the duration and intensity of phytoplankton blooms that increase benthic shading, leading to loss of seagrasses and other submerged aquatic vegetation (Kemp et al. 1983; Twilley et al. 1985; Short and Burdick 1996). Additionally, there is growing evidence that increased nutrient inputs can result in increases in the length and duration of hypoxic (low oxygen) events in estuarine waters (e.g. Officer et al. 1990), that may have very deleterious effects on estuarine community structure.

In order to effectively manage the inputs of nutrients into coastal water and control eutrophication, a number of management strategies have been proposed or are in the process of implementation. Typically these management strategies involve source reduction of nutrients, for example tertiary treatment or denitrification systems have been proposed for community sewage treatment facilities (e.g. Arundel 2000), or the use of constructed wetlands as a strategy for nutrient removal (e.g. Frostman 1996). Innovative individual sewage disposal systems have been designed to remove nitrogen from domestic wastewater through denitrification processes (Gold et al. 1992), and such systems are gaining greater acceptance by communities throughout the southern New England Region (e.g. Lucht et al. 1998; Douglas et al. 2000). In addition to strategies to reduce the amount of nutrients at the source, these community wastewater management plans have included analyses of soil types and assessment of environmentally critical areas in the process. Recently, projects to restore submerged aquatic vegetation in coastal areas have begun as a means to mitigate or counteract the effects of coastal eutrophication (*e.g.* Davis and Short 1997).

It has long been known that beds of bivalve mollusks have the ability to clean the water through their filter feeding, and it is well known that many once thriving coastal shellfish beds have been depleted through overfishing or other means. The purpose of this paper is to review the existing literature on the filter feeding by bivalve mollusks to develop a case for the development of shellfish aquaculture projects or the restoration of estuarine and coastal shellfish beds as a complementary part of integrated community coastal water management strategies.

Bivalves as Filter Feeders

All of the economically important bivalve mollusks derive most of their nutritional needs from filtering particles from the water (Jorgensen 1966). These filtered particles include suspended silt and clay particles, phytoplankton, and detritus particles. The bivalves can actively sort particles according to their nutritional value, and they ingest food particles and release the rejected particles as mucous-coated masses called pseudofeces that are deposited to the seabed (Reid 1981). There is also some evidence that bivalves can remove dissolved organic nitrogen (DON) in the form of free amino acids directly from seawater (Manahan et al. 1982). The rates of particle filtration are species specific and size specific, but typical filtration rates fall in the range of 1 to 4 L/h for individual animals (Haven and Morales-Alamo 1970; Mohlenberg and Riisgard 1979; Doering and Oviatt 1986), and many species of bivalves are capable of removing particles in the 1 μm size range, or particles in the size range of most bacteria (Palmer and Williams 1980; Wright et al. 1982).

Bivalve Assemblages Act to Modify their Own Environment

Since Karl Moebius' (1880) pioneering work in oyster reef ecology, it has been known that, "Any change in the relative factors of the bioconose [community] produces changes in other factors of the same. If, at any time, one of the external conditions of life should deviate for a long time from its ordinary mean, the entire bioconose or community would be transformed. It would also be transformed if the number of individuals of a particular species increased or diminished through the instrumentality of man, or if one species disappeared from, or a new species entered into the community."

Since individual bivalves have the capability of filtering anywhere from 1 to 4 liters per individual on an hourly basis, communities of bivalves have the capability of filtering considerable volumes of water. In a classic theoretical exercise, Newell (1988) estimated that in 1870 when massive oyster reefs were intact in the Chesapeake Bay it took a mere 3.3 days for the oysters to filter the entire volume of the bay. By 1988, after decades of overfishing, oyster diseases and pollution had reduced oyster populations to only a small fraction of their original numbers and the time for the oysters to filter the bay had increased to 325 days.

In a similar exercise, Rice et al. (2000) estimated that northern quahogs in the Providence River section of Narragansett Bay are able to filter a considerable amount of water. The standing crop of quahogs in the Providence River averages 9.1 clams/m^2 or about 26,400 metric tons (Figure 2.3). The population of quahogs, however, is composed of mostly older adults with valve lengths in excess of 60 mm. The quahogs collectively filter about 1.0×10^7 m^3 of water daily (Table 2.10). The mean tidal excursion of the Providence River is 1.22 m, and the area of the Providence River is 2119.8 ha (URI Environmental Data Center), so the mean tidal prism of the Providence River is 2.58×10^7 m^3. The semidiurnal tide cycle is about 12.5 hours, so on a single tide cycle the population filtration by quahogs represents a water volume equal to 21.3% of the tidal prism.

A number of studies suggest that filtration by populations of bivalves can in fact control the biomass of phytoplankton in the water column. For example, Cloern (1982) and Officer *et al.* (1982) argue that populations of infaunal clams in South San Francisco Bay are responsible for controlling phytoplankton populations in the overlying water column. Several studies have shown that filter-feeding bivalves can increase water clarity, thereby increasing light penetration (Cohen et al. 1984; Peterson and Black 1991).

A mesocosm experiment was performed at the Marine Ecosystem Research Laboratory (MERL) from June to October, 2000 to determine the effects aquacultured oysters have on the environment (J. Mugg & M. Perron University of Rhode Island, unpublished data). The MERL facility is located adjacent to Narragansett Bay with thirteen 13,000-L mesocosm tanks that simulate the environmental conditions of the Bay. Two hundred oysters (≈35 mm in valve length; nominally filtering about 48 L da^{-1} ind^{-1}) were placed into three mesocosms, and three mesocosms were maintained without oysters as controls. Experiments were run with varying rates of water exchange in the tanks ranging from 0% to 100% per day (0 to13,000 L da^{-1}). Several parameters were measured and compared between the two treatments, which included chlorophyll-a, particulate organic and inorganic matter, sedimentation rates, nitrate, ammonia, phytoplankton analysis and growth rates. Diatoms of the genus *Nitzschia* were predominant in mesocosms with oysters, and in the control tanks, *Skeletonema* were dominant. Tanks with oysters consistently showed rates of sedimentation greater than twice the control tanks. So, filter feeders such as oysters in culture may not only increase rates of sedimentation and thus benthic-pelagic coupling, but influence the species composition in the overlying water column as well.

However, this process of filter-feeding acting to simply clear the water and depositing the filtrate on the bottom is not a completely simple process. Some studies suggest that in many instances once filtered material eventually decomposes to regenerate new nutrients, which in turn stimulates new production of phytoplankton (Doering et al. 1986, 1987). Thus aquatic ecosystems with bivalves have a more active exchange process between the water column and the sediments and are able to process a greater amount of nutrients. There is also some evidence that in rare instances, very intensive raft culturing of mussels or other bivalves could lead to enough pseudofecal and fecal carbon loading to create localized benthic hypoxia. In these cases, farms should be managed with reduced stocking densities (Dahlback and Gunnarsson 1981; Tenore et al. 1982).

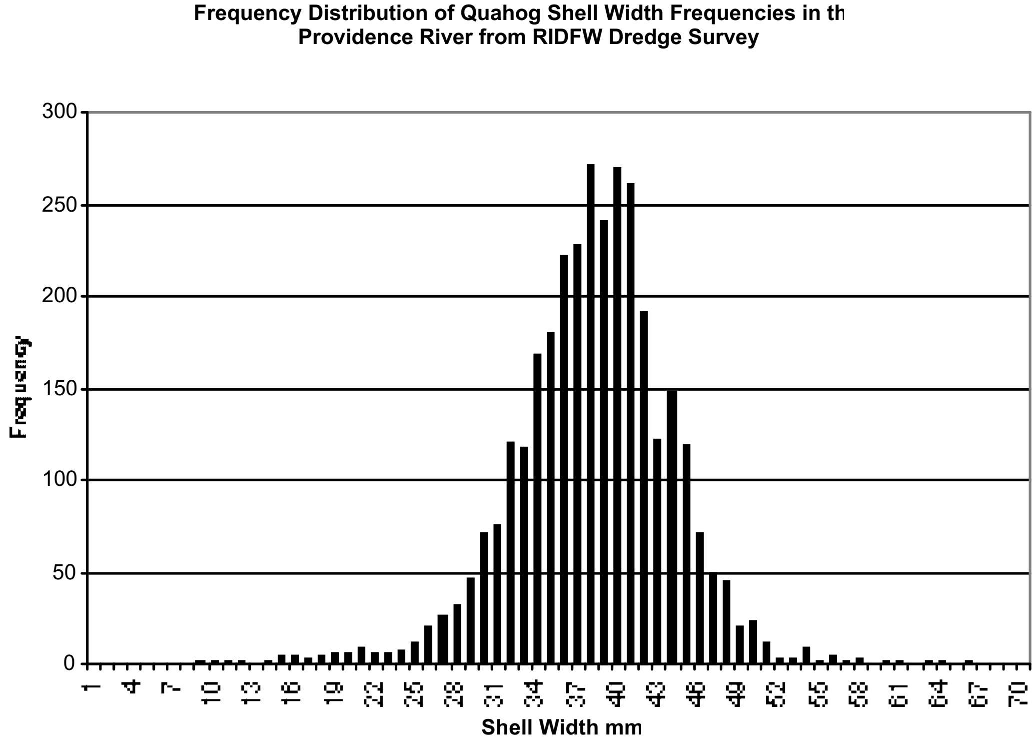

Figure 2.3. Length-Frequency diagram showing quahog population characteristics in the Providence River, Rhode Island. Overall density of quahogs in the Providence River averages 9.1 clams/m^2 or about 26,400 metric tons shell-on weight overall

Table 2.10. Spreadsheet table showing quahog biomass per size class of quahogs *Mercenaria mercenaria*, in the Providence River section of Narragansett Bay. The table also presents quahog filtration rates per size class, and amount of organic nitrogen locked up in quahogs of each size class. Filtration Rate calculated from data in Doering and Oviatt. 1986.

Table 2.10. Biomass Calculations by Size Class with 5 mm group centered on mid-point

Shell Width (mm)	Biomass (metric tons)	Length (cm)	Individual FR (L/hr)	Proportion of total biomass	Quahog numbers	Population FR (cu m/hr)	Organic-N content (tons)
3	0	0.57	0.178	0	0	0	0
8	0.578561	1.52	0.460	2.19E-05	4375	2.013981	0.00167
13	5.12419	2.47	0.735	0.000194	38764	28.53042	0.014793
18	25.93476	3.42	1.008	0.000982	196219	197.8259	0.074868
23	78.96895	4.37	1.277	0.00299	597449	763.4581	0.227968
28	703.7288	5.32	1.545	0.026646	5324295	8229.19	2.031524
33	3580.788	6.27	1.811	0.135583	27091644	49083.04	10.33702
38	9462.755	7.22	2.076	0.358298	71593650	148668.3	27.31708
43	8756.715	8.17	2.340	0.331565	66251971	155044.3	25.27889
48	2878.576	9.12	2.602	0.108994	21778738	56687.34	8.309872
53	491.3499	10.07	2.864	0.018604	3717376	10648.89	1.418429
58	276.9662	11.02	3.125	0.010487	2095470	6549.518	0.799546
63	115.1425	11.97	3.385	0.00436	871197	2949.66	0.332393
68	33.65765	12.92	3.645	0.001274	254565	927.959	0.097163
Totals	26410.29				1.998×10^8	439780.1	76.24121

Nitrogen Removal through Aquaculture and Shellfisheries

Once bivalves have filtered and ingested phytoplankton or detrital particulates, there are four possible pathways for nutrients in the food to be processed: a) incorporated into soft body tissue or shell, b) expelled as pseudofeces before ingestion, c) expelled as feces after ingestion and digestion, and d) excreted as ammonia into the water (Newell 1981). Tissue growth of bivalves in actively growing shellfish beds or in actively managed shellfish aquaculture operations is an effective means of removing nitrogen, phosphorus and other organic nutrients from the water (Rice 1999a). This is not true for bivalves that are mature and not actively growing. In a mature state, the bivalves are generally in a state of nitrogen balance in which the organic nitrogen ingested in the food is equal to the nitrogen defecated or excreted as ammonia. According to U.S. Department of Agriculture Food Tables, shucked oyster or clam meats on a dry weight basis are typically 30 percent protein. Shell material is less that 1 percent protein by weight. Assuming that bivalve soft tissues are 80 percent water and that nitrogen is 14 percent of the weight of typical proteins, it can be concluded that for each kg of shellfish meats harvested from an actively growing shellfish bed or aquaculture farm, 16.8 g of nitrogen would be harvested from the growing waters.

In the case of quahogs in the Providence River of Rhode Island (Figure 2.3), it is recommended that only ten percent of the standing stock of 26,400 metric tons be removed annually by relay or fishing to maintain maximum sustainable yield (Rice et al. 2000). At this rate of exploitation, 8 metric tons of nitrogen would be harvested annually. However, this rate of nitrogen removal is quite miniscule in comparison to the estimated 4580 metric tons of nitrogen discharged into the Providence River annually from sewage treatment plants and surface runoff (Nixon et al. 1995).

In smaller coastal water bodies, shellfish restoration or establishment of aquaculture projects may have a greater impact by removing a greater percentage of the nitrogen and phosphorus reaching the estuary. Because of this, it is often instructive to estimate the number of aquacultured shellfish it would take to mitigate the effects of nitrogen excreted by an average person living in the coastal zone. According to Valiela and Costa's (1988) estimates, an average person excretes and defecates 3.8 kg of nitrogen annually. This being the case, the weight of oyster meats harvested annually to compensate for the nitrogen excreted by the average person would be 3,800g ÷16.8g/kg meat ≅ 225kg of oyster meats. Assuming that oyster meats are harvested at an average 40g, this means that a modest oyster farm producing 5,600 oysters annually would mitigate the nitrogen produced by a single person. Again, this assumes that the shellfish are actively growing. Harvest of mature shellfish from established beds at rates greater than MSY will remove nitrogen on a short-term basis, but it would not be sustainable.

Nitrogen Removal by Deposition and Benthic Processes

Aside from harvest of bivalves as a means to remove nutrients from coastal water bodies, it is likely that the filter feeding and deposition of organic particulates to the benthic sediments is responsible for much greater removal of nutrients. A number of studies suggest that nitrogen can be removed through stimulation of sediment processes. For example, Kelly and Nixon (1984) and Doering *et al.* (1987) showed that increased organic sedimentation or presence of infaunal northern quahogs, *Mercenaria mercenaria,* increase the rates of inorganic nitrogen turnover in sediments suggesting that biogeochemical processes in the sediments are stimulated.

One biogeochemical process of extreme importance in coastal waters is denitrification. For example, about half of the nitrogen deposited into Narragansett Bay from rivers or in sewage discharges is removed via denitrification, and about 35% of the organic nitrogen mineralized in the sediments is removed from the ecosystem as elemental nitrogen (N_2) that eventually diffuses to the atmosphere (Seitzinger et al. 1984). Since 4580 metric tons of nitrogen are discharged into the Providence River annually (Nixon et al. 1995), the estimated rate of denitrification would be 4580 tons/yr ÷ 2 = 2290 tons/yr, which is considerably higher than the recommended 8 metric tons per year nitrogen removal in a well-managed natural harvest shellfishery (Rice et al. 2000).

There is some evidence that the rate of denitrification in marine sediments is a function of the amount of nitrogen supplied. In mesocosm experiments, Seitzinger and Nixon (1985) enriched sediments in experimental tanks with up to 65 times the inorganic nitrogen of control tanks and observed a 2.5 fold increase in the production of elemental nitrogen (N_2) and a hundred fold increase in the production of nitrous oxide (N_2O), an intermediary denitrification product. Several studies have shown that filter-feeding bivalves do increase the rate of nitrogen deposition to the benthos (e.g. Cloern 1982; Officer et al. 1982; Rice 1999b), so it is attractive to hypothesize that this organic nitrogen enrichment of the sediments will result in increased denitrification. Kaspar et al. (1985) showed that sediment denitrification rates increased under long-line mussel farm in New Zealand. However, these experiments deserve replication either in mesocosms or at aquaculture sites with species of bivalves and aquaculture systems that are popular in the United States.

Conclusions and Recommendations

As a general policy for coastal zone management, molluscan aquaculture projects may be overlooked as an environmentally benign form of economic activity. Instead of contributing incrementally to degrading of coastal environments, bivalve aquaculture and shellfish fishery enhancement may in fact be an effective mitigation practice to counteract the effects of excessive nutrient enrichment. Development of aquaculture farms for bivalve mollusks in coastal water bodies most threatened by eutrophication may be a very economical means to mitigate the effects of excessive coastal housing development or other forms of economic activity that discharge excessive nutrients. While growing to marketable size, the bivalves like oysters, scallops, and quahogs will filter the water and deposit organic material to the sediments. The bivalve culturist would be managing stocks for a stable year-to-year harvest, so it is in his best interest to manage his stocks well and add seedstock on a regular basis, thereby keeping the biomass of his shellfish and the rate of population filtration of his animals relatively constant. Additionally, the regular harvesting shellfish for market provides for regular removal nutrients from the estuary.

In addition to private aquaculture development, publicly funded shellfish restoration projects may be a cost effective complementary adjunct to coastal wastewater management strategies and other projects designed to mitigate the impacts of coastal eutrophication. Shellfish beds act to filter particulates from the water column and increase water clarity, so projects to restore eelgrass or other submerged aquatic vegetation may be improved if there were parallel efforts to restore shellfish in adjacent areas. Shellfish beds act to deposit particulate organic nitrogen into the sediments, so there is good reason to suspect that restoring shellfish beds can mitigate eutrophication by promoting sediment denitrification. It may be an interesting exercise for the economics community to estimate the relative costs of a community shellfish restoration project in comparison to the cost of filters and engineered tertiary denitrification systems of a sewage treatment plant processing water at the same rates.

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References

Arundel, J. 2000. *Sewage and industrial effluent treatment,* 2nd ed. Oxford, England & Malden, MA.: Blackwell Science,

Cloern, J. E. 1982. Does the benthos control phytoplankton biomass in South San Francisco Bay? *Marine Ecology Progress Series* 9:191-202.

Cohen, R. R. H., P. V. Dresler, E. J. P. Philips, and R.L. Cory. 1984. The effect of the Asiatic clam *Corbicula fluminea* on phytoplankton of the Potomac River. *Limnology and Oceanography* 29:170-180.

Dahlback, B. and L. A. H. Gunnarsson. 1981. Sedimentation and sulfate reduction under a mussel culture. *Marine Biology* 63:269-275.

Davis, R. C. and F. T. Short. 1997. Restoring eelgrass, *Zostera marina* L., habitat using a new transplanting technique: The horizontal rhizome method. *Aquatic Botany* 59:1-15.

Doering, P. H. and C. A. Oviatt. 1986. Application of filtration rate models to field populations of bivalves: An assessment using experimental mesocosms. *Marine Ecology Progress Series* 31:265-275.

Doering, P. H., C. A. Oviatt and J. . Kelly. 1986. The effects of the filter feeding clam *Mercenaria mercenaria* on carbon cycling in experimental marine mesocosms. *Journal of Marine Research* 44:839-861.

Doering, P. H., J. R. Kelly, C. A. Oviatt, and T. Sowers. 1987. Effect of the hard clam *Mercenaria mercenaria* on benthic fluxes of inorganic nutrients and gasses. *Marine Biology* 94:377-383.

Douglas, B., J. Federico, P. Fohlin, D. Cooper, and B. Heigis. 2000. Developing a comprehensive wastewater management program for Tisbury, Massachusetts. *Environment Cape Cod* 3(1):40-47.

Frostman, T. M. 1996. Constructed wetlands for water quality improvement. *Land and Water* 40(1):14-17.

Gold, A. J., W. Sullivan, and R. J. Hull. 1989. Influence of fertilization and irrigation practices on waterborne nitrogen losses from turfgrass. In *Integrated Pest Management For Turfgrass and Ornamentals*. ed. A.R. Leslie and R.L. Metcalf , 143-152. Washington, D.C.: U.S. Environmental Protection Agency, Office of Pesticide Programs.

Gold, A. J., B. E. Lamb, G. W. Loomis, J. R. Boyd, V. J. Cabelli, and C. G. McKiel. 1992. Wastewater renovation in buried and recirculating sand filters. *Journal of Environmental Quality* 21(4):720-725.

Gowen, R. J. and N. B. Bradbury. 1987. The ecological impact of salmon farming in coastal waters: a review. *Oceanography and Marine Biology Annual Reviews* 25:563-575.

Haven, D. and R. Morales-Alamo. 1970. Filtration of particles from suspension by the American oyster, *Crassostrea virginica. Biological Bulletin* 139:248-264.

Jorgensen, C. B. 1966. *The Biology of Suspension Feeding*. London: Pergamon Press.

Kaspar, H. F., P. A. Gilespie, I. C. Boyer, and A. L. MacKenzie. 1985. Effects of mussel aquaculture on the nitrogen cycle and benthic communities in Keneparu Sound, Marlbourough Sounds, New Zealand. *Marine Biology* 85:127-136.

Kelly, J. R. and S. W. Nixon. 1984. Experimental studies on the effect of organic deposition on the metabolism of a coastal marine bottom community. *Marine Ecology Progress Series* 17:157-169.

Kemp, W. M., W. R. Boynton, R. R. Twilley, J. C. Stevenson, and L. G. Ward. 1983. Decline of submerged vascular plants in upper Chesapeake Bay: Summary of results concerning possible causes. *Marine Technical Society Journal* 17:78-89.

Lucht, J., R. Pendall and L. Joubert. 1998. *Green Hill pond and its watershed: Pollution problems and management options. Fact sheet 98-2,* University of Rhode Island, Kingston: Rhode Island Cooperative Extension.

Manahan, D. T., S. H. Wright, G. C. Stephens, and M. A. Rice. 1982. Transport of dissolved amino acids by the mussel *Mytilus edulis*: Demonstration of net uptake from seawater by HPLC analysis. S*cience* 215:1253-1255.

McClelland, J. W., and I. Valiela. 1998. Changes in food web structure under the influence of increased anthropogenic nitrogen inputs to estuaries. *Marine Ecology Progress Series*. 168:259-271.

Moebius, K. 1880. *The oyster and oyster culture. Report of the U.S. Commissioner of Fisheries 1880:683-751.* Washington, D.C.: U.S. Bureau of Commercial Fisheries.

Mohlenberg, F. and H. U. Riisgard. 1979. Filtration rate, using a new indirect technique, in thirteen species of suspension feeding bivalves. *Marine Biology* 54:143-147.

Morton, T. G., A. Gold, and W. M. Sullivan. 1988. Influence of overwatering and fertilization on nitrogen loses from home lawns. *Journal of Environmental Quality* 17(1):124-130.

Newell, R.I. E. 1981. Molluscan bioenergetics: a synopsis. In *Proceedings of the Second International Conference on Aquaculture Nutrition: Biochemical and Physiological Approaches to Shellfish Nutrition.*, ed. G.D. Pruder, C. Langdon, and D. Conklin, 252-271. Louisiana State University, Baton Rouge: World Aquaculture Society.

Newell, R.I.E. 1988. Ecological changes in Chesapeake Bay: Are they the result of overharvesting the American oyster, *Crassostrea virginica*? In *Understanding the Estuary: Advances in Chesapeake Bay Research. Proceedings of a Conference.* Publication no. 29, 536-546. Chesapeake Research Consortium, Baltimore, MD.

Newell, R I. E. and J. A. Ott. 1999. Macrobenthic communities and eutrophication. In *Ecosystems at the Land-Sea Margin: Drainage Basin to the Coastal Sea.* ed. T.C. Malone, A. Malej, L.W. Harding, N. Smodlaka, and R.E. Turner, 265-293. Washington, D.C.: American Geophysical Union.

Nixon, S. W. 1995. Coastal marine eutrophication: A definition, social causes and future concerns. *Ophelia* 41:199-219.

Nixon, S.W. 1997. Prehistoric nutrient inputs and productivity in Narragansett Bay. *Estuaries* 20(2):253-261.

Nixon, S. W., S. L. Granger, and B. L. Nowicki. 1995. An assessment of the annual mass balance of carbon, nitrogen and phosphorus in Narragansett Bay. *Biogeochemistry* 31:15-61.

Officer, C. B., T. J. Smayda and R. Mann. 1982. Benthic filter feeding: A natural eutrophication control. *Marine Ecology Progress Series* 9:203-210.

Officer, C. B., R. B Biggs, J. L. Taft, L. E. Cronin, M. A. Tyler, and W. R. Boynton. 1984. Chesapeake Bay anoxia: Origin, development, and significance. *Science* 223(4631):22-26.

Palmer, R. E. and L. G. Williams. 1980. Effect of particle concentration on filtration efficiency of the bay scallop, *Argopecten irradians*, and the oyster, *Crassostrea virginica*. *Ophelia* 19:163-164.

Pearson, T. H. and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanography and Marine Biology*

Annual Review 16:229-311.

Peterson, C. H. and R. Black. 1991. Preliminary evidence for progressive sestonic food depletion in an incoming tide over a broad tidal sand flat. *Estuarine, Coastal and Shelf Science* 32:405-413.

Postma, F. B., A. J. Gold, and G. W. Loomis. 1992. Nutrient and microbial movement from s easonally-used septic systems. *Journal of Environmental Health* 55(2): 5-11.

Reid, R.G. B. 1981. Aspects of bivalve feeding and digestion relevant to aquaculture nutrition. In: *Proceedings of the Second International Conference on Aquaculture Nutrition: Biochemical and Physiological Approaches to Shellfish Nutrition.* ed. G. D. Pruder, C. Langdon, and D. Conklin, 231-251 Louisiana State University, Baton Rouge: World Aquaculture Society.

Rice, M.A. 1999a. Control of eutrophication by bivalves: Filtration of particulates and removal of nitrogen through harvest of rapidly growing stocks. *Journal of Shellfish Research* 18(1):275.

——— 1999b. Uptake of dissolved free amino acids by northern quahogs, *Mercenaria mercenaria*, and its relative importance to organic nitrogen deposition in Narragansett Bay, Rhode Island. *Journal of Shellfish Research* 18(2):547-553.

Rice, M. A., A. Valliere, M. Gibson and A. Ganz. 2000. Ecological significance of the Providence River quahogs: Population filtration. *Journal of Shellfish Research* 19(1):580.

Seitzinger, S. P. and S. W. Nixon. 1985. Eutrophication and the rate of denitrification and N_2O production in coastal marine sediments. *Limnology and Oceanography* 30(6):1332-1339.

Seitzinger, S. P., S. W. Nixon, and M.E. Q. Pilson. 1984. Denitrification and nitrous oxide production in a coastal marine ecosystem. *Limnology and Oceanography* 29(1):73-83.

Short, F. T. and D. M. Burdick. 1996. Quantifying eelgrass habitat loss in relation to housing development and nitrogen loading. *Estuaries* 19:730-739.

Tenore, K. R., L. F. Boyer, R. M. Cal, J. Corral, C. Garcia-Fernandez, N. Gonzalez, E. Gonzalez-Gurriaran, R. B. Hanson, J. Iglesias, M. Krom, E. Lopez-Jamar, J. McClain, M. M. Pamatmat, A. Perez, D. C. Rhoads, G. de Santiago, J. Tietjen, J. Westrich and H. L. Windom. 1982. Coastal upwelling in the Rias Bajas, NW Spain: contrasting the benthic regimes of the Rias de Arosa and de Muros. *Journal of Marine Research* 40:701-772.

Twilley, R. R., W. M. Kemp, K. W. Staver, J. C. Stevenson, and W. R. Boynton. 1985. Nutrient enrichment of estuarine submerged vascular plant communities. 1. Algal growth and effects on production of plants and associated communities. *Marine Ecology Progress Ser*ies 23:179-191.

Valente, R. M., D. C. Rhoads, J. D. Germano, and V. J. Cabelli. 1992. Mapping of benthic enrichment patterns in Narragansett Bay, Rhode Island. *Estuaries* 15(1):1-17.

Valiela, I., and J.E. Costa..1988. Eutrophication of Buttermilk Bay, a Cape Cod Coastal Embayment: Concentrations of nutrients and watershed nutrient budgets. *Environmental Management* 12(4):539-553.

Wright, R.T., R.B. Coffin, C. Persing, and D. Pearson. 1982. Field and laboratory measurements of bivalve filtration of natural marine bacterioplankton. *Limnology and Oceanography* 27:91-98.