Environmental Impacts of Marine Fish Farming and their Mitigation

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Abstract

The environmental impact of marine fish farming depends on species cultured, culture method, stocking density, feed type, hydrography of the site, and husbandry practices. In all cultured systems, however, a very large percentage of organic carbon and nutrient input into a marine fish culture system as feed may be lost into the environment through feed wastage, fish excretion, faeces production, and respiration. The high pollution loading have caused considerable environmental concern in many countries, especially in water with limited carrying capacity. Furthermore, the use of chemicals (therapeutants, vitamins, pigments, and anti-foulants) and the introduction of pathogens and new genetic strains have also raised environmental concerns.

Despite the high pollution loadings, results from various studies show that some 23% of C, 21% of N and 53% of P of feed input into the culture system is being accumulated in the bottom sediments and the significant impact is normally confined to within 1 to 1.5 km of the farm. The major impact is on the sea bottom, where high sediment oxygen demand, anoxic sediments, production of toxic gases, and a decrease in benthic diversity may result. Decreases in dissolved oxygen and increases in nutrient levels in the water are normally confined to localized areas, and it is unlikely that fish farming activities will cause eutrophication over large areas. There is also no good evidence to support the suggestion that fish farming would increase the incidences of harmful algal blooms, nor that the present use of therapeutants, vitamins and antibiotics, and the introduction of pathogens and new genetics strains would pose a significant threat to the environment.

Practical ways to mitigate environmental impact of fish farming include keeping stocking density (and hence, pollution loadings) well below the carrying capacity of the water body. Computer simulation and hydraulic models have been applied to estimate maximum stocking density in which water quality could be maintained in a sustainable manner. Pollution loading and environmental effects can also be significantly reduced by improved feed formulation and integrated culture (using macroalgae, filter-feeders and deposit-feeders).

Introduction

Globally, marine fish culture has grown dramatically in recent years, and further growth is expected in the coming decade (New and Csavas, 1995). The rapid growth of the industry has already led to growing concerns over environmental impacts and conflicts with other coastal usage in Europe, North America, Australia, and Asia (Hammond, 1987; Waldichuk, 1987a,b; Morton, 1989; Miki et al., 1992). Indeed, environmental concerns have led to a tighter control measures being introduced in many countries. For example, moratoriums on new developments and tighter control have been introduced in New Zealand, Denmark, Norway, Canada, and Hong Kong (Duff, 1987; Wu, 1988, Morton, 1989; BC Ministry of Environment, 1990). In Scotland and Hong Kong, there is a general tendency to force marine fish-farming offshore (Aldridge, 1988; Wu et al., 1994). This paper reviews our existing knowledge on environmental impact of marine fish farming, and discusses practical ways in which such impact might be mitigated.

Pollution Loading from Farming Activities

Marine fish farming generates high organic and nutrient loadings, mainly from feed wastage, fish excretion and faecal production (Fig. 1). Feed wastage (Ackefors and Enell, 1990; Seymour and Bergheim, 1991) may range from 1 to 38%, depending on the feed type, feed practices, culture method, and species (Fig. 2), and constitutes one of the most important pollution sources. It is noteworthy that feed wastage is much higher in open-sea cage culture systems where trash fish is used as feed. Deposition of organic waste was estimated at 3 kg per m^2 per yr in the vicinity of a farm and 10 kg per m^{-2} per yr or 1.8-31.3 kg C per m^2 per yr underneath (Gowen and Bradbury, 1987). Fluxes and mass balances of C, P and N determined for a salmonid cage farm (rainbow trout fed with dry feed) indicated that 80% of C, 76% of N, and 82% of P of feed input into the system were lost to the environment (Hall et al., 1990; Holby and Hall, 1991; Hall et al., 1992).

Leung *et al.* (1999) constructed a N-budget for groupers cultured in open-sea cages, and estimated

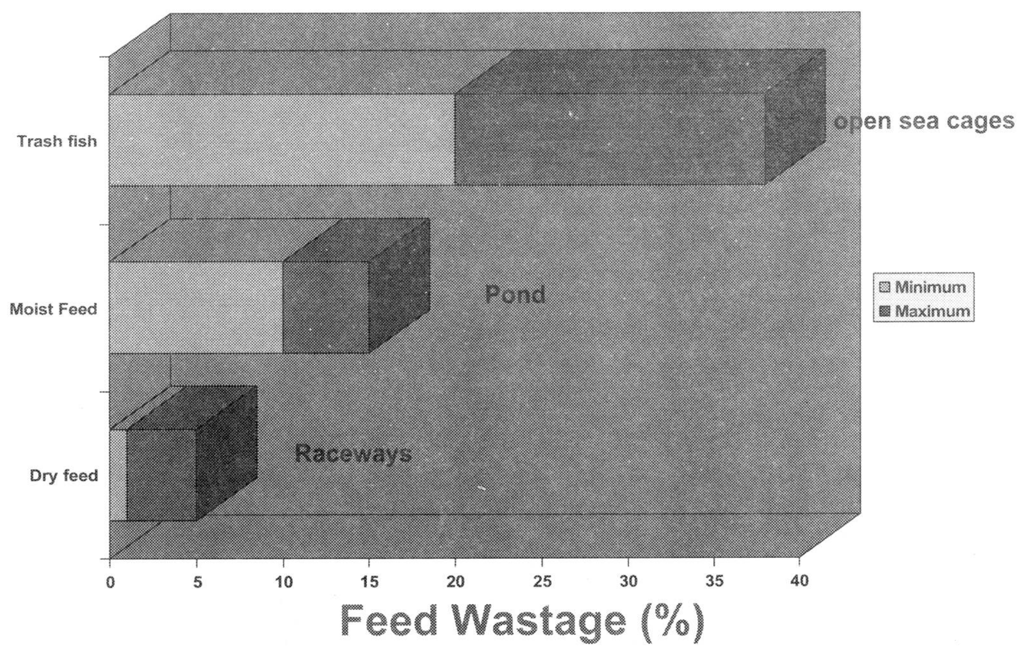

Figure 1. **Feed wastage arising from various culture methods (Warren-Smith, 1982 and Leung *et al.*, 1999)**

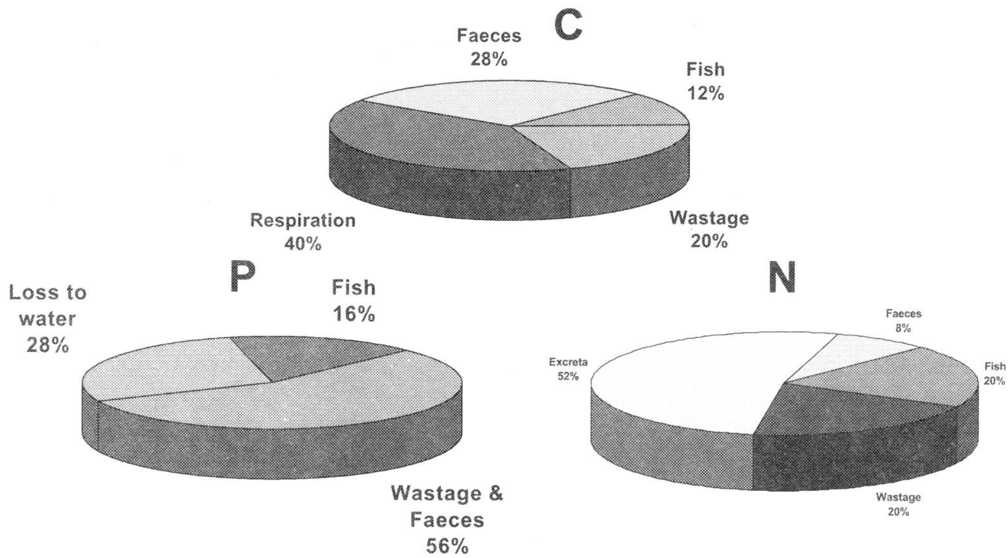

Figure 2. **Fate of C, N and P in land based fish farm (Gowen & Bradbury, 1987)**

that 87.7% of the total N input into the open-sea-cage grouper farm was lost to the environment (equivalent to 321 kg N per t of fish production). Ammonia excretion was the most important N loss (46%), followed by feed wastage (37.7%) and then faecal production (4%). In a European salmonid farm using artificial feed, it was estimated that eventually some 80-84% of C, 52-95% of N and 82% of P in the feed may be lost to the environment. (Fig. 2). The total environmental loss of N for salmonid sea farming was estimated at between 95 and 102 kg N per t production (Hall *et al.*, 1992), and deposits from fish farms covered some 3.8 times the area of the farm itself (Hall *et al.*, 1990).

It was estimated that only 10% of organic matter deposited onto the sediment underneath the salmonid farms is broken down annually (Aure and Stigebrandt, 1990). Sediment fluxes of C, N and P are very low, and 79% of C, 88% of N and 95% of P in farm deposits (equivalent to 23% of C, 21% of N and 53% of P of the feed input) will be accumulated in the sediment and become unavailable. Handy and Poxton (1993) estimated that 59-66% of P in the feed is accumulated in the sediment. Phosphorus can be recycled to water by desorption and biological processes but the release becomes insignificant when the deposit is > 7 cm thick (Hansen *et al.*, 1990). Since N mineralization mainly occurs in oxic surface sediments, the rate would be very slow in deposits underneath farms where the sediment is normally anaerobic and without bioturbation and epifaunal reworking (Rublee, 1982; Huettel, 1990). Indeed, Kaspar *et al.* (1988) failed to detect in-situ nitrification in sediment directly under a salmonid farm.

Vitamins (e.g., B_{12} and biotin), antibiotics (e.g., aureomycin, oxytetracycline, terramycin, furazolidone, and nitrofurazone) and pigments are often added to artificial feed in temperate fish culture. Additions of these chemicals, however, are uncommon in the tropics and sub-tropics where trash fish is predominantly used as a feed. Therapeutants (e.g., malachite green, formalin, copper sulphate, and dipterex) are commonly used to treat fish diseases. Toxic chemicals (metals and sometimes tributyl tin, TBT) are often used to treat cage netting to control fouling (Davies and McKie, 1987; Thrower and Short, 1991), although their use has been banned in many countries since 1990. Alabaster (1982) estimated that the use of therapeutants (aureomycins, oxytetracycline, terramycin, ivermectin, furazolidone, and nitrofurazone) was between 70 and 2000 mg per kg feed. In Norway, the use of therapeutants for fish-farming (nitrofurazolidone, oxytetracycline, oxolinic acid, sulphamerazine, and trimethoprim-sulphadiazine) has increased from 3.7 t in 1980 to 32.6 t in 1988 (Grave *et al.*, 1990); and it was estimated that some 430 g of antibiotics was used in producing 1 t of salmon (Rosenthal *et al.* 1988).

Statistics on the use of chemical therapeutants in other countries is not available, but the quantity used is expected to be large (Chua, 1993). The quantities of therapeutic chemicals released into the environment, however, remain virtually unknown.

The P and N loading generated from fish farming corresponds to a negligible fraction (0.6% and 0.2%, respectively) of overall loadings on coastal areas of Sweden, although local effects may be significant (Ackefors and Enell, 1990). Similarly, the impact of marine fish farming was considered low compared with other waste generating activities along the west coast of Canada (BC Ministry of Environment, 1990). In Hong Kong, where open-cage culture is practised and trash fish is used, BOD and N generated by the mariculture industry constituted about 3% of total loading discharged into Hong Kong waters (Ove Arup *et al.*, 1989). The total discharge of P from a farm with a production of 50 t per yr would correspond to treated discharge from 7000 people, assuming 90% of P is removed from the discharge (Holby and Hall, 1991). However, it should be noted that fish-farm waste is not directly comparable to domestic sewage, mainly because of different C:N:P ratios and significant differences in settleable and soluble wastes (Rosenthal *et al.*, 1988).

Fouling on net cages is often significant, and disposal of fouling biomass into the water after cage cleaning may occasionally add a high pollution loading to the environment. For example, the fouling biomass on fish cages in Hong Kong was estimated at 1.78 t (wet biomass) per t fish production per yr (equivalent to 31 kg BOD, 5 kg N and 70 g P) (Mak, 1982; Ove Arup *et al.*, 1989). Pollution loading could be substantial if these fouling biomass is disposed into the culture water.

Environmental Impact

The environmental impact of fish-farming depends largely on species, culture method, hydrography of culture site, feed type, and husbandry practices. In general, major impact of marine fish culture is on bottom sediment and, to a much lesser extent, on water quality.

Impact on bottom sediment

Organic matters and nutrients derived from fish-farm wastes deposited on the sea bottom may cause an increase in sediment oxygen demand, anoxic conditions, and production of toxic gases (e.g., methane and H_2S) in bottom sediments, thereby adversely affecting benthic organisms (Tucholski *et al.*, 1980; Enell, 1982; Hall and Holby, 1986).

Changes in benthic diversity and soft benthos near fish farms as a result of organic enrichment of sediment and anoxic bottom conditions have been well demonstrated in Norway, Scotland, Japan, and Hong Kong (Olsgard, 1984; Skogheim and Bremnes, 1984; Brown et al., 1987; Tsutsumi et al., 1991; Wu et al., 1994). An azoic zone was typically found underneath the cages and a decrease in benthic diversity occurred in the vicinity of the farm (Ritz et al., 1989; Tsutsumi et al., 1991; Wu et al., 1994). Benthic assemblages were normal between 25 and 150 m away from the cages in which salmonids were fed with artificial feed (Brown *et al.*, 1987; Weston, 1990) while the affected area may extend to 1 km where trash fish are used and flushing is poor (Wu *et al.*, 1994). A recent study by Lu and Wu (1998) showed that benthic recolonization on sediments enriched by fish farm deposit occurred within months, suggesting that fish farming is unlikely to have a long term impact on benthic communities once farming activities are reduced/ceased.

Organic matter settled on the sea bed may lead to the development of anoxic and reducing conditions in the sediment and the production of toxic gases (e.g. ammonia, methane, and hydrogen sulphide). Sediment oxygen demand (SOD) of bottom sediments enriched by fish-farming activities may increase by two to five times (Wu, 1990a; Wu *et al.*, 1994), while total sediment metabolism may be ten times higher (Holmer and Kristensen, 1992). The dramatic increase in SOD may, in turn, cause hypoxia/anoxia in bottom waters.

In general terms, despite high organic and nutrient loadings generated from farming activities, marine fish culture only has a localized effect on bottom sediment, and did not appear to extend beyond a distance of 1-1.5 km from the fish rafts. (Gowen and Bradbury, 1987; Wu *et al.*, 1994). The localized impact may partly be due to the low dispersal of wasted food and faecal materials (Frid and Mercer, 1989; Lumb *et al.*, 1989), and the lock-up of organic matters and nutrients in the sediment.

Impact on water

An increase in the levels of suspended solids, BOD and nutrients (P, organic and inorganic N, total C) and a decrease in oxygen in the water column were generally found around fish farms (Muller and Varadi, 1980; Bergheim *et al.*, 1982; Beveridge and Muir, 1982: Enell, 1982: Penczak *et al.*, 1982; Enell and Lof, 1983; Wienbeck, 1983; Beveridge, 1985; Bohl, 1985; Phillips and Beveridge, 1986; Molver et al., 1988). Eloranta and Palomaeki (1986) demonstrated that phytoplankton biomass, chlorophyll *a* and primary production increase in response to nutrient loading from fish farms. Changes in suspended solids, light extinction coefficient, chlorophyll *a* and phaeopigment were either insignificant or localized (Beveridge *et al.*, 1994; Wu *et al.*, 1994).

In Hong Kong, levels of ammonia, inorganic phosphate, nitrite and nitrate as well as phytoplankton numbers in water column are generally higher, while levels of dissolved oxygen are lower in many fish culture zones (Wu, 1988). The study of Wu *et al.* (1994) at four fish culture zones demonstrated a clear gradient of DO, BOD and nutrients when moving away from the fish farms, and these water quality parameters resembled those of background values 1 km away from the farms.

Nitrogen is considered to be the limiting nutrient for primary production in coastal areas (Gundersen, 1981). Ammonia and urea excreted by fish can be readily taken up by phytoplankton and hence may stimulate their growth. It is noteworthy that fish excreta and waste food have a N:P ration close to 7:1 w/w (the Redfield ratio) (Aure and Stigebrandt, 1990), and hence provide well-balanced nutrients for phytoplankton requirement. Leung *et al.* (1999) estimated that some 53% of nitrogen input into grouper culture system (in open sea culture cages) is lost into the environment as ammonia, and the loading was estimated at 169.8 g ammonical-N per kg production. Un-ionized ammonia is acutely toxic to marine life and toxicity is dependent upon salinity, temperature and pH, and may also promote algal blooms. Phosphorus, on the other hand, is not important in promoting algal growth in the marine environment and, therefore, unlikely to have a significant effect (for a review, see Handy and Poxton, 1993).

Eutrophication caused by cage farming has been documented in several studies (e.g. Enell and Lof, 1983). However, it appears unlikely that marine fish-farming may cause eutrophication on a large scale, although the possibility of localized eutrophication occurring in areas of poor flushing cannot be excluded (Gowen and Bradbury, 1987; Aure and Stigebrandt, 1990; Wu *et al.*, 1994). Although there is laboratory evidence suggesting a relationship between fish-farm discharge and red tides and algal blooms (Nishimura, 1982; Takahashi and Fukazawa, 1982; Molver *et al.*, 1988), there is no conclusive evidence to support that fish farming will promote the occurrence of red tides.

The environmental effects of pigments and vitamins are poorly known. Biotin has been shown to stimulate growth of certain phytoplankton species and is implicated in the toxicity of the dinoflagellate *Gymnodinium aureoles* (Gowen and Bradbury, 1987). Vitamin B_{12} has been shown to be one of the growth-promoting factors of the alga *Chrysochromulina polylepis* (which caused massive kill of caged culture salmon in Scandinavian waters) and the dinoflagellate *Heterosigma akashiwo* (Graneli *et al.*, 1993; Honjo, 1993). Fish meat and faeces have been shown to stimulate the growth of the red tide species *Gymnodinium* type 65 and *Chatonella antigua* in laboratory culture (Nishimura, 1982). Despite these laboratory results, there is no good scientific evidence to relate the field occurrence of red tides to fish farm wastes.

The use of antibiotics in fish farms may lead to the development of resistance in bacterial pathogens of fish, and the possibility of transfer of resistance to human pathogens has also raised concern (Aoki, 1989; Dixon, 1991). The development of a resistant bacterial population in the sediment has been documented (e.g. Austin, 1985; Homer, 1992). For example, up to 100% of oxytetracycline-resistant bacteria have been recorded from marine sediment near fish farms after medication; and resistance persisted for more than 13 months afterwards (Torsvik *et al.*, 1988; Samuelsen *et al.*, 1992). The area of sediment containing oxytetracycline residue however was found to be very localized (< 100m away from the farm). Only trace amount of residue was found in oysters collected near the farms, while levels of residue in crabs were well in excess of the US Food and Drug Administration limit for commercial seafood of 0.1 µg per g (Capone *et al.*, 1996). On the other hand, furazolidone can be rapidly degraded by microbes and hence antibacterial activity was not detectable in sediments (Torsvik *et al.*, 1988; Samuelsen *et al.*, 1991). Inhibition of sulphate reduction in sediment underneath cages after antibiotic treatment has been reported (IOE, 1992). The effects of vitamins on the marine environment are still not well known. In oxic environments, however, the half-lives of most antibiotics and vitamins is short (e.g., <7 days in seawater and 32-64 days in fish-farm sediments for oxytetracycline and biotin) (Samuelsen, 1989; Capone et al., 1996), and the accumulation of vitamins and antibiotics in the environment is highly likely. Although oxytetracycline may be very persistent in anoxic fish-farm sediments (up to 419 days), it is not biologically available in such cases (Bjoerklund *et al.*, 1990). Ivermectin is widely used to treat sea-lice infection in farmed salmonid fish, and high toxicity of this theuraputant has been demonstrated in a number of marine invertebrates, raising concern about its possible adverse impact on the marine biota (Grant and Bigg, 1998). The long half-life (>100 days) of ivermectin and its high toxicity to polychaetes poses a significant risk to marine benthos around fish cages (Davis *et al.*, 1998).

TBT contamination has been identified from the tissues of cultured fish (Davies and McKie, 1987; Waldichuk, 1987b), and water (Balls, 1987; Thrower and Short, 1991) where TBT-treated net pens were used. However, no significant changes in mortalities and growth rates were observed in TBT-contaminated fish (Thrower and Short, 1991). Imposex has also been reported in dogwhelks from sea lochs in Scotland where TBT was used in treating fish cages (Davies *et al.* 1987).

Despite growing concern regarding the spread of disease from farmed fish to wild stock (Hill, 1991), there have been very few documented examples. In most cases, disease identified in one population cannot be positively traced as having spread from another population (Brackett, 1991).

Cultured species may be less adaptable to the natural environment, and escaped cultured fish may inter-breed with the wild stock, thereby altering the gene pool of the latter. However, there is insufficient evidence to ascertain the ecological impact of escaped stock. It is likely that the introduced gene in wild stock might be eliminated by natural selection very quickly.

Mitigating Environmental Impacts

Keep pollution loading under carrying capacity

Environmental impacts of marine fish farming are highly dependent on water circulation, stocking density, husbandry practices, and feed types (Wu *et al*., 1994). Pollution effects were less significant at sites where water circulation was good and culture stock was low, suggesting that the environmental impact of fish farming can be greatly reduced by selecting sites with good water circulation and tidal flushing, and by keeping the stocking density under the carrying capacity of the culture water (Wu *et al*., 1994).

The carrying capacity of the water depends on tidal flushing, current and assimilative capacity of the water body to pollutants. Oxygen consumption of culture species ranges from 83 to > 400 g 02 per t per h (Wu, 1990b; McLean *et al*., 1993). Assuming that dissolved oxygen in seawater is 7 mg O_2 per l, at least 17-57 m^3 of fresh seawater would be required to compensate for the oxygen consumption alone of 1 t of culture fish, not to mention the additional oxygen demand exerted by wastes from the farming activities. In open-water cage culture systems, it has been suggested that an annual production of 200 t fish would require 1 m^3 per sec of current flow (Tervet, 1981).

Once the acceptable limits for water and sediment quality parameters to support fish growth and marine life have been defined, the maximum permissible stock that the defined water/sediment quality should not exceed can be estimated by water quality modeling techniques (Fig. 4). Water quality models have been developed for determining the carrying capacity of water in relation to culture stock and fish culture zones (Wu *et al*., 1999). In this study, two deterministic models (viz., a hydrodynamic model and a water quality model) were used. The first model is a 2-D, 2-layer hydrodynamic model of tidal flow and salt transport, which calculated the water level, velocity, and salinity in each grid cell of 50 m^2 in each layer within the culture zone approximately 30 sec during a tidal cycle. Results from this flow model provide hydrodynamic data for input into a 3-D tidal water quality model, which was run to simulate water quality due to specific pollutant loadings from the marine fish culture operations (Fig. 5). The water quality model quantifies the relationships between major biotic components (i.e., bacteria, phytoplankton, zooplankton, macroalgae, benthos, and fish) and abiotic components (i.e., salinity, organic carbon, dissolved oxygen, nitrogen, phosphorous, and oxygen) at a fish culture site, and also describes the prevailing major biological and chemical processes (including oxidation of organic carbon, nutrient and phytoplankton dynamics, hydrolysis and oxidation of organic and inorganic N, growth, photosynthesis and respiration and decay of plant carbon, SOD, fish respiration, and BOD). Fish biomass and the resulting pollutants (organic waste and nutrients) generated from various activities (e.g., food wastage, fish faeces and excreta, etc.) were quantified and inputted into the model. Based on the input organic and nutrient loadings from a given stocking density, the model calculates the resulting levels of NH_3, NO_2 NO_3, total organic N, dissolved oxygen, and BOD in the receiving water.

The results of the simulation are in close agreement with those from a field study of the same site reported upon earlier (Wu *et al*., 1994). By comparing the output of water quality data under different scenarios of stocking density, the model serves as an effective tool to help management decisions on the maximum fish stock permissible at a particular fish culture site so that acceptable water quality objectives can be met for the sustainable development of the industry.

In Scotland, a suite of simple box model has also been developed to provide a basis for assessing the impact of marine fish farming and regulating farming activities in sea loches (Gillibrand and Turrell 1995).

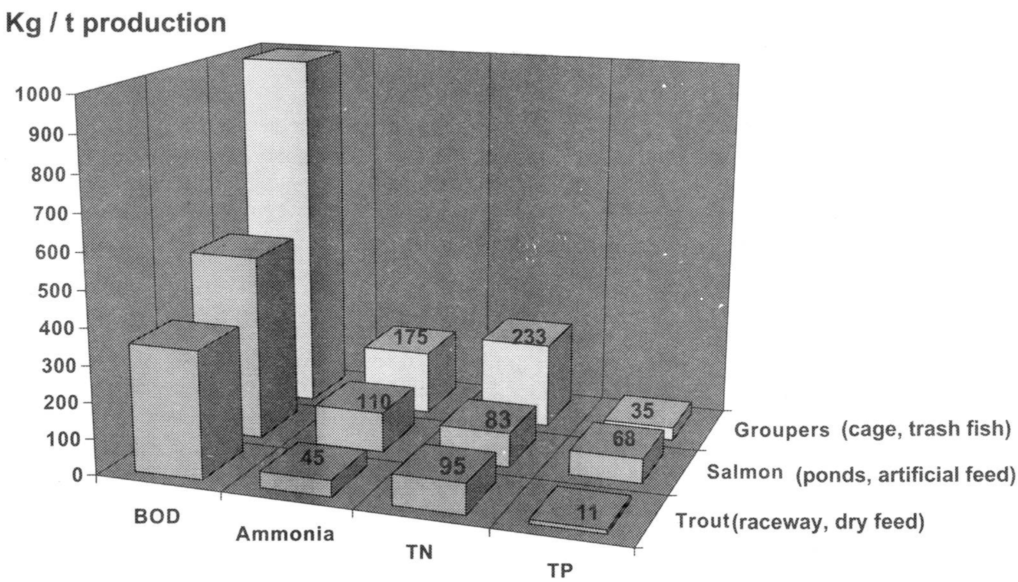

Figure 3. **Pollution loading from various types of marine fish culture**

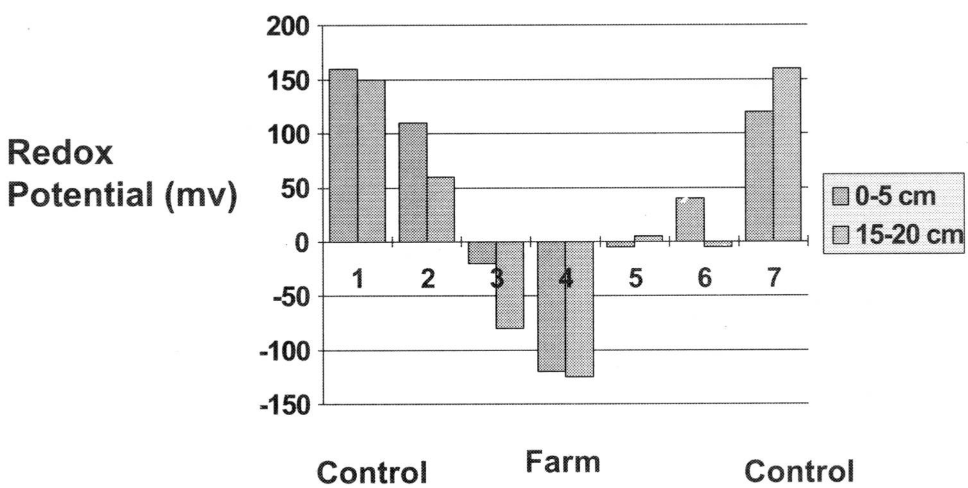

Figure 4. **Changes in redox potential in sediment along a transect across a fish culture Hong Kong (Wu et *al.*, 1994)**

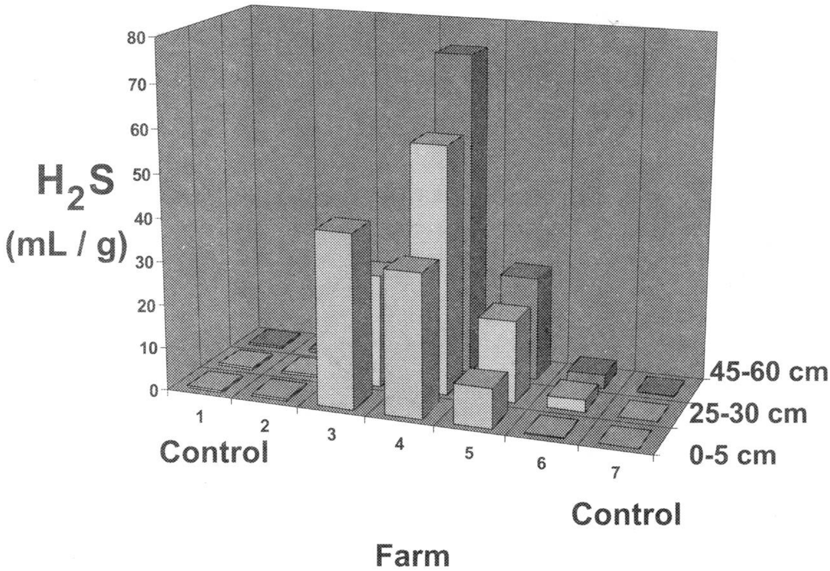

Figure 5. **Changes in hydrogen sulphide in sediment along a transect across a fish culture zone in Hong Kong (Wu *et al.*, 1994)**

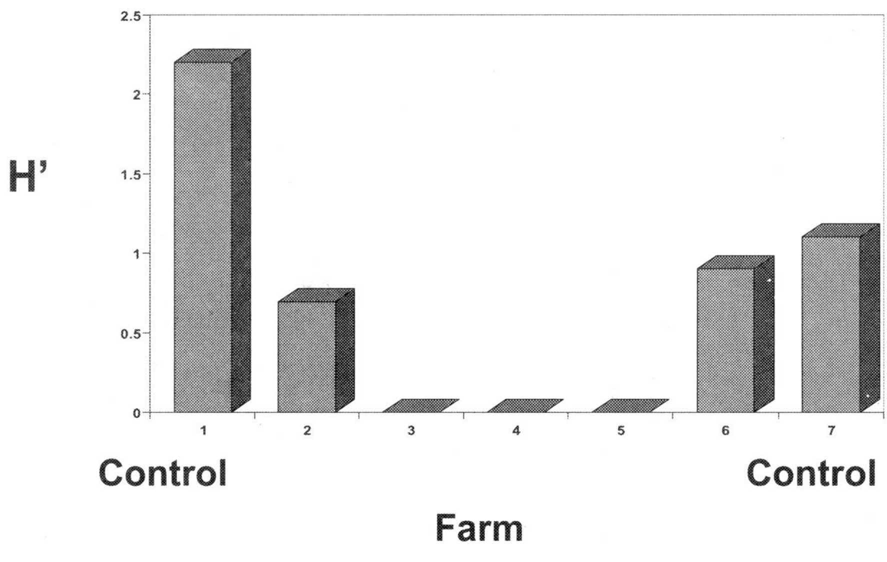

Figure 6. **Changes in diversity index (H[1]) along a transect across a fish culture zone in Hong Kong (Data from Wu *et al.*, 1994).**

Reduce pollution loading by feed manipulation

About 80-84% of C, 52-95% of N, and 82% of P in feed input to the culture system may be lost through feed wastages, excretion and faecal production (Fig. 3), and such losses can be minimized by improved feed technology.

Feed wastage can be reduced by increasing the stability and reducing the sinking rate of feed, and providing the fish with an optimal size of feed at different stages of development. Ammonia excretion by fish is a function of protein intake and can be kept to a minimum; with a highly digestible feed an optimal protein/energy ratio can be provided for each species and its developmental stages. The energy requirements of fish can be satisfied by carbohydrates and fat, so that protein can be spared for body tissue construction. It has been shown that protein retention in *Sparus aurata* can be increased from 24.3 to 31.3% by increasing the dietary lipid by 37% (Kissil and Lupatsch, 1992). Obviously, reduction of N in the diet can only be achieved if artificial feed is used. There is little doubt that formulated artificial feed is superior to trash fish, in terms of its nutritional value, storage, supply, and pollution loading. The reason that trash fish is still widely used in Asia (e.g., Japan, Hong Kong, Singapore, and Thailand) is due to our poor understanding of the nutritional requirements of the various non-salmonid species cultured. This points to the urgent need for research into the nutritional requirements of these non-salmonid species.

Integrated culture

Marine macroalgae can take up N, whereas filter feeders (e.g. bivalves) can remove particulate and phytoplankton from water at remarkable rates. (Inui *et al*., 1991; Prins *et al*., 1994). Harvesting nutrients generated by marine fish farming by macroalgae and filter-feeders would be an attractive option since this would alleviate nutrient pollution on one hand and increase productivity on the other.

The use of mussel/oyster beds to control phytoplankton growth and eutrophication has been suggested by Cloern (1982) and Loo and Rosenberg (1989). Recently, integrated culture in closed or semi-closed culture system using macroalgae (*Ulva* sp. and *Gracilaria* sp.), shellfish and fish has been tried out successfully in Israel, Chile, Canada, France, and Norway (Kaas, 1998). It has been shown that 1 kg of *U. lactuca* can remove 90% of ammoniacal nitrogen in effluents produced by 75 kg of fish and give a maximum yield of 55 g per m^2 per day dry wt in a land-based fish farm (Cohen and Neori, 1991; Neori *et al*., 1991). Jimenez *et al.* (1996) also showed that 153 m^2 of *Ulva rigida* tank surface (at a stocking density of 2.5 g fresh weight per l) is required to recover 100% of DIN produced by 1 t of *Spaus aurata.* Similarly, culturing brown macroalgae (*Laminaria* and *Macrocystis*) near fish farms for nutrient removal was considered to be both economically and technically feasible by Petrell *et al.* (1993). An increase in fish yield (by 1.5%) and dissolved oxygen (by 9%) has been reported from a Japanese open-cage culture system consisting of red sea bream and sea lettuce (Anonymous, 1994).

In a land-based culture system, Shpigel *et al.* (1993) demonstrated that some 63% of N in the feed can be recovered by bivalves (*Crassostrea gigas* and *Tapes semidecussatus*) and seaweed (*Ulva lactuca*) cultured in the same system. Culturing shellfish to remove nutrients derived from farming activities appears to be a viable and practical option and should be adapted to open-cage culture systems. Furthermore, seaweed and shellfish of economic values may be used to improve culture profit.

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